N°: 2016 AGPT 0011

Doctorat AgroParisTech

T H È S E

pour obtenir le grade de docteur délivré par

L'Institut des Sciences et Industries du Vivant et de l'Environnement

(AgroParisTech)

Spécialité : Sciences de l'environnement

présentée et soutenue publiquement par

Paul-Emile Noirot-Cosson

16 février 2016

Optimisation de l'insertion des Produits Résiduaires Organiques dans les systèmes de cultures d'un territoire francilien: évolution des stocks de carbone organique des sols et substitution des engrais minéraux

Directeur de thèse : **Sabine Houot**

Co-encadrement de la thèse : **Emmanuelle Vaudour**

Jury

M. Denis ANGERS, Chercheur, Agriculture et Agroalimentaire Canada (Québec)	Rapporteur
M. Gilles BILLEN, Directeur de Recherche, METIS, UPMC CNRS	Rapporteur
M. Eric JUSTES, Ingénieur de Recherche, AGIR, INRA	Examinateur
Mme Maria ALBUQUERQUE, Chef de Projet, Véolia Environnement R&I	Examinateur
M. Benoît GABRIELLE, Professeur, ECOSYS, AgroParisTech	Président du Jury
Mme. Claire CHENU, Professeur, ECOSYS, AgroParisTech	Examinateur
Mme Sabine HOUOT, Directrice de Recherche, ECOSYS, INRA	Directrice de Thèse
Mme Emmanuelle VAUDOUR, Maître de Conférences, ECOSYS, AgroParisTech	Encadrante Principale

INRA/AgroParisTech

Ecologie fonctionnelle et écotoxicologie des agroecosystèmes

Avenue Lucien Brétignières 78850 Thiverval-Grignon

Abstract:

The use of Exogenous Organic Matter (EOM) in agriculture could be an efficient way to substitute mineral fertilisation and increase soil organic matter (SOM) enhancing soil fertility and storing carbon (C). It could also cause nitrogen (N) pollutions such as nitrate leaching and gas emissions. Better understanding of C and N fate after EOM applications on cropped soils would allow improving these benefits while limiting environmental impacts.

This thesis aims at: (i) predicting EOM impacts when applied on cropped soils, (ii) studying the effects of various scenarios of EOM applications in terms of C storage, synthetic N saving and N pollutions in the context of the Plain of Versailles region (221 km²) and taking into account soil diversity, crop successions and soil organic C contents, (ii) studying the potential for improving these benefits at the regional scale with an optimal distribution of EOM.

The CERES-EGC crop model was used to simulate the effects of repeated applications of EOM over 13 years on both soil C and N dynamics in the soil-crop-water-air system of the long-term field experiment QualiAgro located within the region. The sub-model NCSOIL was parameterised from C and N mineralisation kinetics of EOM measured in laboratory conditions. When transposing the parameters into the CERES-EGC model, C storage at the field scale was well simulated, together with crop N uptake and yields, as well as soil mineral N contents.

The kinetics of C and N mineralisation of the 18 EOM available in the region were used along with EOM biochemical fractionations for parameterising the NCSOIL model. The soil type did not significantly change EOM parameters. Four groups of EOM were distinguished based on their C and N dynamics: (i) stable composts, (2) more reactive and less mature composts and stable manures, (3) manures with reactive OM corresponding to horse manures and (4) very reactive EOM as sludges, litters that should be used as fertilisers.

Numerous scenarios of EOM applications, constrained on the phosphorus and N quantities they bring (and limiting the input in trace elements), were simulated for 20 years in all regional contexts of soil, crop successions and soil organic C contents. The soil type was the main factor controlling C storage and N leaching while it was crop successions for N saving. Some composts allowed C storage up to 1.1 t C ha^{-1} yr^{-1} reaching 63% of C applied. N saving of 74 kg N ha^{-1} yr^{-1} were possible with a dried sewage sludge and a compost. N substitution could reach more than 90% of N applied with EOM, these high percentages being related with the indirect effect of EOM on soil OM and the hypothesis made for N substitution

An optimisation model was developed to select EOM application scenario for each crop plot (soil x crop succession x initial soil OC content x area) accounting for EOM availability in the region with the objectives of maximising C storage or synthetic N saving or minimising N leaching at the regional scale. Applying preferentially the most stable EOM on soils with the highest potential for C storage i.e. with the highest calcareous and clay contents, up to 0.47 t C ha^{-1} yr^{-1} could be stored. Applying preferentially fertilising EOM on crop succession with maize and amending EOM on succession with rapeseed, up to 53 kg N ha^{-1} yr^{-1} could be saved.

Résumé :

L'utilisation des Produits Résiduaires Organiques (PRO) par l'agriculture peut permettre de substituer une partie des engrais minéraux et augmenter les niveaux de matière organique des sols. Elle peut également générer des pollutions azotées via la lixiviation de nitrate ou des émissions gazeuses. Mieux connaitre le devenir du carbone (C) et de l'azote (N) après applications de PRO sur les sols agricoles contribue à améliorer les bénéfices de cette pratique et à en limiter les impacts environnementaux.

Cette thèse a pour objectif: (i) de prédire les dynamiques du C et du N en cas d'applications de PRO sur les sols agricoles, (ii) d'étudier les effets de scenarios d'apports de PRO sur les stocks de carbone dans les sols, la substitution des engrais azotés et les pollutions azotées dans le contexte du territoire de la Plaine de Versailles, en prenant en compte la diversité des sols, de leurs teneurs en C et des successions culturales, (iii) d'améliorer ces bénéfices à l'échelon du territoire via une distribution optimale des PRO.

Le modèle CERES-EGC a été utilisé pour simuler les effets de 13 ans d'apports de PRO sur les dynamiques de C et de N dans le système sol-plante-atmosphère de l'expérimentation QualiAgro, située au cœur du territoire d'étude. Le sous-modèle NCSOIL a été paramétré à partir de cinétiques de minéralisation de C et N mesurées en conditions contrôlées de laboratoire. Le transfert de ces paramètres dans CERES-EGC a permis de simuler correctement les évolution des stocks de carbone dans les sols, les rendements et les prélèvements de N par les cultures ainsi que l'évolution des stocks de N minéral dans les sols.

Les dynamiques de minéralisation de C et N des 18 PRO disponibles sur le territoire ont été déterminées. NCSOIL a été paramétré à partir des caractéristiques analytiques et des résultats de fractions biochimiques des PRO. Les types de sol du territoire n'ont pas impacté significativement le paramétrage. Les PRO ont été classés selon 4 types : (1) composts stables, (2) composts moins matures restant plus réactifs et les fumiers stables, (3) des fumiers très réactifs correspondant à des fumiers de chevaux, (4) des PRO très réactifs tels que des boues et des fientes pouvant plutôt être utilisés comme fertilisants.

De nombreux scenarios d'apport de PRO, contraints par les flux de phosphore et de N apportés, ont été simulés sur 20 ans dans tous les contextes de sols, successions culturales et teneurs en C organique des sols du territoire. Le type de sol a impacté le plus le stockage de C et les lixiviations de nitrate tandis que les successions culturales étaient prépondérantes sur la substitution des engrais. Des composts ont permis de stocker 1.1 t C ha^{-1} an^{-1} allant jusqu'63% du C apporté. Des substitutions de 74 kg N ha^{-1} an^{-1} ont été atteintes avec une boue mais aussi un compost en prenant en compte la substitution des engrais liée aux augmentations de matière organique dans les sols. La substitution des engrais a pu atteindre plus de 90% du N apporté par les PRO, ceci étant lié aux arrières-effets sur le stockage de matière organique dans les sols et dépendant des hypothèses de calcul de la substitution des engrais azotés.

Un modèle d'optimisation a été développé pour sélectionner les scenarios d'apports de PRO pour chaque ilot du territoire (sol x succession de culture x teneur en C organique initiale x surface) tenant compte de la disponibilité des PRO et avec pour objectif de maximiser le stockage de C ou les substitutions en engrais azotés ou de minimiser la lixiviation de nitrate au niveau du territoire. En apportant préférentiellement les PRO les plus stables sur les sols à fort potentiel de stockage de C (fortes teneurs en argile et calcaire), jusqu'à 0.47 t C ha^{-1} an^{-1} pourrait être stocké. En appliquant préférentiellement les PRO fertilisants sur les successions à maïs, et les PRO amendants sur les successions à colza jusqu'à 53 kg N ha^{-1} an^{-1} de N de synthèse pourrait être économisés.

Remerciements

Ce travail de thèse a été réalisé au sein de l'Unité Mixte de Recherche « Ecologie fonctionnelle et écotoxicologie des agroécosystèmes » (ECOSYS) du centre INRA de Versailles-Grignon. Il s'intègre dans les projets PROSTOCK et SOCSENSIT et a été financé par le labex BASC de la Fondation de Coopération Scientifique Paris-Saclay auquel j'adresse mes remerciements pour le financement de cette thèse.

Je tiens d'abord à remercier ma directrice de thèse, Sabine Houot (Directrice de Recherche, INRA-ECOSYS), pour son encadrement et son aide précieuse, pour tout le temps qu'elle m'a consacré en relecture ou au cours de nombreuses réunions qui ont d'ailleurs été toujours agréables et motivantes. Sa sympathie et son humour ont toujours agrémenté les moments partagés au cours de ces trois ans.

Je remercie également particulièrement Emmanuelle Vaudour (Maître de Conférences, AgroParisTech), mon encadrante principale, pour son suivi et sa disponibilité, son aide ne s'est jamais fait longtemps attendre. Merci aussi à elle pour sa gentillesse et ses nombreux encouragements!

Merci aussi à Jean-Marc Gilliot (Maître de Conférence, AgroParisTech), Benoît Gabrielle (Professeur AgroParisTech), Christine Aubry (Ingénieur de Recherche, AgroParisTech), Catherine Ottlé (Chargé de Recherche, LSCE), Laure Bamière (Ingénieur de Recherche, INRA-Economie Publique), Emmanuelle Vaudour et Sabine Houot, tous membres de mon comité de thèse, qui ont été forces de proposition, ont aidé aux choix scientifiques, ont suivi ce travail de thèse et veillé à son bon déroulement.

Merci à Jean-Marc pour ses aides multiples en informatique, en programmation, en SIG, merci aussi pour la réalisation du programme qui m'a servi à réaliser l'ensemble de mes simulations territoriales.

Merci à Benoît pour son expertise avec le modèle CERES-EGC et pour ses aides répétées en modélisation ainsi qu'à Patricia Garnier.

Merci à Laure Bamière pour son aide appuyée pour la réalisation de mon programme d'optimisation avec le langage GAMS.

Merci à Christine pour son aide efficace et son expertise d'agronome sur les systèmes de cultures de la région, ainsi qu'à Marie-Hélène Jeuffroy et Michel Bertrand.

Je tiens à remercier Martin Van Ittersum pour son accueil dans le groupe "Plant Production Systems" à l'université de Wageningen. Merci également à Marcel Lubbers et Argyris Kanellopoulos pour leur aide précieuse en programmation avec le language GAMS.

Je remercie Denis Angers (Agriculture et Agroalimentaire Canada (Québec) et Gilles Billen (METIS, UPMC CNRS) qui ont été les rapporteurs de cette thèse et membres du jury de la soutenance.

Je remercie mes examinateurs: Eric Justes (AGIR, INRA), Maria Albuquerque (Véolia Environnement R&I), Claire Chenu (Professeur, AgroParisTech) et Benoît Gabrielle (Professeur, AgroParisTech) pour avoir suivi ce travail en tant qu'enseignant référent auprès de l'école doctorale

Je remercie tous les membres de l'unité ECOSYS que j'ai pu côtoyer en thèse ou même avant en stage de fin d'étude, mais plus particulièrement ceux de l'équipe sol avec qui j'ai travaillé 3 ans. Merci à Jean-Noël Rampon, Véronique Etiévant, Vincent Mercier à la fois pour toutes les données qu'ils ont permis de récolter et dont je me suis beaucoup servi et à la fois pour leur gentillesse et sympathie au cours de nombreux moments passés ensemble, en pause ou sur le terrain. Merci en particulier à Jean-Noel et ses nombreux gâteaux et fromages partagés et à Vincent pour les discussions "sport". Merci à Myriam Germain pour son appui statistique et sa constante bonne humeur!

Je veux remercier tout particulièrement mes collègues de bureau et associées: Karen Yemadje-Lamoglia, Fiona Obriot, Ophélie Sauzet et Marie Stauffer. Nombreuses ont été les heures passées à travailler côte à côte, à s'aider, se soutenir, mais également à discuter et se divertir. Les journées sans vous ont toujours été plus ternes. Merci d'être venues me rendre visite pendant mon séjour au Pays-Bas et Karen et Adrien pour leur accueil à Tours.

Je tiens à remercier très chaleureusement ma famille, mes parents Céline et Pierre, mon frère Antonin, ma sœur Charlotte et mon beau-frère Vincent pour leur présence, leur amour et soutien inconditionnel. Ma pensée va également à mes nombreux cousins, mes oncles et tantes, mes parrain et marraine et mes grands-parents.

Je remercie également ma deuxième famille, les hockeyeurs du CHSPC en particulier mes coéquipiers et mes coachs, avec qui j'ai partagé des moments aussi nombreux qu'intenses.

Je remercie chaleureusement tous mes amis. Une pensée particulière va à mes amis de l'agro avec qui je ne compte plus les heures passées ensemble: en vacances, en week-ends, en soirées à l'incontournable "Les Marsouins", par emails. Merci aux Vicky, Mono, Lucas, Alice, Hebrard, Boulch et Claire pour leur nombreux accueils parisiens. Merci à Boulch comme promis pour son aide en analyse dimensionnelle. Merci à Mono et ses multiples casquettes d'ami, de collègue, de covoitureur.

Merci à tous ceux qui auront fait l'effort d'écouter parler de PRO et de stockage de carbone pendant un peu plus de trois ans, et notamment ceux venus à ma soutenance. Merci aussi à tous les autres que je n'ai pas cités.

Introduction..... 11
References 19

Chapter 1. Literature Review 21
1. Soil organic matter in croplands 23
1.1. Agronomic role and environmental concern 23
1.2. Dynamics of organic matter in soils 23
1.3. Modelling carbon and nitrogen cycle from SOM turn over 25
1.4. The CERES-EGC/NCSOIL model 27
1.4.1. The CERES-EGC model 27
1.4.2. The NCSOIL model 27
2. Exogenous organic matter use in agriculture 28
2.1. EOM definition and sources of production 28
2.2. Regulatory framework of EOM use: the French case 29
2.2.1. EOM legal qualification as « waste», « by-product» ou « product» 30
2.2.2. EOM legal qualification 30
2.2.3. EOM use 31
2.3. EOM agronomic interests and environmental impact 35
2.3.1. EOM capacity to increase SOM, amending potential 35
2.3.2. EOM capacity to substitute mineral fertilisers 37
2.3.3. Potential environmental impacts 40
2.3.4. Modelling and parameterising EOM 41
3. Spatial distribution and temporal evolution of soil organic matter at the regional scale 42
3.1. Mapping soil organic matter and parameters driving its fate 42
3.1.1. Mapping methodology 43
3.1.2. Mapping issues 44
3.1.3. Mapping methods used 46
3.2. Spatially-explicit EOM impacts on C and N for regional concerns 47
3.3. Regional EOM management 48
4. Conclusion and objectives 49
5. Supplementary data 53
6. References 54

Présentation du territoire d'étude 73
Références 79

Chapter 2. Modelling the long-term effect of urban waste compost applications on carbon and nitrogen dynamics in temperate cropland 81
1. Abstract 83
2. Introduction 84
3. Materials and methods 85
3.1. The QualiAgro long-term experiment 85
3.2. EOM characterisation 87
3.2.1. Physico-chemical analysis and biochemical fractionation 87
3.2.2. C and N mineralisation under laboratory controlled conditions 88
3.3. Model description 88
3.3.1. CERES-EGC model 88
3.3.2. NCSOIL model 89
3.4. Model parameterisation 92
3.4.1. CERES-EGC parameterisation for the QualiAgro experiment 93
3.4.2. NCSOIL parameterisation 93
3.4.3. Evaluation of model goodness of fitting 94
4. Results 95
4.1. Initial parameterisation of Pool II 95
4.1.1. Estimation of initial Pool II size with CERES-EGC 95
4.1.2. Estimation of Pool II C/N ratio with NCSOIL 95
4.2. EOM parameterisation based on lab-scale results and simulations with NCSOIL 96
4.2.1. Results obtained from the NLR procedure 96
4.2.2. Variation of C and N mineralisation with the EOM origin 96
4.2.3. Goodness of fitting of simulated EOM kinetics of mineralisation 99
4.3. Field-scale CERES-EGC simulations over a 13 y-time series 99
4.3.1. Soil organic matter 99
4.3.2. Crop yield and N uptake 103
4.3.3. Soil mineral N 105
4.3.4. N leaching 105
4.3.5. N fluxes and balances 108
5. Discussion 109
5.1. Relevancy of NCSOIL parameterisation methods 109
5.2. Scale-transfer between lab NCSOIL parameterisation and CERES-EGC long-term field simulations 109
5.3. Impacts of EOM applications 111
6. Conclusion 112
7. Supplementary data 114

8. References 117

Chapter 3. Parameterisation of the NCSOIL model to simulate C and N short-term mineralisation of exogenous organic matters in different soils. 125

1. Abstract 127
2. Introduction 128
3. Materials and methods 129
   3.1. Soils and EOM 129
      3.1.1. Soils 129
      3.1.2. Exogenous Organic Matters 130
      3.1.3. Incubation experiments 130
   3.2. Modelling 133
      3.2.1. The NCSOIL model 133
      3.2.2. Model parameterisation 135
      3.2.3. statistical evaluation of soil effect on EOM mineralisation 138
      3.2.4. Factor analysis: relation between EOM parameters and characteristics 138
4. Results 140
   4.1. Variation of C mineralisation kinetics of EOM with soil type 140
   4.2. Classification of EOM for k_{EOM} parameterisation 140
   4.3. Simulation of EOM mineralisation 143
      4.3.1. EOM mineralisation behaviours 143
      4.3.2. Appropriateness of parameterisation methods 144
      4.3.3. EOM characterisation requiring another parameterisation approach 148
   4.4. Use of Van Soest fractionation for parameters and group characterisation 148
5. Discussion 150
   5.1. Parameterisation methods 150
   5.2. General scheme for relating model parameters to measured values 151
   5.3. EOM types available in the “Plain of Versailles” region 151
6. Conclusion 152
7. Supplementary data 154
8. References 156

Chapter 4. Regional-scale scenarios of organic amendment use on cropped soils: impacts on soil organic carbon stocks and substitution of mineral fertilisation simulated with the CERES-EGC crop model 161

1. Abstract 163
2. Introduction 164

3. Materials and methods 167
3.1. Study Area 167
3.2. Set of geographical layers included in the modelling 167
3.3. Characterisation of regional EOM 171
3.4. The CERES-EGC model: overall presentation, parameterisation and running 173
3.4.1. The CERES-EGC model 173
3.4.2. The NCSOIL model 173
3.4.3. Model parameterisation 173
3.5. Building scenarios of EOM application for each cropping system 174
3.5.1. Crop residues and intercrops 176
3.5.2. EOM applications 176
3.5.3. Crop succession constraints for EOM application 177
3.6. Insertion of mineral N saving in the modelling 177
3.6.1. Principle 177
3.6.2. Some additional rules 179
3.7. Running the model 181
3.8. Analysis of the factors driving carbon storage, nitrogen saving and nitrate leaching 181
4. Results 181
4.1. EOM characteristics and consequences on their insertion in scenarios 181
4.2. C storage 182
4.2.1. C storage drivers 182
4.2.2. Differences in C storage 183
4.3. N savings 186
4.3.1. N savings drivers 186
4.3.2. Differences in N savings 187
4.4. Nitrate leaching 192
4.4.1. Nitrate leaching drivers 192
4.4.2. Differences in N Leaching 192
5. Discussion 195
5.1. Simulations of C and N cycles 195
5.2. Nitrogen Savings and other impacts 196
5.3. Uncertainties and how to reduce them 197
5.4. Feasibility of EOM use scenarios at crop plot and regional scale 198
6. Conclusion 201
7. Supplementary data 203
8. References 205

Chapter 5. Optimisation of Exogenous Organic Matter use by agriculture at the regional scale: case study of a peri-urban area near Paris, France 213
1. Abstract 215
2. Introduction 216
3. Material & methods 218
3.1. Study area 218
3.2. Spatially-explicit simulations of EOM applications 218
3.2.1. Spatially-explicit data for model parameterisation 218
3.2.2. Scenario simulations of EOM use 219
3.2.3. Simulations with the CERES-EGC crop model 219
3.3. Optimisation of EOM distribution 220
3.3.1. Preparation of optimisation entries 220
3.3.2. Optimisation model 220
3.3.3. EOM availability situations 221
3.3.4. Optimisation results analysis 222
4. Results 225
4.1. EOM availability situations 225
4.2. Potential impacts related to EOM use at the regional scale 226
4.2.1. Optimised distributions of current EOM (situation A) 227
4.2.2. Impact of composting development (situation B) 228
4.2.3. Impacts on NH_3, N_2O and N_2 228
4.3. EOM distributions 229
4.3.1. EOM distribution per soil types 230
4.3.2. EOM distribution per crop succession 233
4.3.3. EOM application concentration 233
4.3.4. Spatial patterns of EOM distribution 235
5. Discussion 237
5.1. Regional strategies for EOM management 237
5.2. Regional potential benefits and risks 238
5.3. Method reproducibility 239
5.4. Multi-criteria analysis 240
5.5. Applicability 240
6. Conclusion 241
7. Supplementary data. 242
8. References 245

Discussion générale et conclusion 251

Perspectives 261
References 263

Introduction

En 2050, 9,6 milliards d'êtres humains peupleront la planète (United Nations, 2012) et les nourrir constitue l'un des enjeux majeurs du 21e siècle. Les céréales sont les éléments principaux de notre alimentation (FAO, 2004) et sont aujourd'hui en majeure partie produites par les pays industrialisés et en développement au sein de systèmes intensifs forts consommateurs d'intrants chimiques (FAO, 2002; USDA, 2015), mettant généralement en danger la ressource sol et sa matière organique. Par ailleurs, la production de déchets ne cesse d'augmenter parallèlement à la population mondiale. Le recyclage de ces déchets, et en particulier de leur fraction organique, par l'agriculture, pourrait en partie remédier aux dérives induites par les systèmes de production agricole intensifs. Cette thèse s'inscrit dans ce cadre de valorisation des déchets organiques par l'agriculture, et la diminution de son empreinte sur l'environnement. Cette valorisation par l'agriculture est d'ailleurs encouragée en particulier en France par la loi portant engagement national pour l'environnement dit « Grenelle 2 » (article 204-I), la loi de transition énergétique pour la croissante verte (article 7-V), et le programme de recherche international « 4 pour 1000 ».

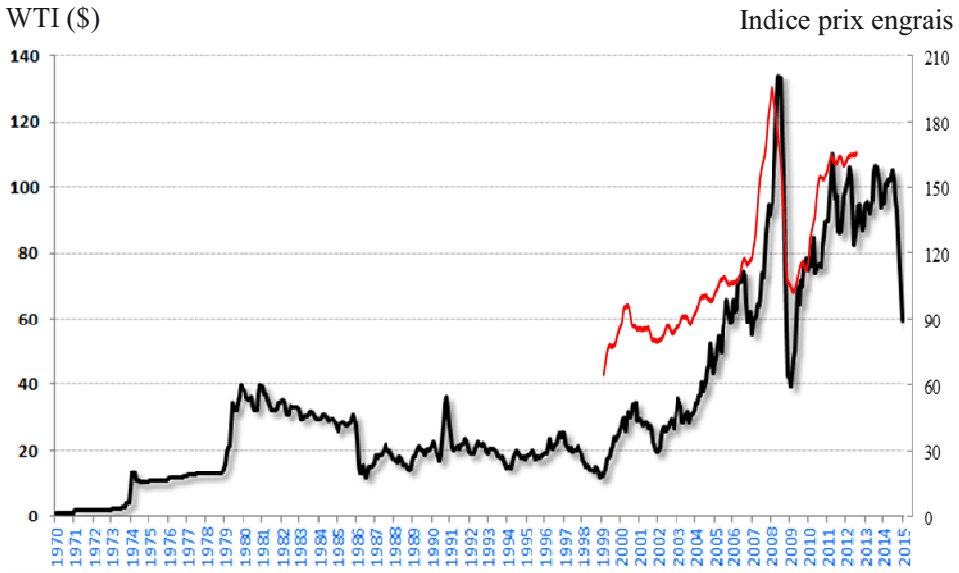


Figure 1. Cours du baril de pétrole (WTI) en dollars courant, moyennes mensuelles, source France-Inflation.com (noir, axe gauche). Indice mensuel brut des prix d'achat des moyens des engrais azotés en base 100 en 2005, source INSEE (rouge, axe droite).

La fertilisation minérale requiert l'exploitation de ressources fossiles, de phosphates et de potasse qu'il faut extraire de mines, ou indirectement, pour la fixation d'azote (N) de l'air, via le procédé d'Haber-Bosch, lui-même très coûteux en énergie fossile (Wood and Cowie, 2004). C'est pourquoi, la fertilisation minérale azotée via, principalement, la synthèse de fertilisant, est dépendante du prix du pétrole, coûteux

et volatile. En effet, le prix du pétrole a fortement augmenté ces dernières années, et a beaucoup fluctué, se répercutant sur le prix de la fertilisation minérale azotée (Fig. 1).

De plus, la production agricole céréalière dans les systèmes de production des pays développés et en voie de développement, s'est généralement spécialisée et souvent territorialisée (Exemple pour la France : Fig. 2a). Autrefois liée à l'élevage, elle bénéficiait des effluents pour sa fertilisation, puis, du fait de l'essor de la fertilisation minérale, elle s'en est affranchie. Ainsi, aujourd'hui, céréaliculture et élevage sont souvent déconnectés territorialement, rendant la céréaliculture dépendante de la fertilisation minérale, et sensible à son coût.

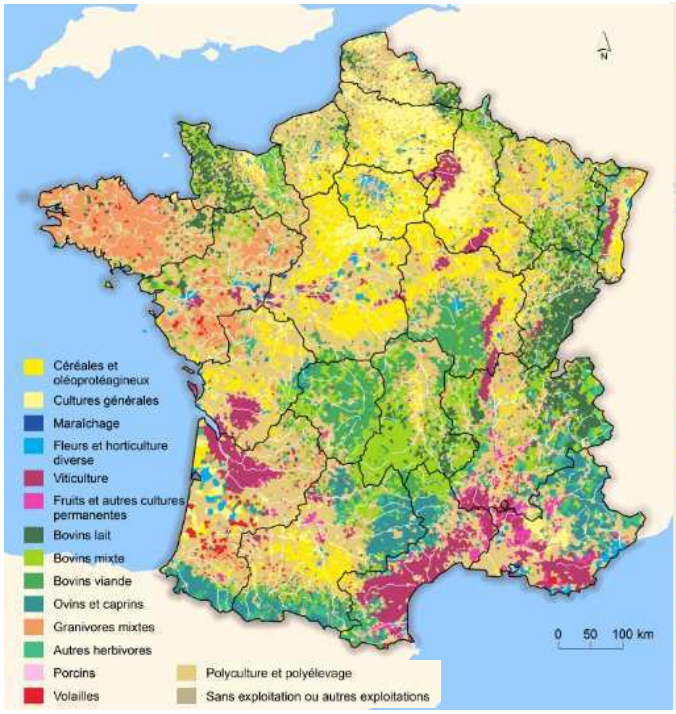


Figure 2a. Répartition des Orientations techniques d'exploitations par commune en France métropolitaine (2010), Source: Agreste, recensement agricole 2010.

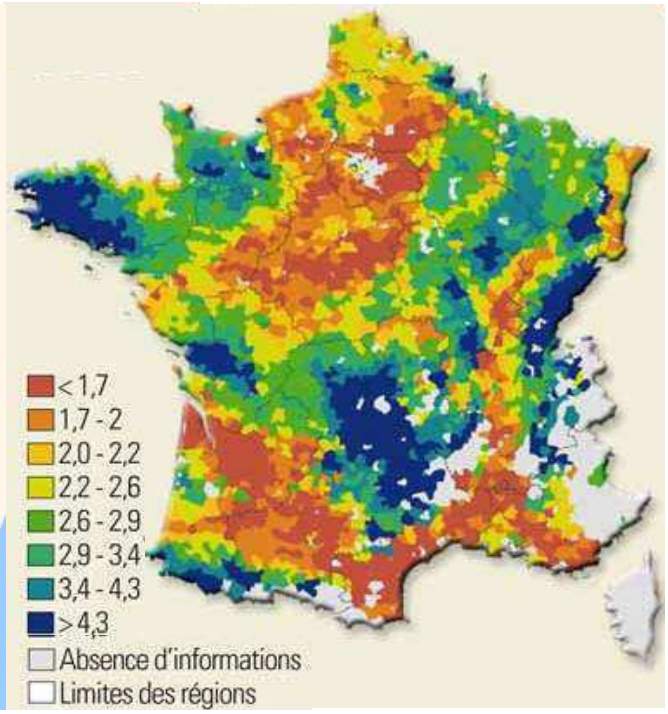


Figure 2b. Teneur en matière organique de l'horizon de surface des sols cultivés, médianes cantonales sur la période 1990-2004, source: La France agricole, GIS, INRA Orléans

La séparation d'une grande partie de cette production céréalière et de l'élevage a aussi eu pour effet une arrêt du retour de la matière organique aux sols cultivés, générant un déclin important de leur teneur en carbone organique (Ciais et al., 2010). La pratique d'exportation des pailles a également participé à cet effet (Balesdent, 1996). Cependant, la matière organique du sol présente de nombreux intérêts en particulier pour la production agricole (Kundu et al., 2007): elle est source de fertilité du sol, elle permet une structuration du sol ce qui a pour effet l'amélioration de la porosité, l'augmentation de la rétention en

eau, de la résistance à l'érosion et à la formation de croute de battance, elle stimule également l'activité microbienne (Chenu, 2002). L'augmentation des stocks de carbone organique des sols signifie une séquestration du carbone atmosphérique, permettant ainsi de compenser les émissions de Gaz à Effets de Serre (GES) et le réchauffement climatique.

Un des moyens pour contrer cette chute et accroître cette matière organique du sol et ses bénéfices potentiels est la pratique d'amendement organique. Elle consiste à apporter aux sols agricoles des Produits Résiduaires Organiques (PRO), sous-produits organiques, d'origine animale ou végétale ou résultant du traitement des déchets, provenant de l'agriculture, des industries agro-alimentaires ou des activités urbaines. Un large panel de PRO existe comme par exemple composts, fumiers, boues, lisiers, fientes… En France, les quantités de PRO produites sont importantes et le recyclage par l'agriculture peut encore progresser (Biomasse Normandie, 2002; ADEME, 2009, 2012; Houot et al., 2014): si les effluents d'élevage, qui constituent la principale source de PRO, avec 297 800 kt/an, sont quasiment totalement réutilisés par l'agriculture, il n'en va pas de même pour les autres types de PRO. Les PRO venant de l'industrie représentent 15 600 kt/an dont la moitié seulement est recyclée, et pour les PRO d'origine urbaine, 75% des 1200 kt/an de boues est utilisé, les déchets municipaux, eux, représentent 38 000 kt/an et ne sont recyclés par l'agriculture qu'à 14%.

Si la céréaliculture en France est bien souvent située à distance des sources d'effluents d'élevage (Fig. 2 et 3), une partie de cette agriculture se localise autour des villes disposant donc d'une manne importante de PRO urbains pouvant ainsi se substituer aux PRO issus de l'élevage et, pourquoi pas, à tout ou partie de la fertilisation minérale. L'utilisation de ces PRO urbains par cette agriculture pourrait donc permettre un recyclage des déchets organiques urbains tout en diminuant la dépendance de l'agriculture à la fertilisation minérale et en augmentant les stocks de matière organique de ses sols et ses effets positifs sur la productivité. Cette synergie créée pourrait également protéger cette agriculture qui subit de plein fouet le grignotage des terres agricoles par l'urbanisation. En effet, sur la période 2006-2010, 78 000 ha de terres agricoles ont été consommés, soit l'équivalent d'un département moyen tous les 4 ans (Safer, 2011).

Néanmoins, l'amendement de PRO sur les sols agricoles malgré tous ses atouts, peut présenter un certain nombre de risques sanitaires et environnementaux. En effet, les amendements organiques peuvent aussi apporter certains éléments polluants pouvant s'accumuler dans le sol, être lixiviés vers les eaux souterraines ou émis dans l'atmosphère, ou se retrouver dans les plantes cultivées. Il peut s'agir de métaux lourds et polluants organiques, de polluants azotés tels que nitrates, ammoniac, oxydes d'azotes, de pathogènes humains ou de gènes résistants aux antibiotiques (Houot et al., 2014).

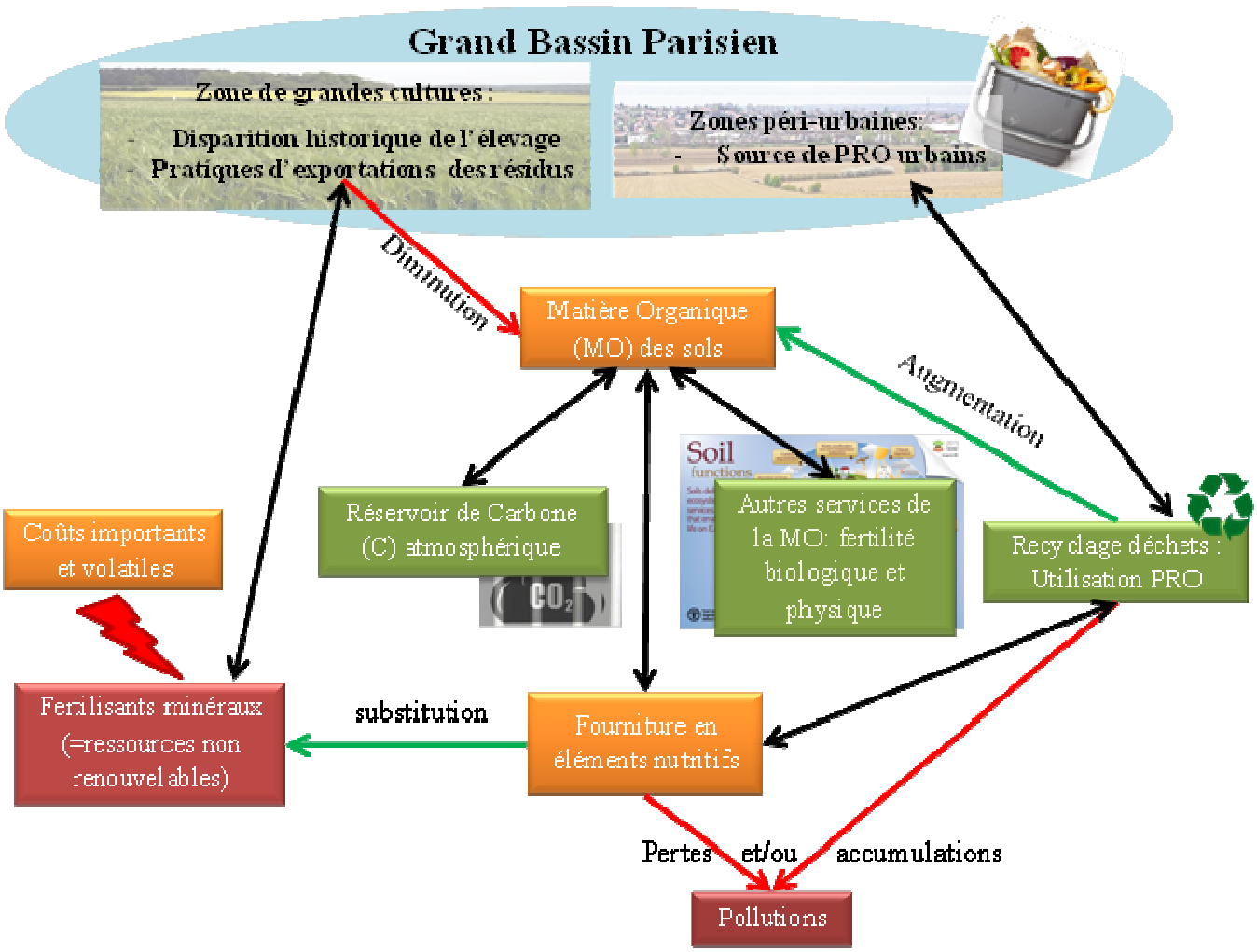


Figure 3. Enjeux autour de matière organique du sol et de l'utilisation des Produits Résiduaires Organiques dans le contexte de l'agriculture du Bassin parisien.
En orange, les éléments structurant du système, en vert les services rendus par le système, en rouge les dysservices.

De plus, les impacts positifs et négatifs de l'utilisation des PRO en agriculture sont très variables, car ils dépendent à la fois du type de PRO et du contexte pédo-climatique ainsi que du système de culture dans lequel le PRO est utilisé. C'est pour cela que les impacts de cette pratique doivent être étudiés et quantifiés pour pouvoir bénéficier au maximum de ses intérêts en prévenant au mieux ses impacts environnementaux et sanitaires. Les caractéristiques des PRO dépendent d'abord de leur nature, des compositions chimiques de leurs constituants, et ensuite des différents traitements qu'ils auront subis: séchage, compostage, méthanisation, chaulage, ainsi que les caractéristiques des procédés de traitements : temps de traitements, température et humidité (Askri, 2015)… La caractérisation au laboratoire des PRO qui doivent être épandus permet leur caractérisation chimique, biochimique voire biologique et permet d'appréhender leurs effets potentiels. Néanmoins, l'étude de leurs effets grâce à des expérimentations au champ s'avère

indispensable pour évaluer leurs effets réels lorsque les PROs sont épandus sur les sols agricoles, tenant ainsi compte du contexte pédoclimatique et cultural. A condition d'être bien calibrés, les modèles déterministes de culture, permettent également d'appréhender les effets contextualisés de diverses pratiques agricoles (Constantin et al., 2015; Therond et al., 2011). Leur utilisation ouvre par ailleurs à l'exploration de nouveaux scénarios d'utilisation de PRO, appréhendant les nombreux impacts potentiels d'une diversité de PRO dans de nombreux contextes, ce qui est impossible à mettre en œuvre par l'unique expérimentation.

Différents contextes se côtoient justement si l'on considère une certaine emprise territoriale, d'autant plus diversifiés que l'emprise spatiale considérée est grande ou le niveau d'organisation spatiale est élevé (Girard et Girard, 2010). La production de PRO étant généralement ponctuelle et le transport de ces produits étant coûteux, la gestion de l'utilisation des PRO doit préférentiellement se faire au niveau d'organisation d'un territoire. A ce niveau, on peut penser, étant donnée la variabilité de contextes et de PRO, qu'il est possible d'améliorer voire d'optimiser la distribution et l'utilisation des PRO dans le sens de maximiser leurs services rendus et de minimiser leurs dysservices.

Cette thèse a pour objectifs de d'évaluer le potentiel de l'utilisation des PRO en termes de stockage de carbone dans le sol, d'économies en engrais de synthèse azoté engrais phosphaté et d'émissions azotées au niveau d'organisation spatiale d'un territoire. Il s'agit d'évaluer les potentiels variés d'insertion des PRO disponibles dans les itinéraires techniques de fertilisation des différents contextes pédo-culturaux du territoire. Ceci est rendu possible par l'utilisation de la modélisation mécaniste. Cette thèse vise d'ailleurs également à mieux comprendre les dynamiques de C et N lors de l'apport de PRO dans le sol notamment grâce aux modèles. Elle a pour but d'améliorer et de simplifier la prédiction de ces dynamiques. Elle a également pour objectif d'utiliser cette modélisation afin de mieux estimer les économies d'engrais azotés réalisables. Enfin, elle cherche à apprécier le potentiel de l'organisation de la distribution de ces produits au niveau du territoire.

La thèse est constituée de 5 chapitres. Le chapitre 1 propose un état de l'art sur le sujet, il introduit les objectifs de ce travail et les raisons ayant poussé au choix de la démarche utilisée. Les 4 chapitres suivants sont rédigés sous formes d'articles scientifiques. Le chapitre 2 présente une étude sur les effets d'apports répétés de PRO sur la dynamique du carbone et de l'azote dans un essai de longue durée localisé au sein du territoire d'étude. Ces résultats ont été utilisés pour paramétrer et valider l'utilisation du modèle de culture CERES-EGC pour simuler les dynamiques d'évolution des stocks de C organique dans le sol et la dynamique du N dans l'agrosystème. Le chapitre 3 concerne la caractérisation au laboratoire du panel de

PRO potentiellement disponible au sein du territoire d'étude et leur paramétrage dans le module matière organique de CERES-EGC. Le chapitre 4 présente la construction de nombreux scenarios d'insertion de PRO dans les systèmes de cultures dans le contexte du territoire et leur évaluation par le biais de la modélisation CERES sur une durée d'une vingtaine d'années. L'évaluation a été basée sur le stockage potentiel de C organique dans les sols, la substitution potentielle des engrais azotés et phosphatés et la lixiviation de nitrates. Le dernier chapitre propose une démarche d'optimisation de l'insertion des PRO dans les systèmes de cultures à l'échelle de ce même territoire. La discussion générale conclut ce travail et introduit à de nouvelles perspectives de recherche.

References

ADEME, 2012. Rapport Déchets.

ADEME, 2009. Les déchets en chiffres en France : Données et Références.

Askri, A., 2015. Valorisation des digestats de méthanisation : effets sur les cycles biogéochimiques du carbone et de l'azote. PhD thesis. AgroParisTech, Paris, France.

Balesdent, J., 1996. Un point sur l'évolution des réserves organiques des sols en France. Etude et Gestion Des Sols, inra.fr.

Biomasse Normandie, 2002. Evaluation des quantités actuelles et futures des déchets épandus sur les sols agricoles et provenance de certaines activités. Lot 3 : Effluents d'élevage.

Chenu, C., 2002. Conséquences agronomiques et environnementales du stockage du carbone dans les sols agricoles, in: Contribution à La Lutte Contre L'effet de Serre. Stocker Du Carbone Dans Les Sols Agricoles de France ? D. Arrouays, Paris, pp. 60–62.

Ciais, P., Wattenbach, M., Vuichard, N., Smith, P., Piao, S.L., Don, A., Luyssaert, S., Janssens, I.A., Bondeau, A., Dechow, R., Leip, A., Smith, P., Beer, C., Van Der Werf, G.R., Gervois, S., Van Oost, K., Tomelleri, E., Freibauer, A., Schulze, E.D., Carboeurope Synthesis Team, 2010. The European carbon balance. Part 2: croplands. Global Change Biology 16, 1409–1428. doi:10.1111/j.1365-2486.2009.02055.x

Constantin, J., Le Bas, C., Justes, E., 2015. Large-scale assessment of optimal emergence and destruction dates for cover crops to reduce nitrate leaching in temperate conditions using the STICS soil–crop model. European Journal of Agronomy 69, 75–87. doi:10.1016/j.eja.2015.06.002

FAO, 2004. L'eau, l'agriculture et l'alimentation [WWW Document]. URL http://www.fao.org/docrep/007/y4683f/y4683f06.htm (accessed 7.16.15).

FAO, 2002. Agriculture mondiale: horizon 2015/2030 [WWW Document]. URL http://www.fao.org/docrep/004/y3557f/y3557f08.htm (accessed 7.16.15).

Houot, S., Pons, M.-N., Pradel, M., Caillaud, M.-A., Savini, I., Tibi, A., 2014. Valorisation des matières fertilisantes d'origine résiduaire sur les sols à usage agricole ou forestier. Impacts agronomiques, environnementaux, socio-économiques. Expertise scientifique collective. INRA-CNRS-Irstea. Synthèse 113pp (http://institut.inra.fr/Missions/Eclairer-les-decisions/Expertises/Toutes-les-actualites/Expertise-Mafor-effluents-boues-et-dechets-organiques#).

Kundu, S., Bhattacharyya, R., Prakash, V., Ghosh, B.N., Gupta, H.S., 2007. Carbon sequestration and relationship between carbon addition and storage under rainfed soybean–wheat rotation in a sandy

loam soil of the Indian Himalayas. Soil and Tillage Research 92, 87–95. doi:10.1016/j.still.2006.01.009

Safer, 2011. Rapport d'orientation 2011, l'action des Safer en zones périurbaines. Congrès des Safer, Paris.

Therond, O., Hengsdijk, H., Casellas, E., Wallach, D., Adam, M., Belhouchette, H., Oomen, R., Russell, G., Ewert, F., Bergez, J.-E., Janssen, S., Wery, J., Van Ittersum, M.K., 2011. Using a cropping system model at regional scale: Low-data approaches for crop management information and model calibration. Agriculture, Ecosystems & Environment, Scaling methods in integrated assessment of agricultural systems 142, 85–94. doi:10.1016/j.agee.2010.05.007

United Nations, 2012. Worl population 2012. Department of Economic and Social Affaires, Population Division.

USDA, 2015. Crop production 2014 summary. USDA, National Agricultural Statistics Service, USA.

Wood, S., Cowie, A., 2004. A review of Greenhouse Gas Emiisions factors for fertilizer production. IEA Bioenergy Task.

Chapter 1. Literature Review

1. Soil organic matter in croplands

1.1. Agronomic role and environmental concern

Soil Organic Matter (SOM) is the largest reservoir of terrestrial carbon (C), the amount of which is estimated between 1500 to up to 2400 Gt (Batjes, 1996; Ciais et al., 2013). It can behave as a sink or a source of CO_2 (Arrouays et al., 2001) depending on climate, land use and soil management (Malhi et al., 1999; Bolin and Sukumar, 2000; Guo and Gifford, 2002) and has a great potential to sequester C and offset CO_2 emissions (Lal et al., 2007; Ciais et al., 2013). Presently, the contemporary rise of the atmospheric CO_2 concentration and its consequence on global warming is commonly admitted (IPCC, 2013): 191 countries have ratified the Kyoto protocol and are thus committed to abate their GHG emissions, including CO_2, to fight against this global threat. SOM in arable soils and its management, especially in croplands, can play an important role for this purpose. Arable soils cover 1708 million hectares in the world (Fischer et al., 2000) and represent more than 15 million hectares in France (Agreste, 2012). One fourth to one third of the world annual increase of CO_2 emissions could be offset by means of those agricultural practices that favour SOC increase (Lal, 2004). More, the use of fossil fuel annually causes 8.9 Gt of C emissions (Le Quéré et al., 2014), which represent about 4‰ of the world soil C stocks. In France, an international research program called “4 per 1000” was launched in 2015 to develop agronomical research following the purpose of an annual C storage in cropped soils.

For croplands, carbon storage into soil has also many other benefits, such as improvement of soil chemical, physical and biological properties (Kundu et al., 2007) leading to enhanced agricultural productivity (Lal et al., 2004) Indeed, an optimum level of Soil Organic Carbon (SOC) content is needed to hold water and nutrients, decrease risks of erosion and degradation, improve soil structure and tilth, and provide energy to soil microorganisms (Lal et al., 2004). Low level of SOC for croplands has become a European threat for soil sustainability (Ciais et al., 2010; European Commission, 2006a), a SOC content of 2% being considered critical for soil in temperate regions (Loveland and Webb, 2003). A European directive aiming at protecting soils and their SOM was even projected (European Commission, 2006b).

1.2. Dynamics of organic matter in soils

Organic matter (OM) in cropped soils comes from organic inputs such as crop residues including roots or rhizodeposition or exogenous organic matter applications (see §2.1). These organic inputs are decomposed and used by soil microorganisms, transformed into stabilised organic material on one side and mineralised into CO_2 on the other side. The SOM composition and dynamics of evolution depend on many factors such as OM inputs characteristics, soil characteristics, climatic conditions, farming practices. Over time, SOM quantity tends toward some equilibrium determined by rates of inputs and decomposition. A

maximum level of SOC seems to exist based on clay and fine silt contents in a soil as used for the concept of SOC saturation (Angers et al., 2011; Six et al., 2002).

Decomposition rate of SOM depends on its quantity: the more SOM content, the quicker the decomposition. Very often the decompositions of organic pools in soil are described by first-order kinetics which rates of degradation depend on their biochemical characteristics (see §1.3). Indeed, chemical and biochemical composition of OM inputs are important drivers of decomposition. Biochemical composition, for example determined by sequential fractionation based on Van Soest's method (Van Soest, 1963; Van Soest et al., 1991; Van Soest and Wine, 1967), describe the composition of an organic input in several fractions: water soluble, neutral detergent soluble, hemi-cellulose-like, cellulose-like, lignin-like components. Such biochemical characterisation has been used for OM inputs rather than for SOM directly. Van Soest's fractions are generally associated with decreasing degradability, and the OM proportion of each component determines the degradability of the whole OM (Morvan et al., 2006). For modelling, Van Soest's fractions are associated with decreasing decomposition rates (Nicolardot et al., 1994; Corbeels et al., 1999), The OM and its degradability in soil can also be characterised through its organic C/N ratio (Nicolardot et al., 2001; Giacomini et al., 2007).

As for OM inputs, SOM can be chemically or physically fractionated, distinguishing homogeneous fractions that can be characterised separately, the proportion of the fractions and their specific properties such as their degradability characterising the whole SOM. Chemical composition can be determined by separation of OM into humic acids, fulvic acids and humins (Stevenson, 1994; Piccolo, 1996), or by separation of non-hydrolysable SOM fraction with acid hydrolysis (Paul et al., 1997; Augris et al., 1998). The OM stability in soils also depends on its association with soil constituents through aggregation and physical protection and interactions with minerals (Chenu and Plante, 2006). The particle size and density fractionation methods consist in dispersing soil more or less strongly and break the aggregates and separate free OM, physically protected OM and OM protected physico-chemically.

Physical parameters such as soil humidity and temperature also drive SOC decomposition. These effects can be described by a number of mathematical functions (Rodrigo et al., 1997). An increase in soil temperature enhances SOC decomposition rate. Temperature effect was first described by an exponential function as by Balesdent and Recous (1997). However, this trend seemed to be valid until a specified temperature (25°C) only and (Valé, 2006) demonstrated that logistical functions were preferable, showing that from a certain level, a temperature increase leads to decrease SOC decomposition. As well as for temperature, soil moisture effect was first described within a too narrow interval of soil moisture and was thus described by a linear or logarithmic increasing function. It has been further described with a logarithmic function in which decomposition rate was maximum at an optimum moisture value that varied from a soil to another (Rodrigo et al., 1997; Leirós et al., 1999).

Soil texture and mineralogy composition are other factors influencing OM decomposition rate: the more clay and calcareous content, the lower this rate. Functions were found by Delphin et al. (1986) and Chaussod et al. (1986) and later validated by Valé (2006) and Saffih-Hdadi and Mary (2008). Soil pH may also have effects on SOC decomposition. A decrease in pH (from 8.3 to 4.5) results in lowering global microbial activity and hence SOC mineralisation (Rousk et al., 2009).

Crop production system also influences SOM dynamics: (i) the grown crops and their yields control biomass production and therefore organic inputs into soil either through root exudates (Rasse et al., 2005) or though shoot and leaf residues left on soils (Abiven et al., 2005; Redin et al., 2014) , (ii) cultivation practices determine shoot and/or leaf residues exportation and soil management. Among farming practices likely to enhance carbon storage in soil, we can mention the following: fertilisation compared to no fertilisation (Alvarez, 2005), more intensive cropping (> 1 crop/year; Young et al., 2009), no-till or reduced till practices compared to tillage (Angers and Eriksen-Hamel, 2008; Luo et al., 2010), residues incorporation compared to exportation (Dersch and Böhm, 2001; Wilhelm et al., 2004) and Exogenous Organic Matter (EOM) applications (Valé, 2006; Diacono and Montemurro, 2010). The latter, in combination with crop type and crop rotation, is decisive for SOM dynamics (Valé, 2006).

SOM is composed of about 58% of SOC (Nelson and Sommers, 1982), the rest being essentially organic N, the quantity of which being defined by the soil organic C/N ratio. Soil C/N ratios generally fall in the range of 9-13 (Kirkby et al., 2011). Indeed, SOM allows storage and destocking through mineralisation of many nutrients such as N that can be then used by plants and soil microorganisms (Rasool et al., 2008). Depending on the rate of OM degradation and its N contents, N is mineralised and becomes available for plant uptake or other N losses in the environment.

1.3. Modelling carbon and nitrogen cycle from SOM turn over

One of the oldest models describing C dynamics is the model proposed by Hénin and Dupuis as early as 1945 (Hénin and Dupuis, 1945), (Fig. 1). It is very simple and requires few parameters only, the SOM content following equation [1].

$$dy = (K_1\omega - K_2 y).dt \qquad [1]$$

where y: the SOM content ($t.ha^{-1}$)
ω: the annual C input to soil ($t.ha^{-1}.yr^{-1}$)
K_1: the fraction of ω that enters into SOM, also called iso-humic coefficient, the remaining being assumed to be mineralised instantaneously, K_2 (yr^{-1}) is the decay rate of SOM, *t*: the time (yr)

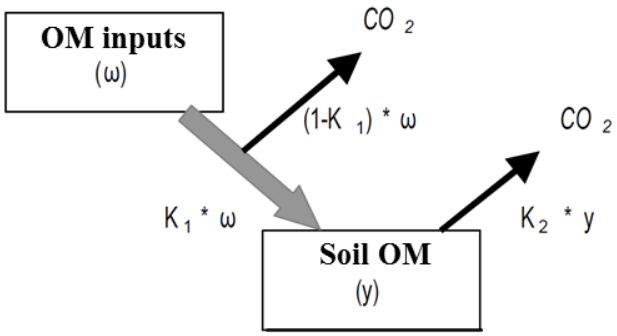


Figure 1. Description of Hénin & Dupuis model, 1945

The K_2 parameter varies according to soil and climatic conditions (Le villio et al., 2001).

From the simple Hénin-Dupuis model, a number of other models have been built to simulate organic matter turnover. Some of them remain simple, focused on carbon, with a limited number of OM compartments (or pools) and parameters. For instance, the AMG model (Andriulo et al., 1999) relies on the Hénin-Dupuis model but the soil OM compartment is subdivided into an active organic carbon and a stable organic carbon pool, the latter having a limited role in C dynamics as its size remains constant over time. Other models as Roth-C (Coleman and Jenkinson, 1996; Jenkinson et al., 1990, 1987), still focused on C, are more complex and composed with a higher number of pools and therefore parameters. They may be a priori capable of better accounting for OM dynamic complexity in soil.

SOM models have then allowed to predict N dynamics by coupling N to C dynamics. Among them, NCSOIL (Hadas and Molina, 1993; Molina et al., 1983; Nicolardot et al., 1994; Molina, 1996) or CANTIS (Garnier et al., 2003) had first been designed for simulating OM dynamics of laboratory measurements, and SOMM (Chertov, 1990; Chertov and Komarov, 1997) for SOM dynamics of forest soil. These first categories of models generally consider effect of climate, soil or cropping systems with simple functions to adapt decomposition rates to various contexts. They do not integrate a plant model but predictions of SOM dynamics over cultivated soils is allowed by manually setting organic inputs.

To better take into account of varying factors influencing organic inputs and SOM decomposition, models simulating SOM dynamics are coupled with deterministic crop models. An AMG-equivalent model (Nicolardot et al., 2001) is incorporated in the crop model STICS (Brisson et al., 2009), NCSOIL in CERES (Gabrielle et al., 2004), CANTIS in PASTIS (Garnier et al., 2003). These crop models simulate processes in the Soil, Water, Atmosphere and Plant compartments also called SWAP models. They are quite similar in their crop growth and development modelling however they may differ in the types of crop they can model. They simulate fluxes more or less precisely and especially water and solutes fluxes. (Addiscott and Wagenet, 1985) distinguished two types of models of water fluxes in soil: (i) mechanistic flux-gradient models and (ii) functional reservoir or capacity based model. DAISY (Hansen et al., 1991; Mueller et al., 1996) and PASTIS are directly based on Richard's equation (i), while STICS and CERES-EGC use semi-empirical Darcy's law i.e. they mixed water tipping budget modelling (ii) and fluxes based on Darcy's law (i). The first modelling (i) allows a fine description of water and solutes fluxes however it requires many parameters to be set and time-consuming calculations. The second is usually sufficient for

simulating water and N fluxes between soil horizons in order to simulate crop growth and major N transformation correctly and its parameterisation is more robust than for the first types of models (Gabrielle et al., 1995). Eventually, such models differ in modelling OM dynamics.

Smith et al., (1997) also tested some of these models along with long-term experiments: CANDY (CArbon-Nitrogen-Dynamics; Franko et al., 1995); DNDC (DeNitrification and DeComposition; Li et al., 1992, 1997); DAISY; CENTURY (Parton et al., 1987; Parton, 1996). These models were tested and generally performed better for the ecosystem they were designed for. The SOMM model was better adapted for forest and grasslands, while CENTURY and DNDC performed better for grasslands. CANDY, DAISY seemed to perform well on all datasets tested. Gabrielle et al. (2002) compared the ability of STICS and NCSOIL to correctly simulate long-term SOC evolution and short-term soil mineral N dynamics in crop fields and they highlighted a trade-off between the two predictions quality. While NCSOIL well simulated long-term SOC evolution, STICS rather performed well for short term simulations of N cycle.

Models also differed on the temporal scale they are designed for. For instance, Roth-C and CENTURY simulate SOM dynamics on a monthly time step being able to predict SOM evolution over centuries, as SOMM. The other models presented here generally simulate on a daily time step being better adapted to predict SOM evolution over months. NCSOIL first devised for short term N and C dynamics can simulate on hourly time step, but it performed well for simulating SOM dynamics in a 4-y field experiment (Gabrielle et al., 2005).

1.4. The CERES-EGC/NCSOIL model

1.4.1. The CERES-EGC model

The CERES-EGC model (Gabrielle et al. , 1995, 2002, 2006, 2005) was adapted from CERES (Jones et al., 1986), a mechanistic model, simulating dynamics of water, carbon and nitrogen in soil-crop systems. CERES-EGC runs on a daily time step and is composed of three main modules. First, a “physical” module for heat, water and solute transfers in soil, as well as plant water uptake and evapo-transpiration governed by climatic demand. Second, a module addresses SOM dynamics in the ploughing layer including microbiological mineralisation and immobilisation being assured by the coupling with the NCSOIL model. Third, a module addresses crop growth through crop net photosynthesis.

1.4.2. The NCSOIL model

NCSOIL simulates plant residues and EOM decomposition after incorporation into soil, nitrification, immobilisation and mineralisation of nitrogen, along with SOM decomposition and formation. It considers four endogenous soil compartments or pools, resulting in 6 pools involved in OM turnover: (i) pool 0, the

"zymogenous biomass", (ii) pool I, the "microbial biomass", (iii) pool II, the "mineralisable SOM", (iv) pool III, composed of highly-humified OM, (v) pool EOM, (vi) pool plant residues. EOM and soil pools (except pool III) are divided into 2 "fractions", one being only decomposed slowly, referred to as "resistant", and the second which is labile, i.e. readily decomposed. Following the same scheme plant residues are divided into 3 fractions, carbohydrates, cellulose and lignin. Each pool fraction is characterised by its C content (mg $C.kg^{-1}$), a decomposition rate (day^{-1}), an organic C to N ratio (C/N). C transformation rates follow a first-order kinetics and the rate of N flows follow the C flows respecting the C/N ratio of a given pool being decomposed, mineral N being added or withdrawn by the following pool to maintain its C/N ratio.

More detailed description of CERES-EGC/NCSOIL is given in chapter 2.

2. Exogenous organic matter use in agriculture

2.1. EOM definition and sources of production

EOM are organic materials, i.e. from degraded vegetal or animal constituents, resulting from agriculture, agro-industries or city activities, The EOM definition generally includes crop residues because of agronomic use and characterisation methods similarities (Marmo et al., 2004), however, here we clearly distinguish EOM and crop residues. EOM can be used in agriculture through applications on cropped soils either directly or after treatments. Various treatments exist: (i) biological treatments, being either composting or anaerobic digestion and (ii) physical or chemical treatments as drying or liming applied to sludges, digestates, slurries or poultry litters.

Composting is an aerobic decomposition of OM into a more stable OM, a process composed of a heating fermentation phase then a maturation phase. Composting results in a loss of weight of about 50% due to water evaporation and organic matter mineralisation, and to a modification of the chemical and biochemical composition of OM. The heating reaching up to 70°C allows pathogens, pests and diseases destruction. The compost obtained is characterised by a higher homogeneity than the raw material, and higher stability. Digesting is an anaerobic decomposition of OM that emits biogas (CO_2 and usable CH_4). The process can be in a liquid or dry way, at various temperatures and the residue named "digestate" can be applied either directly or after post-treatments such as "phase separation" and/or "composting" (Peltre, 2010).

We can classify the different EOM into 3 main types (Dhaouadi, 2014):

- EOM from agriculture, mainly livestock effluents, with manures (mix of effluents and litter, being usually straw), slurries (liquid effluents), poultry litters;

- EOM from cities, with two main sub-types: sewage sludge and waste from cities and household. Urban wastes are often composted and various composts can be found according to the type of composted waste and/or waste sorting: green waste compost, co-compost of sludge and greenwastes, biowaste compost obtained from the fermentescible fraction of household waste and often mixed with green waste, and municipal solid waste from household waste after the removing of packaging.
- EOM from industries, such as agro-industries or chemical industries, sludges issued of industrial waste water treatment plants as for example paper-mill sludges.

In France, EOM produced by livestock farming amount to 297.8 $Mt.yr^{-1}$ (Biomasse Normandie, 2002), from which about 50% is not recyclable because remaining in pastures where animals are breeded. The recyclable part is almost totally reused in agriculture with 109 Mt.yr-1 applied in 2012 (Agreste, enquêtes de pratiques culturales 2011; FranceAgriMer, 2012). In comparison to agricultural effluents, less EOM from industries and cities which represent lower quantities are reused. Indeed, industries generate an organic waste production of 15,600 $Mt.yr^{-1}$, urban sewage sludge represents a production of 1.2 Mt Dry Matter (DM).yr^{-1} and 38 $Mt.yr^{-1}$ for municipal solid waste and they are reused in agriculture in the following proportions: 63%, 75 % and 14% respectively (ADEME, 2012, 2009; Biomasse Normandie, 2002). In 2008, 1.8 Mt DM of organic industrial wastes were applied on fields while 0.3 Mt DM were sent for composting or anaerobic digestion. In 2011, among the 73% of sewage sludge from cities applied on fields, 31% was composted. In 2011, about 40 Mt Fresh Matter (FM) of household waste was used to make 2.2 Mt FM of compost and digestates, 80% of which were applied on field in 2006 (Houot et al., 2014).

2.2. Regulatory framework of EOM use: the French case

EOM and their uses are regulated by superposition of regulations summarised in Table 1 depending on:

- Their status as “waste”, “by-product” or “product”;
- their source either from establisment classified for an environmental protection in accordance with French Law No 76-663 of 18 July 1976 (“Installation Classée pour la Protection de l’Environnement” (ICPE)) or from whatever other origin;
- the location of field application: either in a nitrate-vulnerable zone (“Zone Vulnérable, ZV”), in a nitrate-vulnerable zone with additional remedial measures(“Zone à Action Renforcée, ZAR”) under the nitrates Directive of December 1991 (91/676/EEC) or outside such zones.

2.2.1. EOM legal qualification as « waste», « by-product» ou « product»

2.2.1.1. « Wastes »

A waste is first defined in article L. 541-1-1 of the Environment Code ("Code de l'environnement"). A non-exhaustiv waste list is mentioned in annex II article R. 541-8 of the Environment Code. Some of the organic wastes are regulated by specific dipositions: for instance, straws and agricultural effluents are not qualified as "waste" anymore if used in the farm where they were produced (L. 541-4-1). Sewage sludge are considered as "waste" (R. 211-27) but are also regulated by a supplementary directive (n° 86/278/CEE du 12 juin 1986) relative to the environment preservation and especially soil for sewage sludge use by agriculture. The directive determines maximum levels of heavy metals in soil and sludge. At a national level, sewage sludge regulations result from the law N°92-3 of January 3rd 1992 concerning water, decree N°97-1133 of december 8th 1997 (for the waste qualification principle) and the ministerial order of January 8th 1998 determining technical prescription for sludge application on agricultural soils.
Waste can stop being "waste" (Enckell, 2013) if recycled or treated, in a classified establishment (ICPE) or an establishment regulated by the water law, in order to be reused, as set out in the article L. 541-4-3 in the Environment Code, deriving from the directive transposition 2000/98/CE.

2.2.1.2. « By-product »

A by-product is defined in the article L. 541-4-2 of the Environment Code. Animal dejections from livestock, are by-product, as long as they take part of the farm functioning and they respect regulations as defined in the order of September 8th 2005 of the Commission/Spain C121/03. Livestock effluents are not qualified as waste anymore if they are used as soil fertiliser respecting application rules, whatever the farm origin of these effluents.

2.2.1.3. « Products »

They can also be qualified as "product" if, after treatment, they are subject to normalisation or authorisation procedures, implying respect of quality criterias or standards as regulated by article L.252 from the Rural Code. It concerns product either normalised "NFU" or "CE fertiliser".

2.2.2. EOM legal qualification

EOM production is regulated by laws concerning classified establishments or aquatic environments. According to the article L. 511-1 of the Environment Code, "establishments (…) hazardous or inconvenient for neighbourhood, public health and safety, agriculture, nature preservetion, environments and landscapes (…) are subject to ICPE regulations".

- EOM from ICPE

Waste from ICPE is subject to general ministerial rules set out in the order of February 2nd 1998 also called "integrated order" modified by the order of June 17th 2014. This order set the metallic trace elements (ETM) and trace organic components (CTO) maximum contents that waste and effluents from ICPE must not exceed.

- Livestock effluents from ICPE

The order of February 2nd 1998 does concerned livestock establishment. Three orders of December 27th 2013 set the rules for these establishments. Concerning livestock effluents production, these 3 orders mainly regulate collection and effluents storage.

- Other EOM

Aquatic environment laws mainly regulate livestock effluents (not from ICPE) and sewage sludge (not ICPE).

2.2.3. EOM use

2.2.3.1. General rules

EOM use rules depend on EOM status (waste, by-product, product) and their production regime (ICPE or not). For EOM qualified as "product", use conditions are not very detailed and are mainly determined by producer's indications and possibly completed by ANSES for approved or authorised products. By contrast, for EOM qualified as "waste" and "by-product", the use is highly regulated by ICPE, water and departmental sanitary regulations (RSD)

With the water quality degradation, application regulations were strengthened by transposition of the directive n°91/676 of December 12th 1991 concerning nitrates and "protected area" setting, where EOM application is even more regulated.

2.2.3.2. Nitrate Directive transposition into French law

Regulations impose EOM application surveillance. They imply EOM application monitoring ("Land application plan") and crop plot identification for application (localisation, application perimeter, crop plots possibly amended, crop plots amended). The national actions program set common measures for vulnerable areas that can be possibly completed by regional action program. For instance, the national actions program imposes fertilisation forcast plans assuring the fertilisation balance for EOM application, set out banned period for EOM application depending on their type and cropping systems (Table 1S), limit the quantity of N (organic+mineral) applied with EOM to 170 kg N per ha of utilised agricultural land on average over the farm.

Regional groups of “nitrates” expertise (GREN) were created by decree n°2011-1257 of December 10th to propose technical dispositions for this implementation of the national regulation at the region scale. GREN set out regional references for fertilisation balance calculation.

Table 1. Summary of the french regulatory context for organic amendments

EOM legal qualification	Waste		By-product		Product (authorised, standardised)
Legal regime	From ICPE	From whatever other origin	From ICPE	From whatever other origin	
EOM concerned	Sewage sludge from industry or mixed (Industrial and urban waste water, industrial waste, ashes, not approved compost, not approved methanisation digestates	Urban sewage sludge	Effluents, livestock effluents	Livestock effluents	*NF U42-001/A 10 (organic fertiliser), NF U44-051 (organic amendment composted or not), NF U44-095/A1 (Organic amendment composted)*
Legal regime applicable	Environment Code (ICPE regulation) + ministerial integrated order of February 2nd 1998	Environment Code (Water regulation)	Environment Code (ICPE regulation) + ministerial integrated order of February 2nd 1998	Environment Code (Water regulation) + RSD	Rural Code + Authorisation/standardisation
Responsibility	Producers until/to the final step of desctruction or recycling. Liability if contract with transporter or user (except if application conditions not respected by user)		Producer. Responsibility possibly transferable by contract if effluents given given to another farmer		Producer until on market. User if prescriptions not respected
EOM application regulations	- regulations or ban of (organic or mineral) fertiliser spreading on specific crop plots (high slope, ensnowed, frozen or flooded soils) depending on EOM types. - ban of EOM application nearer than 35m from surface waters. - ban of EOM application on leguminous crops.				
	Prefect can limit the N budget at the farm scale either in cantons in structural N surplus as defined art.R. 211-82 Env. C. in ZAR, or in ZV concerned by pottutions as defined in art. R. 211-81-1 II Env. C. In water catchment areas as defined in art. R. 211-83 Env. C. the prefect can either limit the N quantity brought whatever the origin at the farm scale. In ZV, national actions program imposes: - fertilising forcast plan ensuring N fertilisation balance - banned period for EOM N application (organic and mineral) depending on crop and fertiliser types - effluents storage capacity corresponding a minima to banned period for application - limitations of N quantity applied with livestock effluents to 170 kgN per ha of utilised agricultural land on average over the farm. - bare ground banned during rainy season - regulations or ban of (organic or mineral) fertiliser spreading on specific crop plots (high slope, ensnowed, frozen or flooded soils) depending on EOM types.				
					Possible indications mentioned on the product to be respected

Applied EOM characteristics

- pH between 6,5 et 8,5

- Application Dose determined depending on crop type and yield objective, on crop needs in fertilising elements (majeur, secondaire et oligo-élements)

- Limit of N application (whatever the origin, EOM or mineral fertiliser): 350 kg/ha/an on pastures; 200 kg/ha/an on other crops

- Application Dose determined depending on crop type and yield objective, on crop needs in fertilising elements taking into account other susbtances applied

- pH between 6,5 et 8,5

- Applicable N quantity from establishment animals to be applied or directly applied by animal must not exceed N exportation capacity of crop and pastures

- Limitation of organic and inorganic contaminant contents in EOM and fluxes through application

ETM	Max content (in mg/kg de dry matter)	Max fluxes (g/ha/10ans) except pastures
Cadmium (Cd)	10	150
Chrome (Cr)	1000	15000
Cuivre (Cu)	1000	15000
Mercure (Hg)	10	150
Nickel (Ni)	200	3000
Plomb (Pb)	800	15000
Zinc (Zn)	3000	45000

- Conditions set out by standart must be respected (criteria concerning contents of dry matter, organic matter, N, P, K, Ca, C/N)

Standards NFU 44 051 and NFU 44-095/A1: imply maximal contents of ETM, CTO and pothogens

ETM max concentration	NF U44-051	NF U44-095
	mg/kg Dry Matter	
Arsenic (As)	18	18
Cadmium (Cd)	3	3
Chrome (Cr)	120	120
Cuivre (Cu)	300 (et 600 mg/kg MO)	300
Mercure (Hg)	2	2
Nickel (Ni)	60	60
Plomb (Pb)	180	180
Sélénium (Se)	12	12
Zinc (Zn)	600 (et 1200 mg/kg MO)	600

ETM max fluxes	NF U44-051 & NF U44-095	
	g/ha/yr	g/ha/10yrs
Arsenic (As)	270	900
Cadmium (Cd)	45	150
Chrome (Cr)	1800	6000
Cuivre (Cu)	3000	10000
Mercure (Hg)	30	100
Nickel (Ni)	900	3000
Plomb (Pb)	2700	9000
Sélénium (Se)	180	600
Zinc (Zn)	6000	30000

Physico-chemical characteristics of receveing soils

- Maximum inorganic contaminants contents in soil amended

ETM du sol	Teneurs max (mg/kg)
Cadmium	2
Chrome	150
Cuivre	100
Mercure	1
Nickel	50
Plomb	100
Zinc	300

- pH >= 6 *or* pH>5 + application allows to rise pH beyond 6 respecting max fluxes of inorganic contaminant

2.3. EOM agronomic interests and environmental impact

EOM are composed with two fractions: one mineral, one organic. N, Phosphorus (P) and potassium (K) are the main nutrients required by plants. Where K is only in mineral form, N and P can be in both organic and mineral forms. EOM applications in field directly bring mineral nutrients available for plants, and increases SOM, enhancing all SOM interests and gradually release nutrients through mineralisation. EOM can be considered (i) as fertiliser: the main purpose of their application is to bring quickly available nutrients to plants and therefore substitute conventional mineral fertilisers, or (ii) as amendment: their application first aims at maintaining or even enhancing SOM. Their potential as organic fertiliser or organic amendment (e.g. nitrogen cycle in Fig. 2) mainly depends on their composition and but also on the context of application that influences SOM and mineral N fate. Benefits of this practice may be offset by pollutions such as N losses especially for agricultural systems of developed countries where the large quantities of fertiliser and/or EOM used are likely to strongly impact the environment.

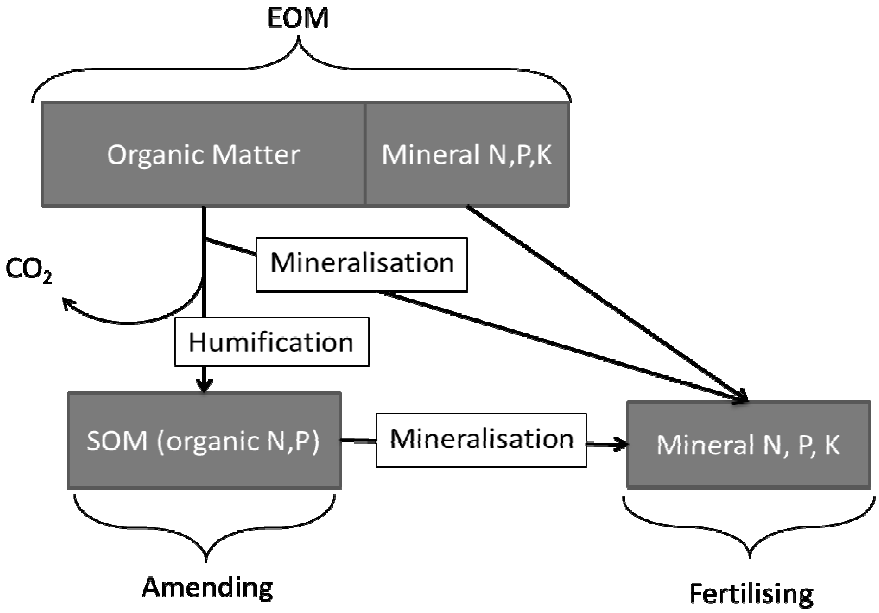


Figure 2. EOM potentials for fertilising and amending. EOM stands for Exogenous Organic Matter, and SOM for Soil Organic Matter. N, P, K stand for nitrogen, phosphorus and potassium

2.3.1. EOM capacity to increase SOM, amending potential

The most reliable estimation of the amending potential is made by measuring SOM with EOM application vs SOM without EOM application in the context of long-term field experiments. Indeed these measurements integrate all factors influencing EOM fate after application in field. Houot et al., (2014) reviewed EOM amending potentials from about 50 long-term field experiments (Table 2). High standard deviations quite important might be due to the large variability of experiment protocols and their contexts: time span, quantity and frequency of application, EOM stability even for specified denomination.

Table 2. Exogenous Organic Matter (EOM) efficiency for soil carbon (C) increase

EOM	Average yield of soil C stock increase in tons of C stored per tons of C applied
Green waste and biowaste compost, livestock effluents	0.41 ± 0.22
Sewage sludge	0.25 ± 0.12
Cattle Manure	0.25 ± 0.13
Slurries (mainly from cattle)	0.21 ± 0.08

In Arrouays et al. (2002), Houot reported different amending potentials with yields of SOC increase by EOM applications of: 0.23 t C/tC with Manures, 0.70 t C/tC with composts of sewage sludge or green wastes, 0.60 t C/tC biowaste compost, 0.30 t C/tC with municipal solid waste composts.

Such experiments are expensive and time consuming, but there are essential as they integrate all factors influencing OM dynamics in field. In France, efforts are being made to share data and standardise experiment protocols for better assessing EOM fate from a network of such long-term experiments (SOERE-PRO network).

Because long-term field experiments requirements may be difficult to meet, the EOM capacity to increase SOM also is evaluated through measurement of the EOM potential of C mineralisation. For this purpose, incubations of soil+EOM mixtures are monitored under controlled laboratory with non-limiting nitrogen and oxygen conditions for soil microbial activity (AFNOR, 2009a). It is generally assumed that EOM mixed to soil does not impact endogenous SOM mineralisation (Mamo et al., 1999; Sikora and Yakovchenko, 1996). Therefore EOM mineralisation is measured from mineralisation difference between incubations with and without EOM, and the amending potential is estimated from the residual EOM carbon at the end of incubations.

Based on such kinetics of C mineralisation under controlled conditions, the I_{ROC} indicator was elaborated (Lashermes et al., 2009). It indicates the proportion of EOM which remains in soil when OM is decomposing at SOM decomposition rate estimated from lab incubation measurements. The I_{ROC} indicator is calculated from biochemical fractionation, determined by stepwise solubilisations according to the Van Soest method (Van Soest, 1963; Van Soest et al., 1991; Van Soest and Wine, 1967) first adapted by Linères and Djakovitch, (1993), then modified and normalised in France (AFNOR, 2009b).

$$I_{ROC}\ (\%\ OC) = 44.5 + 0.5\ SOL - 0.2\ CEL + 0.7\ LIG - 2.3\ MinC3 \quad [2]$$

SOL, *CEL* et *LIG* being the soluble, the hemi-cellulose-like, cellulose-like, cutin-lignin-like fractions (% OM) and *MinC3*, the OC proportion mineralised in the first 3 days of incubation.

Concerning the proportion of C from EOM that will increase the soil humus fraction, Houot et al. (2014) suggested the following sorting by increasing order of soil C storage (and corresponding percentage of C stored), based on a large panel of long-term field experiments: liquid agro-industrial effluents and organic fertiliser (0-20%) < slurries (10-20%) < Non-composted sewage sludge and manures (25-50%) < solid digestates (40-80%) ≤ composts (40-80%) < biochars (80-98%). Lashermes et al., (2009), characterised groups amongst 440 EOM with the I_{ROC} (Fig 3).

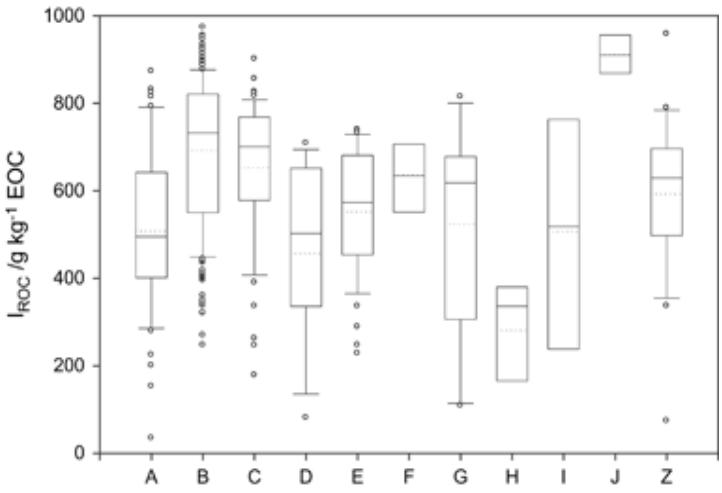


Figure 3. Distribution of the indicator of residual organic carbon I_{ROC}
EOM groups of: sludges (A); urban composts (B); composted manures (C); animal wastes without litter (D); animal wastes with litter (E); mulches (F); plant materials (G); fertilizers (H); animal by-products (I); anaerobically digested wastes (J); and other products (Z). Upper and lower lines of each box correspond to 25th and 75th percentiles, and upper and lower bars correspond to 10th and 90th percentiles. The horizontal continuous line in the box is the median, and the dotted line is the mean; closed circles are extreme experimental data. I_{ROC} expressed in g of residual carbon divided by kg of exogenous organic carbon

Such lab measurements their derived indicators are interesting as they allow a strict comparison between EOM, however, they cannot assess interactions with soil-climate and cultural practices that occur when applied on crop fields.

2.3.2. EOM capacity to substitute mineral fertilisers

This capacity to substitute mineral fertiliser refers to the fertilising potential as described Fig. 2. In EOM, where K is only in a mineral form, P and N can also be contained in organic molecules, however, most of P content is mineral contrarily to N. Therefore, where K and P brought with EOM is mostly available for plant, calculation of EOM potential for mineral fertiliser substitution is more tedious concerning N.

For N, a direct effect often refers to mineral N brought by both mineral N from EOM and mineralisation of organic N the first year of application, while the indirect effect is the N mineralised the years after. This fertilising value can be evaluated thanks to field experiments by estimating indicators such as K_{eq} [3] or the Mineral-Fertiliser Equivalent (*MFE*) which both indicate the EOM efficiency for crop fertilisation compared to synthetic mineral fertiliser, usualy calculated for the first year after application. K_{eq} is calculated from the apparent N use coefficient (*AUC*). The *AUC* [4] is defined as the proportion of N from EOM used by a crop amended with EOM compared to a crop not amended and not fertilised. MFE calculation [5] slightly differs from K_{eq} and AUC, as it represents the N uptake ratio (%) of a crop amended or fertilised over a crop not amended nor fertilised.

AUC = (N uptaken (Crop amended or fertilised) – N uptaken (crop not amended nor fertilised))/ total N applied (from EOM or fertiliser)) [3]

K_{eq} = (AUC_{EOM} x 100)/ $AUC_{fertiliser}$ [4]

MFE = N uptaken (Crop amended and fertilised)/ N uptaken (Crop fertilised) x100 [5]

Gutser et al. (2005) reported the MFE values of EOM (Fig. 4).

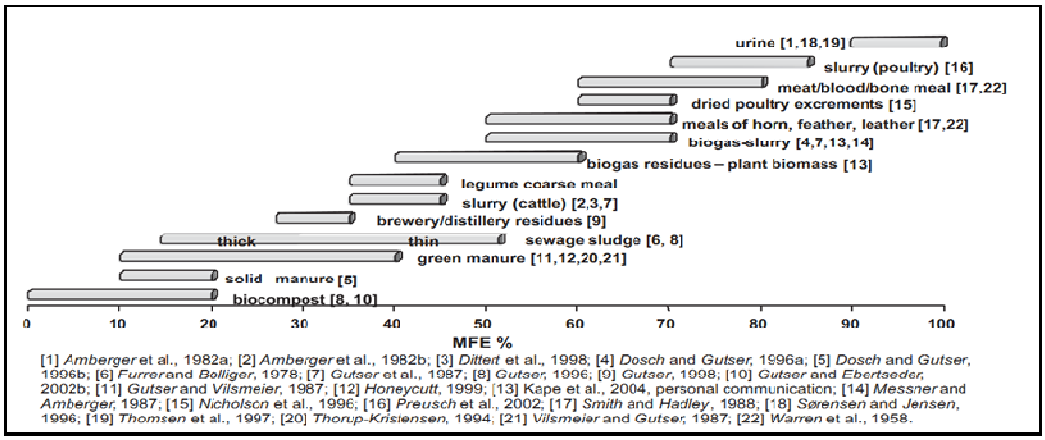


Figure 4. Mineral-fertiliser equivalents (%MFE) for several organic fertilisers characterising N availability in the year of application (Gutser et al 2005)

These indicators for N have practical purposes. Their advantages are a better estimation of the EOM fertilising effect and better adjustment of complementary synthetic fertilisation to avoid yield losses or strong N pollutions. Their values are quite variable for a specified product as its compostion can be variable and its N release vary depending on the pedo-climatic contexts of application.

As for C, measurements of net N mineralisation of EOM+soil mixture incubations, normalised according to the standard (AFNOR, 2009a) is a way to estimate the quantity of N from EOM that will be mineralised and therefore available for crop uptake or for N losses. Indicators to predict EON-N fate are quite rare. EOM organic C/N ratio is one the most used criteria to apprehend the potential of EOM to mineralise. According to Chadwick et al., (2000) N mineralisation can hardly be characterised by other indicator than C/N ratios, however this criteria may not be sufficient for N mineralisation estimate (Ambus et al., 2002), as it also depends on OM stability.

Using these N mineralisation measurements, characterising large EOM types (grouped on the basis of their origin and/or the process they were made with), and/or building EOM groups based on various characteristics and classification methods, to attribute averaged group characteristics could be an efficient tool to help agricultural use.

Lashermes et al. (2010) suggested a classification of EOM potential for N release. It was based on chemical and biochemical fractionation and sorted EOM according to their N mineralised in 91 days of lab incubations (Fig 5). This classification sorts 273 EOM in 6 classes characterised by decreasing N mineralisation ability and composed with: (i) 43% of fertilisers and 43% animal by-products, (ii) 64% of wastewater sludges and 18% livestock farming without litter, (iii) as for (ii) with proportion of 56 and 19%, (iv) 89% of wastewater sludges, (v) 35% of urban composts, 11% manure composts and 17% of livestock farming wastes with litter, (vi) as (v) with proportion of 21, 18 and 39%.

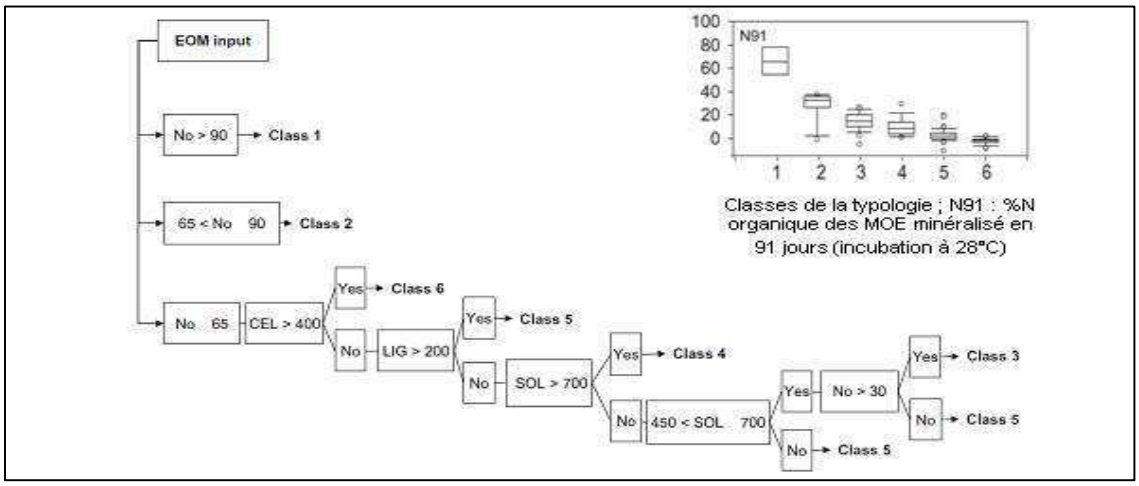
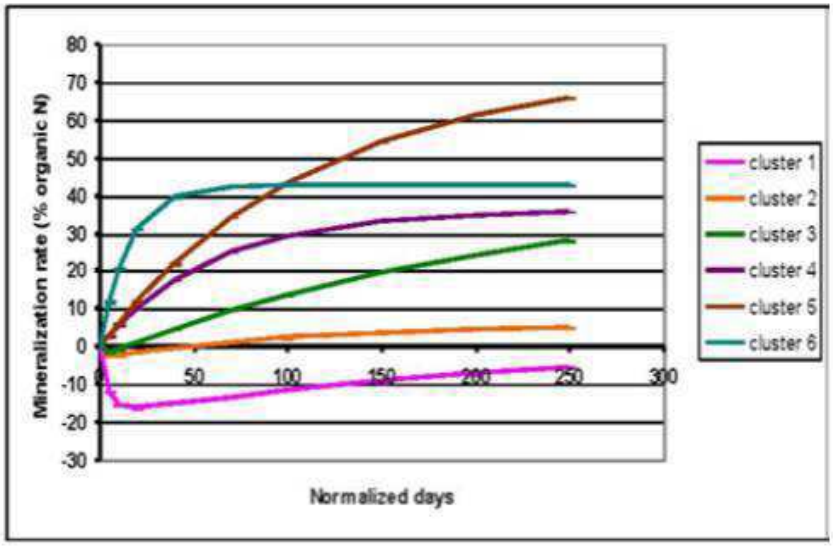


Figure 5. EOM classification procedure based on %N mineralised in 91 days of incubations.
No stands for intial mineral EOM-N content, SOL, CEL and LIG for Van Soest fractions of soluble, cellulose-like and lignin-like components respectively.

Other authors (Bouthier et al., 2009; Parnaudeau et al., 2009) used N mineralisation field measurements and built 6 EOM classes by Ascending Hierarchical Classification (AHC), sorted by increasing short-term N mineralisation ability (Fig. 6). They were composed mostly with : (i) cattle manure composted and green waste composts, (ii) as (i) + green waste and sewage sludge compost, (iii) cattle manure, (iv) poultry litters and dried sewage sludges, (v) concentrated cheap wine, (vi) poultry litters, thick sewage sludge, distillery effluents, starch production effluents.



Cluster	EOM
1	Cattle manures composted, Green wastescomposts
2	Cattle manures composted, Green wastes composts, Green waste and sewage sludgecomposts
3	Cattle manures
4	Poultry litters, Dried sewagesludge
5	Concentratedvinasse
6	Poultry litters, Pasty sewage sludge, Distillery effluents, Starch plant effluents

Figure 6. N Mineralisation of Exogenous Organic Matter (EOM) clusters (% of organic N) (Bouthier et al., 2009)

2.3.3. Potential environmental impacts

Despite interests for crop production, release of P and N can also lead to different form of pollutions: P (Senthilkumar et al., 2012, 2011) and NO_3 leaching (Basso and Ritchie, 2005; Daudén and Quílez, 2004; Torstensson and Aronsson, 2000), emissions of NH_3 (Webb et al., 2010), N_2O (Lessard et al., 1996; Rochette et al., 2000, 2004). They are related to the fertilising potential of EOM as losses of P or N in readily available forms directly decreased the EOM fertilising potentials.

Nitrate leaching can pollute water when reaching groundwaters or rivers: it turns water into non-drinkable and can cause eutrophication i.e. disturbance of aquatic ecosystem balances. It is caused by soil drainage and is increased by high soil nitrate contents synchronous to a period of rains. It can be prevented by inter-crop, generally sowed at the end of summer in order to uptake large N quantities mineralised during autumn that can be lost with winter rains.

Gaseous NH_3 can enhance fine particle synthesis, cause acid rains and water eutrophication. Ammonia volatilisation is a physico-chemical process that can lead to large and quick N losses from EOM mineral N applied. This loss is higher for EOM with large fraction of ammoniacal N, when pH of EOM or soil amended is low, if EOM is applied on bare soil, because of wind or high air temperatures. Gaseous NH_3 can either disturb ecosystems balance after deposition or enhance fine particle synthesis

Nitrous oxide is a powerfull GHG. N_2O derived from denitrification, a biological process of anaerobic decomposition, allowed by water satured soil and enhanced by high soil nitrate contents.

NO_3 leaching, emissions of NH_3, N_2O can be measured in laboratory and field. N_2O emissions measurements are described by Laville et al., (2014). EOM N_2O emissions represented 0.28% of the total

N applied for pig and cattle manures, 0.57% for pig and cattle slurries and 0.75 for poultry effluents (Chadwick et al., 2011). NH_3 emissions field measurements are described in Cohan et al., (2013) and NO_3 leaching field measurements by Jemison and Fox (1992, 1994).

Characterisation of all these impacts of EOM use together is quite challenging, as field measurements are generally tedious and specific of one soil-climate-cropping system context while quicker approximations thanks to lab measurements, or through EOM classifications allow characterisation of a wider range of products but do not take environmental factors of influence into account. Using mechanistic models can be a tool to overcome these issues.

2.3.4. Modelling and parameterising EOM

Crop models successfully simulated C and N impacts of EOM applications, especially SOC stocks evolution: STICS (Brisson et al., 2003; Dhaouadi, 2014), DAISY (Bruun et al., 2006; Gerke et al., 1999; Hansen et al., 1991); PASTIS (Chalhoub et al., 2013), CERES–EGC (Gabrielle et al., 2005), Roth-C (Peltre et al., 2012), DNDC (Sleutel et al., 2006). Indeed, correctly simulating mineral N cycle as affected by EOM application is more challenging (Gabrielle et al., 2005). Crop mechanistic modelling requires nevertheless to be parameterised and calibrated. For this purpose data from long-term field measurements should be used, however, as many factors influencing OM decomposition are involed and thus many parameters, parameterisation/calibration is not straightforward.

C and N mineralisations measurements in controlled conditions of soil+EOM mixture incubations are often used along with SOM dynamics modelling. They were first used adjust the following simple mathematical model of EOM degradation [6].

$$C_{residual} = C_l.\exp(-k_l.t) + C_r.\exp(-k_r.t) \qquad [6]$$

With $C_{residual}$ the organic C remaining after a certain time t (day) of incubation, C_l a labile fraction of the organic C and C_r the resistant fraction (equal to 100-C_l). $C_{residual}$, C_l and C_r are expressed as the % of intial organic C. k_l and k_r the decomposition rates of the labile and resistant fractions respectively expressed in day^{-1}.

More complex SOM models were also used to describe EOM C and N mineralisation dynamics in lab conditions: STICS-Residues (Lashermes, 2005; Nicolardot et al., 2001), CANTIS (Chalhoub et al., 2013; Garnier et al., 2003), NCSOIL (Antil et al., 2011). Modelling and simulating EOM fate in such conditions enable to understand EOM mineralisation dynamics, then relate mineralisation behaviours to physical, chemical or biochemical EOM characteristics and discriminate EOM through such characteristics (Antil et al., 2011; Burger and Venterea, 2008; Corbeels et al., 1999; Lashermes et al., 2010, 2009; Thuriès et al.,

2002). NCSOIL was already used for many types of EOM (Antil et al., 2011; Beraud et al., 2005; Burger and Venterea, 2008; Corbeels et al., 1999).

Fitting C and/or N mineralisation of such incubations allowed to parameterise these SOM models, may be a first step prior to extrapolation to field simulations by transposing their related parameters into crop models (Houot et al., 2014). Among studies modelling long-term EOM effect in field: some authors used optimised EOM parameters based on lab measurements (Chalhoub et al., 2013), while others used lab made indicators (Gabrielle et al., 2005; Peltre et al., 2012). This transposition may not be straightforward. For instance, Lashermes et al. (2009) found some bias in carbon storage after EOM applications when comparing field measurements to direct predictions with I_{ROC}.

EOM applications in crop fields are likely to benefit crop production however they can also bring environmental damages despite regulations on « EOM » products and their use. Assessing these impacts of EOM use by agriculture is necessary to enhance the agricultural practice and its benefits, and this assessment can be done with models that can take into account a number of factors influencing EOM fate in soil after application. More, EOM is a product that cannot be easily transported, so that EOM distribution can be managed at small scale only, such as a region of a limited area. At a regional scale, EOM constitutes a limited resource and factors influencing EOM fate in soil are spatially variable. Therefore, impacts of EOM use should be spatially assessed in order to apprehend a better EOM use and a better EOM distribution at the regional scale.

3. Spatial distribution and temporal evolution of soil organic matter at the regional scale

Studying the fate of SOM, and particularly the effect of EOM application on it, at a regional scale have two purposes: (i) estimating global potential of the region in terms of carbon storage or of N releases, and (ii) orienting the practices in order to enhance their benefits or limit their risks taking into account spatial variability of SOM and its controlling factors.

3.1. Mapping soil organic matter and parameters driving its fate

To simulate regional SOM fate with crop models, the initial state of SOM and of parameters controlling its fate such as soil characteristics or crop production systems or climatic parameters need to be all determined in a spatially-explicit way to infer models. The mapping of SOC was widely studied (Minasny et al., 2013) and mapping methods and issues were studied here mainly through this variable.

To study SOC stocks (SCS) spatial estimate, a little review was made based on 64 studies modelling spatially SCS or SOC contents (Ardö and Olsson, 2003; Arrouays et al., 2001; Arrouays and Pelissier, 1994; Batjes, 2008; Bellamy et al., 2005; Bergstrom et al., 2001; Bernoux, 2002; Bliss and Maursetter, 2010; Cerri et al., 2007; Chaplot et al., 2010; Cheng et al., 2004; Chuai et al., 2012; Davidson, 1995; Davidson and Lefebvre, 1993; Dou et al., 2010; Frogbrook et al., 2009; Gnanavelrajah et al., 2008; Grimm et al., 2008; Guo et al., 2006; Homann et al., 1998; Howard et al., 1995; Huang et al., 2007; Kern, 1994; Kumar et al., 2012; Lettens et al., 2005a, 2005b, 2004; Liu et al., 2006, 2011, 2013; Martin et al., 2011; Matsuura et al., 2012; McGrath and Zhang, 2003; Meersmans et al., 2008, 2009, 2011, 2012; Mishra et al., 2009, 2010; Mishra and Riley, 2012; Mora-Vallejo et al., 2008; Muukkonen et al., 2008; Nyssen et al., 2008; Saby et al., 2008; Santra et al., 2012; Schwartz and Namri, 2002; Simbahan et al., 2006; Simón et al., 2013; Sleutel et al., 2003; Spielvogel et al., 2009; Tan et al., 2004; Tornquist et al., 2009; Ungaro et al., 2010; Vågen and Winowiecki, 2013; Wang et al., 2012; Xu et al., 2011, 2012, 2013; Yoo et al., 2006; Yuan et al., 2013; Yu et al., 2007; Zhang and McGrath, 2004; Zhao et al., 2005; Zhou et al., 2003).

3.1.1. Mapping methodology

SOC stocks (SCS) are often defined as the mass of organic carbon within a unit area of soil for a given depth. SCS cannot directly be measured but result from the following calculations, either eq. 7 or eq. 8:

- Stock = d × C × BD × (1-RF) / 100 (Goidts et al., 2009) [7]
- Stock = d × C × BD_{fe} (Don et al., 2007 ; Holmes et al., 2011) [8]

Where Stock is the SCS (here expressed in t $C.ha^{-1}$), d the sampling depth (m), C the soil organic carbon concentration (g $C.kg^{-1}$), BD the bulk density ($kg.m^{-3}$), and RF the mass proportion of rock fragment content (dimensionless). BD_{fe} is the fine earth bulk density.

The stock equation is composed by various parameters the values of which need somehow to be mapped for SCS mapping. In that sense, it is comparable to the set of parameters controlling SCS fate that all need to be mapped on the same spatial-extent for spatially-explicit simulations of SCS dynamics.

We can distinguish two main approaches to spatially infer SCS (Fig. 7): (i) stocks may first be computed for

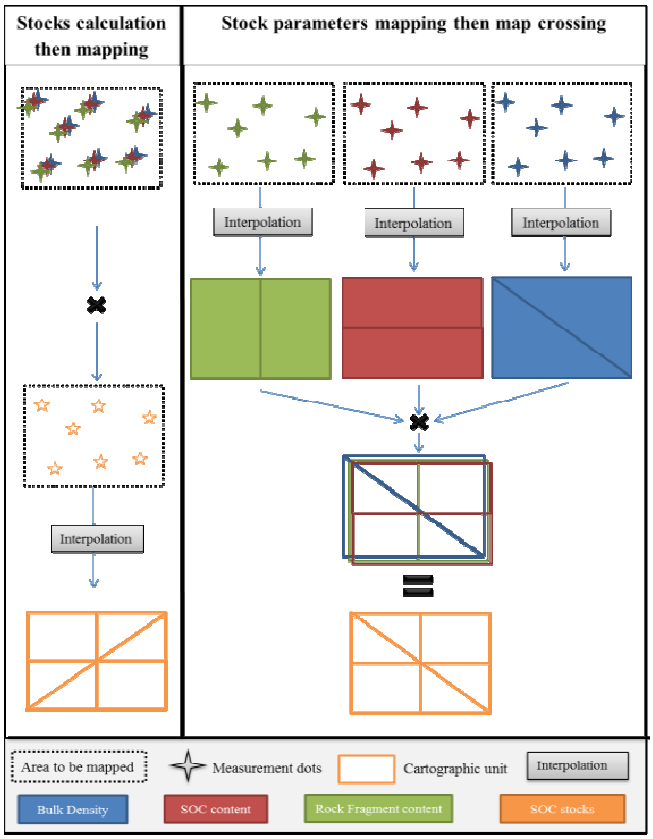


Figure 7. The 2 main approaches for SCS mapping

a number of individual measurement sites then spatially inferred to the whole territory studied or (ii) parameters of the stock equations may be mapped separately then maps intersected to obtain the SCS map. This method (ii) allows mapping of SCS with parameters measurements or estimates made at the different locations (heterogeneous datasets) and with different interpolation methods depending on parameters. Among the interpolation methods, some of them consist in extrapolating estimate to specific areas (cartographic units), in that case, the choice of the cartographic units (the delineation) is of great importance and might also be made differently from a parameter to another. Where for SCS the equation parameters can be measured or estimated at the same locations, concerning parameters of SCS fate datasets are obviously heterogeneous.

Three main methods are used to spatially infer SCS: (i) thematic map design (M methods), (ii) spatial statistics and geostatistical methods (G methods), (iii) knowledge-based models and regressions (R methods). A lot of authors also tested hybrid methods for SOC mapping (Huang et al., 2007; Meersmans et al., 2012; Mishra et al., 2010; Mora-Vallejo et al., 2008; Xu et al., 2011). (i) The M method, which is the most common, consists in defining areas being either cell grids or homogeneous areas/polygons (with environmental coherences as soil type or land use) and attribute the mean/median value of the variable measured to the whole area where it was measured. (ii) G methods consist in using measurements distributed in the area studied and statistics tools to obtain a spatial representation of a studied variable. They were well described by Liu et al., (2011): based on regionalised variable theory (Matheron, 1963), geostatistics uses semivariograms (Yost et al., 1982) to quantify spatial autocorrelations and provide input parameters for spatial interpolation, i.e., kriging (Krige, 1951). (iii) Knowledge-based and regression models consist in fitting SOC (or any soil parameter) prediction models on reference datasets. The models can be simple or multiple regressions, on the whole studied areas or calculated and/or applied separately in/to the different parts of the studied area or dataset as in the Classification And Regression Tree (CART) models (Wiesmeier et al., 2011). Mapping SCS with such models requires mapped entries, that are usually soil characteristics such as texture, pH, humidity, bedrock type or parent material, drainage class; climate characteristics such as rainfall, temperature; elevation or slope; land use, land cover and land management.

3.1.2. Mapping issues

Mapping SOC is not straightforward and faces a number of issues from measuring or estimating it punctually to interpolating it to a given spatial-extent. Indeed, punctual estimates or measurements can be erroneous, and their precision depends on the methods used as reported by Goidts et al., (2009) for SCS parameters. Mapping methods have also their intrinsic issues leading to errors or uncertainties in their estimates compared to punctual reference (preferably direct measurements) and the choice of one method may depend on the parameter to interpolate, its spatial variability, the precision of its punctual estimation,

the spatial extent considered, the sampling design. Despite numerous approaches directed towards mapping SCS in the past 25 years, not all of them addressed their specific uncertainties, as did many authors in the last five years (de Brogniez et al., 2015; de Gruijter et al., 2016; Goidts et al., 2009; Kempen et al., 2010; Lacoste et al., 2014; Malone et al., 2009, 2011; Poggio and Gimona, 2014). Concerning SCS assessment, the most important stage is SOC content estimation, as SOC content variability explains 20 to 75% of the total SCS variability depending on spatial scale (Goidts et al., 2009). They showed that bulk density is generally the parameter contributing the least to the carbon stock variability, while Ellert et al., (2002) considered its variability not negligible when studying carbon sequestration in soil. Coarse or rock fragment is usually defined by a size, greater than 2mm, where smaller particles compose fine earth (Novák et al., 2011; Rytter, 2012). Rock fragments include stones which could be defined by a lower size limit of 20 mm based on a soil classification according to Atterberg (1905). Stoniness in croplands is often quite low because big stones have been removed, however, stoniness impact on carbon stock estimates may be consistent: ignoring stoniness leads to an overestimate in C and N stocks of up to 9% in fields (Rytter et al, 2012). In stony soils, where RF constituted between 19 and 24% of the soil mass, RF were the main sources of SCS variability at the field scale (Goidts et al., 2009). Concerning parameters of SCS fate, where other soil characteristics, as well as climate, can be punctually estimated before interpolation, it is not the case for crop production systems for which the smallest degree of variability is the crop plot. In order to choose interpolation methods for these first parameters, their spatial variability relatively to the precision of their estimations should preferably be assessed and relatively to each other and considering the spatial extent studied.

SCS also depend on the depth considered. The most studied soil depth are 0-20cm, 0-100cm, and mostly 0-30cm because considered as the reference depth layer for several reasons: (i) SCS seem to be concentrated in the surface layers (Jobbágy and Jackson, 2000), (ii) when studying agricultural soils the first 30 cm are usually considered as the working layer where most of the C and N transformations take place (transformation only in 30cm in most models), (iii) this layer is also the most sensitive to land use change (Batjes and Dijkshoorn, 1999; Batjes and Sombroek, 1997; Bouwman, 1990), (iv) studying surface layer is faster and easier, and SOC content estimates precision decreases with depth (Wang et al., 2004). Indeed, the feasibility, the repeatability and the cost of a sample analysis are also determinant for the estimations density over a certain area and therefore the interpolation accuracy and the method chosen for it. For instance, where SOC measurements are quite repeatable, BD and CF estimates are more time-consuming, and even more that of hydraulic parameters.

Parameters estimations face issues caused by variabilities in time or depth. For instance, SOC are variable along the soil profile mainly due to the fact that organic inputs come from the surface (Prechtel et al., 2009). Tillage operations lead to mix the ploughing layer and homogenise SOC, therefore measurements

made before and after tillage may highly vary depending on the depth considered. This issue can be overcome by measuring the whole ploughing layer or always at the same time in the agricultural calendar. Bulk density varies also a lot over the time (Alletto and Coquet, 2009): under conventional tillage, BD increased during the cropping cycle (Osunbitan et al., 2005), and decrease sharply because of the seedbed preparation. To overcome this issue, Ellert et al., (2001) proposed to work with a soil equivalent mass which is a reference soil mass per unit area in a layer. By using a reference mass, the depth studied will vary starting from the topsoil layer (Lee et al., 2009) and will offset the BD temporal variability (Gifford and Roderick, 2003). For the parameters determining SCS fate, such issues are less pregnant for the other soil characteristics. For cropping systems and climate parameters, estimates must be made temporarily for dynamics simulations and such issues depend on the temporal precision of their characterisation. For climate data, daily estimates are usually considered sufficient for crop model simulations.

3.1.3. Mapping methods used

Concerning the interpolation methods for SCS, it appeared that for very large areas, up to many millions of km², M method seems to be the only possible because of the scarcity of soil legacy data. Higher accuracy may be expected from either G or R methods or mixed approaches, according to the spatial density of soil data available. For smaller scales ranging from some km² to a hundred of km², geostatistics are more frequently used because they give a good representation of SOC variability involving acceptable accuracy. As far as we know, for studies using M methods, RMSE ranged from 1.11 to 36.6 kg C m^{-2}, which corresponded to 10 to 110% of the mean, but it was only ranging from 1.11 to 8.10 kg C m^{-2} if considering only the first 30 cm of soil. For the other studies the RMSE ranged from 1.08 to 3.93 kg C m^{-2} for those using geostatistical methods and from 6.4 to 26.3 kg C m^{-2} for those using Empirical models method.

Other methods such as remote sensing are used to directly estimate agro-pedological data in a spatially-explicit way. For instance soil spectroscopy was used to estimate SOC contents by measuring reflectance spectra from airborne hyperspectral images (Stevens et al., 2006, 2012, 2010; Vaudour et al., 2016) or from hyperspectral satellite images (Gomez et al., 2008) or from multispectral satellite images (Berthier et al., 2008; Vaudour et al., 2013). Such approach generally relies on regression equations (R methods) which require calibrations. The magnitude of error increases from aerial or satellite levels and from field to regional levels (Stevens et al., 2006, 2010; Gomez et al., 2008; Vaudour et al., 2013, 2016) and range from 0.09 g $C.kg^{-1}$ (Ben-Dor, 2002) to 10.5 g $C.kg^{-1}$ (Kooistra et al., 2003).

Data directly determined in a spatially-explicit way are also used concerning crop production systems. Some spatial data describing crop allotment may be available through land use identification system, such as for France, the Land Parcel Identification System (LPIS), created and updated by the French Service

and Payment Agency. The LPIS locates all declared agricultural plots in the framework of the common agricultural policy of the European Union, at a scale of 1:5000, by aerial photography and a geographic information system (GIS) (French Service and Payment Agency, 2013). The resulting data layer is named "Registre Parcellaire Graphique" (RPG) (Graphic Parcels Register). However such data are often spatially and semantically imprecise. Remote sensing is a widely use technique to spatially retrieve crops (Atzberger, 2013). Airborne or satellite data with very high resolution may be a straightforward solution for retrieving spatial data on crop systems and agricultural practices at each field over a whole territory (Vaudour et al., 2015). If these latter data can allow a more detailed spatial description of cropping systems, they are not collected systematically as LPIS, and may not yet allow definition of the whole crop succession.

The choice of how to map the different inputs for SOM dynamics simulations also depend on the model chosen and the accuracy of its simulation depending on the contexts considered. For assessing various impacts of EOM uses at a certain spatial scale, the range of the scenarios studied as well as the number of outputs considered can also influence the spatial description requirements of the region.

3.2. Spatially-explicit EOM impacts on C and N for regional concerns

One of the most researched EOM use interest at large scale is C storage in soil in order to offset global warming. For this purpose some authors studied potential EOM effects on SOM dynamics in a spatially-explicit way at national scales (Mondini et al., 2012; sSleutel et al., 2006; Yokozawa et al., 2010). At such scales, correctly studying these effects taking into account all spatial variabilities and existing matters variabilities is very challenging. This led authors to study EOM effect in a very simple way, exploring only one or few scenarios of EOM use, and only focused on C storage. Indeed, spatially-representing all models inputs is not straightforward and requires a lot of data, and models may not be able to simulate in wide range of contexts. Indeed, even mechanist crop models, have specific context of application, preferably contexts where they were calibrated.

Therond et al., (2011) demonstrated the feasibility of using crop models at regional scales by correctly simulating crop yields using low-data approaches for crop management information and model calibration combined with expert knowledge. Coucheney et al., (2015) underlined the importance of models robustness, and assessed STICS accuracy and robustness in a multi-sites study covering a wide range of agro-environmental conditions in France. This issue might be overstepped by studying small regions. At smaller scale, crop parameters can be considered less variable and one punctual calibration might be sufficient for simulations over the whole region. Similarly, punctual estimation of inputs such as climate, soil and cropping systems characteristics might be reasonably extrapolated for regional simulations.

Higher precision on spatial parameterisation that can be obtained at smaller scales would also allow considering more EOM amendments scenarios and more outputs simultaneously.
Taking into account simultaneous effects on C and N cycles in a spatially-explicit way were never made, despite the fact that N_2O emissions or economy of synthetic nitrogen caused by EOM use could also play an important role in GES emissions and global warming. Therefore, such N outputs need to studied spatially. EOM can also lead water pollutions because of nitrate and P, issues that can be spread over a certain area as can be their causes and can involve many stakeholders, requiring integrated studies. For such issues some authors worked at the landscape scales (Duretz et al., 2011; Drouet et al., 2012) however they only considered one criteria among the various EOM application outputs.

3.3. Regional EOM management

If sources of EOM are generally punctual (stud farm, composting plant, sewage sludge plant…), fields where EOM can be applied are often spread on larger areas which also implies transportation. The use of EOM will be driven by matter availability and demand of EOM which imply a sort of market that has a spatial extent, presumably of the size of a small region considering the issue of EOM transportation.
Availability of EOM in the large Paris Basin is quite low compared to potential demand. This is less true for the Plain of Versailles and its proximity to many cities and stud farm of this territory (Dhaouadi, 2014). EOM can be a limited resource for which impacts may vary depending on how they are managed on a region i.e. depending on the distribution pattern among the varying farms systems and the soil contexts of the region. Impacts on carbon storage, synthetic fertiliser saving or pollutions can be increased or decreased at the region scale if managed with this perspective. One study was conducted on EOM management at such scale. The GIROVAR project conducted a regional EOM management study. It economically and environmentally assessed EOM production projects for Reunion Island based on regional needs and raw material production. However, the environmental impacts were not quantitatively and spatially-explicitly evaluated. Water resource management, which may be a limited resource in specific contexts as EOM, was quite studied at a comparable scale (Bergez et al., 2012; Murgue et al., 2015). Water management implies many stakeholders interacting, among which are farmers that need to change their practices to tackle a global issue. For such purpose, Murgue et al., (2015) favoured participatory approach allowing more credibility, saliency and legitimacy of the outcomes. However, impacts on water savings of local changes in farmers' practices were roughly estimated, which may be possible for quantitative water management, but may not for EOM management dealing with C and N cycles. In that case, models may be necessary for outputs estimation taking into account the complexity of the regional soil and cropping systems. Bergez et al. (2012) who used combined models raised the need for robust and generic models when dealing with various spatial scales. They also demonstrated the need

for multi-criteria assessments in order to assure sustainability, flexibility and resilience of the modified cropping systems. Managing the EOM resource at a region scale taking into account various criteria (various C and N outputs of various possible amendment practices) may be even more complex and mathematical tool such as optimisation models can a be used for this purpose. In the agricultural field, Groot et al., (2012) used such tool to optimise allocation of production activities over the 96 ha of a farm following objectives such as profit maximisation while maintaining the SOM balance. Dogliotti et al., (2005) also used such model to explore land use allocation in 7 farms the area of which varied from 5.5 to 65.2 ha, in the purpose of improving farm family income, SOM and decreasing soil erosion. No such study was conducted focused on organic matter management and at large spatial scale. In such study, optimisation models are quite simple, however acquiring reliable data which will be used as entries for these models, is the most challenging part of the process.

4. Conclusion and objectives

In the present thesis, we studied EOM use in agriculture and focused on potential impacts on C and N cycles. As C storage in soil is of major concern because of all the positive effects that it can lead to on both climate and agriculture productivity, we first studied SOC stocks dynamics along with EOM amendments. About the N cycle, EOM applications have two different effects of great concerns: potential of synthetic fertiliser substitution and potential losses causing environment pollutions. We thus focused also on these two effects. We chose to study the small region of the “Plain of Versailles”, 221 km², because at that scale, spatially-explicit description of the region might be easier and simulations more accurate, and EOM management seems to be more relevant. The overall approach is described Figure 8. Many authors studied EOM through their chemical, biochemical composition and some also studied their C and N dynamics through mineralisation measured with lab incubations. However, to appreciate the impacts over the long-term when applied on field, lab characterisations are not sufficient. Mechanistic model can be a tool in order to integrate all factors influencing EOM fate in fields and to better assess EOM impacts on the environment and more specifically on C and N cycles. However, this requires EOM characterisation for model parameterisation and calibration thanks to field experiments which is expensive and labour-intensive compared to lab measurements. In Chapter 2, we focused on the feasibility of transposing SOM and EOM parameters required for such model, determined with laboratory measurements, to simulate long-term field experiment. The model, once calibrated, allowed us to study more precisely C and N dynamics and particularly impacts of EOM applications. We chose to use the model CERES-EGC coupled with NCSOIL because (i) it was first dedicated to crops and cultivars of the studied region and thus a priori better adapted than other models, (ii) it was already tested for the pedo-

climatic conditions of the region (Gabrielle et al., 2005, 2002), and (iii) NCSOIL was already tested for a wide range of EOM.

Existing EOM can be very variable, because of variable inputs and variable treatment processes, and so their C and N dynamics. These dynamics can also vary due to interactions with different soil characteristics on which they are applied. These variable dynamics must be determined in order to assess the potential impacts of EOM amendments depending on their type and on where they are applied. This is on what we focused in chapter 3 thanks to laboratory measurements of C and N mineralisation along with biochemical characterisation of a large panel of EOM, mixed with various soil types for incubations. Efforts to relate C and N dynamics of EOM through model parameters and lab measurements and in a generic way were made in order to allow simpler future model parameterisation.

At a region scale factors determining C and N cycles, even if focused on farm fields, such as soil types, initial SCS, cropping systems and interactions, are very variable. In order to better assess potential impacts of EOM use at this scale, simulations of EOM use scenarios integrated in the actual context of crop production of the specific region “Plain of Versailles” were performed. This allowed to study factors of influence, their relative importance and determined quantitatively impacts of EOM use on C and N cycles more precisely. This is developed in Chapter 4. In order to make this study useable for agriculture in this region and in the French context, potential pollutions on N, P or trace elements were taken into account, based on French regulatory system, to build the scenarios of EOM use.

Quantities of EOM produced are finite and at a regional scale competition may exist between different actors of EOM use. As part of the EOM effects we look for is for global concern, effects shall be appreciated, quantified globally, and therefore EOM use shall be managed at the regional scale. At this scale, factors influencing EOM effects are numerous, number of farms potentially using EOM is high, panel of EOM existing is large, managing EOM use becomes thus very complex. Therefore we used mathematical tools to manage this complexity and optimise the EOM use at the territory scale of the “Plain of Versailles” as described in chapter 5. The optimisation was made following different purposes, from maximising carbon storage to maximising synthetic nitrogen savings and eventually minimising N pollutions.

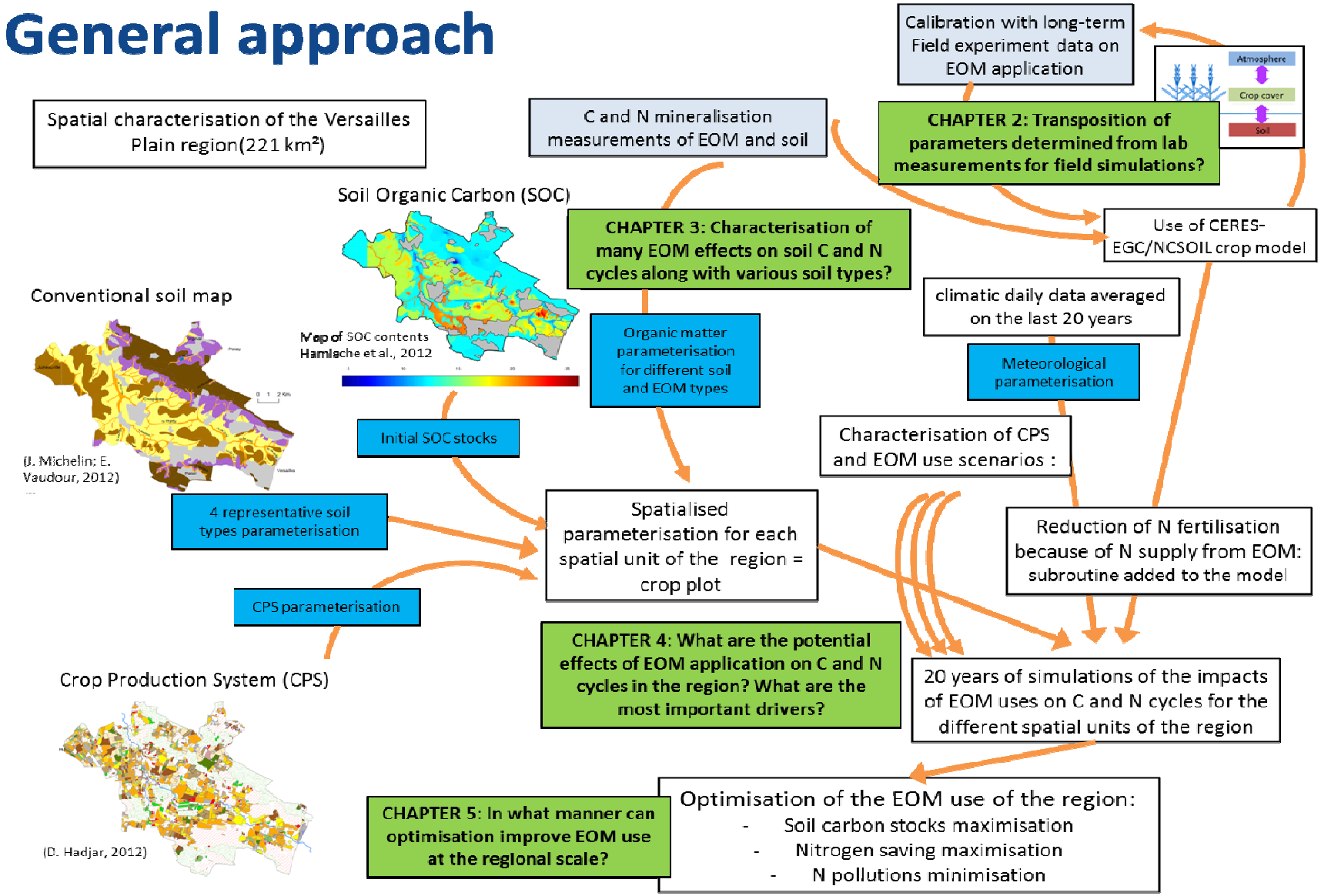


Figure 8. General approach

5. Supplementary data

Table 1S. Periods of ban for EOM application

Tableau 1.2 : Périodes d'interdiction d'épandage des fertilisants en fonction de leur type

Occupation du sol	Type de Fertilisants		
	Type I (ex fumiers)	Type II (ex Lisier)	Type III (engrais)
	C/N>8	C/N≤8	
Sols non cultivés	Toute l'année	Toute l'année	Toute l'année
Grandes cultures implantées à l'automne	Du 15/11 au 15/01	du 01/10 au 31/01	du 01/09 au 31/01
Grandes cultures implantées au printemps	Ia : 01/07 au 31/08 Du 15/11 au 15/01 Autre I : 01/07 au 15/01	du 01/07 au 31/01	du 01/07 au 15/02
Grandes cultures implantées au printemps avec CIPAN	Ia** Possible jusqu'à20j avant destruction CIPAN (≤70kg efficace) Autre* : de 15 j avant semis à 20 j avant destruction	de 15 j avant semis CIPAN à 20 j avant destruction	
Prairies implantées depuis plus de 6 mois	Du 15/12 au 15/01	du 15/11 au 15/01	du 01/10 au 31/01
Vigne	du 15/12 au 15/01	du 01/07 au 15/01	du 01/07 au 15/01
Autres cultures	du 15/12 au 15/01	du 15/12 au 15/01	du 15/12 au 15/01

*Ia : fumiers compacts pailleux, compost d'effluents d'élevage, certaines boues de papeterie
** Autre : autres type I

6. References

Abiven, S., Recous, S., Reyes, V., Oliver, R., 2005. Mineralisation of C and N from root, stem and leaf residues in soil and role of their biochemical quality. Biol. Fertil. Soils 42, 119–128.

Addiscott, T.M., Wagenet, R.J., 1985. Concepts of solute leaching in soils: a review of modelling approaches. J. Soil Sci. 36, 411–424.

ADEME, 2009. Les déchets en chiffres en France : Données et Références.

ADEME, 2012. Rapport Déchets.

AFNOR, 2009a. Norme XP U 44-163. Amendements organiques et supports de culture - Caractérisation de la matière organique par la minéralisation potentielle du carbone et de l'azote. AFNOR, Paris.

AFNOR, 2009b. Norme XP U 44-162. Amendements organiques et supports de culture - Caractérisation de la matière organique par fractionnement bio-chimique et estimation de sa stabilité biologique. AFNOR, Paris.

Agreste, 2012. Exploitations de grandes cultures en France métropolitaine. Agreste Primeur.

Alletto, L., Coquet, Y., 2009. Temporal and spatial variability of soil bulk density and near-saturated hydraulic conductivity under two contrasted tillage management systems. Geoderma 152, 85–94.

Alvarez, R., 2005. A review of nitrogen fertilizer and conservation tillage effects on soil organic carbon storage. Soil Use Manag. 21, 38–52.

Ambus, P., Kure, L.K., Jensen, E.S., 2002. Gross N transformation rates after application of household compost or domestic sewage sludge. Agron. Sustain. Dev. 22, 8.

Andriulo, A., Mary, A.B., Guérif, J., 1999. Modelling soil carbon dynamics with various cropping sequences on the rolling pampas. Agronomie 19, 365–377.

Angers, D.A., Arrouays, D., Saby, N.P.A., Walter, C., 2011. Estimating and mapping the carbon saturation deficit of French agricultural topsoils. Soil Use Manag. 27, 448–452.

Angers, D.A., Eriksen-Hamel, N.S., 2008. Full-Inversion Tillage and Organic Carbon Distribution in Soil Profiles: A Meta-Analysis. Soil Sci. Soc. Am. J. 72, 1370–1374.

Antil, R.S., Bar-Tal, A., Fine, P., Hadas, A., 2011. Predicting Nitrogen and Carbon Mineralization of Composted Manure and Sewage Sludge in Soil. Compost Sci. Util. 19, 33–43.

Ardö, J., Olsson, L., 2003. Assessment of soil organic carbon in semi-arid Sudan using GIS and the CENTURY model. J. Arid Environ. 54, 633–651.

Arrouays, D., Balesdent, J., Germon, J.C., Jayet, P.A., Soussana, J.F., Stengel, P., Bureau, D., 2002. Contribution à la lutte contre l'effet de serre: (stocker du carbone dans les sols agricoles de France ?). Expertise scientifique collective, Paris: INRA.

Arrouays, D., Deslais, W., Badeau, V., 2001. The carbon content of topsoil and its geographical distribution in France. Soil Use Manag. 17, 7–11.

Arrouays, D., Pelissier, P., 1994. Modeling Carbon Storage Profiles in Temperate Forest Humic Loamy Soils. Soil Sci. 157, 185–192.

Atterberg, A., 1905. Die rationale klassifikation der sande und kiese. Chem.-Ztg. 29, 195–198.

Atzberger, C., 2013. Advances in Remote Sensing of Agriculture: Context Description, Existing Operational Monitoring Systems and Major Information Needs. Remote Sens. 5, 949–981.

Augris, N., Balesdent, J., Mariotti, A., Derenne, S., Largeau, C., 1998. Structure and origin of insoluble and non-hydrolyzable, aliphatic organic matter in a forest soil. Org. Geochem. 28, 119–124.

Balesdent, J., Recous, S., 1997. Les temps de résidence du carbone et le potentiel de stockage de carbone dans quelques sols cultivés français. Can. J. Soil Sci. 77, 187–193.

Basso, B., Ritchie, J.T., 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.

Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. Eur. J. Soil Sci. 47, 151–163.

Batjes, N.H., 2008. Mapping soil carbon stocks of Central Africa using SOTER. Geoderma 146, 58–65.

Batjes, N.H., Dijkshoorn, J.A., 1999. Carbon and nitrogen stocks in the soils of the Amazon Region. Geoderma 89, 273–286.

Batjes, N. h., Sombroek, W. g., 1997. Possibilities for carbon sequestration in tropical and subtropical soils. Glob. Change Biol. 3, 161–173.

Bellamy, P.H., Loveland, P.J., Bradley, R.I., Lark, R.M., Kirk, G.J.D., 2005. Carbon losses from all soils across England and Wales 1978–2003. Nature 437, 245–248.

Ben-Dor, E., 2002. Quantitative remote sensing of soil properties. Adv. Agron. 75, 173–243.

Beraud, J., Fine, P., Yermiyahu, U., Keinan, M., Rosenberg, R., Hadas, A., Bar-Tal, A., 2005. Modeling Carbon and Nitrogen Transformations for Adjustment of Compost Application with Nitrogen Uptake by Wheat. J. Environ. Qual. 34, 664.

Bergez, J.E., Leenhardt, D., Colomb, B., Dury, J., Carpani, M., Casagrande, M., Charron, M.H., Guillaume, S., Therond, O., Willaume, M., 2012. Computer-model tools for a better agricultural water management: Tackling managers' issues at different scales – A contribution from systemic agronomists. Comput. Electron. Agric., Multi-scale water and land-use modelling for better decision making in agro-eco systems 86, 89–99.

Bergstrom, D.W., Monreal, C.M., St Jacques, E., 2001. Spatial dependence of soil organic carbon mass and its relationship to soil series and topography. Can. Jounal Soil Sci. 81, 53–62.

Bernoux, 2002. Brazil's Soil Carbon Stocks. Soil Sci. Soc. Am. J. 66, 888–896.

Berthier, L., Vaudour, Pitres, J.-C., 2008. Prédiction spatiale des teneurs en carbone organique des sols par spectroscopie de terrain visible-proche infrarouge et imagerie satellitale SPOT Exemple au niveau d'un périmètre d'alimentation en eau potable en Beauce. Etude Gest. Sols 213–224.

Biomasse Normandie, 2002. Evaluation des quantités actuelles et futures des déchets épandus sur les sols agricoles et provenance de certaines activités. Lot 3 : Effluents d'élevage.

Bliss, N.B., Maursetter, J., 2010. Soil Organic Carbon Stocks in Alaska Estimated with Spatial and Pedon Data. Soil Sci. Soc. Am. J. 74, 565–579.

Bolin, B., Sukumar, R., 2000. Global perspective. Land Use Land-Use Change For., Cambridge University Press 23–51.

Bouthier, A., Trochard, R., Parnaudeau, V., Nicolardot, B., 2009. Cinétique de minéralisation nette de l'azote organique des produits résiduaires organiques à court terme in situ et en conditions contrôlées. 9èmes Rencontres Fertil. Raison. Anal. Terre Comifer-Gemas 6.

Bouwman, A.F., 1990. Soils and the greenhouse effect: The present status and future trends concerning the effect of soils and their cover on the fluxes of greenhouse gases, the surface energy balance and the water balance. John Wiley Sons Inc West Sussex Engl. 575.

Brisson, N., Gary, C., Justes, E., Roche, R., Mary, B., Ripoche, D., Zimmer, D., Sierra, J., Bertuzzi, P., Burger, P., Bussière, F., Cabidoche, Y.M., Cellier, P., Debaeke, P., Gaudillère, J.P., Henault, C., Maraux, F., Seguin, B., Sinoquet, H., 2003. An overview of the crop model stics. Eur. J. Agron., Modelling Cropping Systems: Science, Software and Applications 18, 309–332.

Bruun, S., Hansen, T.L., Christensen, T.H., Magid, J., Jensen, L.S., 2006. Application of processed organic municipal solid waste on agricultural land – a scenario analysis. Environ. Model. Assess. 11, 251–265.

Burger, M., Venterea, R.T., 2008. Nitrogen Immobilization and Mineralization Kinetics of Cattle, Hog, and Turkey Manure Applied to Soil. Soil Sci. Soc. Am. J. 72, 1570.

Cerri, C.E.P., Easter, M., Paustian, K., Killian, K., Coleman, K., Bernoux, M., Falloon, P., Powlson, D.S., Batjes, N.H., Milne, E., Cerri, C.C., 2007. Predicted soil organic carbon stocks and changes in the Brazilian Amazon between 2000 and 2030. Agric. Ecosyst. Environ. 122, 58–72.

Chadwick, D.R., John, F., Pain, B.F., Chambers, B.J., Williams, J., 2000. Plant uptake of nitrogen from the organic nitrogen fraction of animal manures: a laboratory experiment. J. Agric. Sci. 134, 159–168.

Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B., Misselbrook, T., 2011. Manure management: Implications for greenhouse gas emissions. Anim. Feed Sci. Technol., Special Issue: Greenhouse Gases in Animal Agriculture - Finding a Balance between Food and Emissions 166–167, 514–531.

Chalhoub, M., Garnier, P., Coquet, Y., Mary, B., Lafolie, F., Houot, S., 2013. Increased nitrogen availability in soil after repeated compost applications: Use of the PASTIS model to separate short and long-term effects. Soil Biol. Biochem. 65, 144–157.

Chaplot, V., Bouahom, B., Valentin, C., 2010. Soil organic carbon stocks in Laos: spatial variations and controlling factors. Glob. Change Biol. 16, 1380–1393.

Chaussod, R., Nicolardot, B., Catroux, G., Chretien, J., 1986. Relations entre les caractéristiques physicochimiques et microbiologiques de quelques sols cultivés. Sci. Sol 213–226.

Cheng, X.F., Shi, X.Z., Yu, D.S., Pan, X.Z., Wang, H.J., Sun, W.X., 2004. Using GIS spatial distribution to predict soil organic carbon in subtropical China. Pedosphere 14, 425–431.

Chenu, C., Plante, A.F., 2006. Clay-sized organo-mineral complexes in a cultivation chronosequence: revisiting the concept of the "primary organo-mineral complex." Eur. J. Soil Sci. 57, 596–607.

Chertov, O.G., 1990. Specom - a single tree model of pine stand/raw humus soil ecosystem. Ecol. Model. 50, 107–132.

Chertov, O.G., Komarov, A.S., 1997. SOMM: A model of soil organic matter dynamics. Ecol. Model. 94, 177–189.

Chuai, X.-W., Huang, X.-J., Wang, W.-J., Zhang, M., Lai, L., Liao, Q.-L., 2012. Spatial Variability of Soil Organic Carbon and Related Factors in Jiangsu Province, China. Pedosphere 22, 404–414.

Ciais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., Chhabra, A., DeFries, R., Galloway, J., Heimann, M., Jones, C., Le Quéré, C., Myneni, R.B., Piao, S., Thornton, P., 2013. Carbon and Other Biogeochemical Cycles. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Ciais, P., Wattenbach, M., Vuichard, N., Smith, P., Piao, S.L., Don, A., Luyssaert, S., Janssens, I.A., Bondeau, A., Dechow, R., Leip, A., Smith, P., Beer, C., Van Der Werf, G.R., Gervois, S., Van Oost, K., Tomelleri, E., Freibauer, A., Schulze, E.D., Carboeurope Synthesis Team, 2010. The European carbon balance. Part 2: croplands. Glob. Change Biol. 16, 1409–1428.

Cohan, J.P., Charpiot, A., Morvan, T., Eveillard, P., Trochard, R., Champolivier, L., De Chezelles, E., Heurtaux, M., Lorinquer, E., Espagnol, S., Génermont, S., Loubet, B., 2013. Quantification des emissions d'ammoniac suite à l'épandage de produits résiduaires organiques et d'engrais minéraux au champ - 1ere synthèse des résultats acquis dans le cadre du projet CASDAR VOLAT'NH3. Presented at the 11emes rencontres de la fertilisation raisonnée et de l'analyse/COMIFER-GEMAS, Poitiers.

Coleman, K., Jenkinson, D.S., 1996. RothC-26.3 - A Model for the turnover of carbon in soil. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of Soil Organic Matter Models, NATO ASI Series. Springer Berlin Heidelberg, pp. 237–246.

Corbeels, M., Hofman, G., Cleemput, O.V., 1999. Simulation of net N immobilisation and mineralisation in substrate-amended soils by the NCSOIL computer model. Biol. Fertil. Soils 28, 422–430.

Coucheney, E., Buis, S., Launay, M., Constantin, J., Mary, B., García de Cortázar-Atauri, I., Ripoche, D., Beaudoin, N., Ruget, F., Andrianarisoa, K.S., Le Bas, C., Justes, E., Léonard, J., 2015. Accuracy, robustness and behavior of the STICS soil–crop model for plant, water and nitrogen outputs: Evaluation over a wide range of agro-environmental conditions in France. Environ. Model. Softw. 64, 177–190.

Daudén, A., Quílez, D., 2004. Pig slurry versus mineral fertilization on corn yield and nitrate leaching in a Mediterranean irrigated environment. Eur. J. Agron. 21, 7–19.

Davidson, E.A., 1995. Spatial covariation of soil organic carbon, clay content, and drainage class at a regional scale. Landsc. Ecol. 10, 349–362.

Davidson, E.A., Lefebvre, P.A., 1993. Estimating regional carbon stocks and spatially covarying edaphic factors using soil maps at three scales. Biogeochemistry 22, 107–131.

de Brogniez, D., Ballabio, C., Stevens, A., Jones, R.J.A., Montanarella, L., van Wesemael, B., 2015. A map of the topsoil organic carbon content of Europe generated by a generalized additive model. Eur. J. Soil Sci. 66, 121–134.

de Gruijter, J.J., McBratney, A.B., Minasny, B., Wheeler, I., Malone, B.P., Stockmann, U., 2016. Farm-scale soil carbon auditing. Geoderma 265, 120–130.

Delphin, J.-E., Huck, C., Tiquet, J.-L., 1986. Evaluation du pouvoir minéralisateur de sols agricoles en fonction de leurs caractéristiques physico-chimiques. Agronomie 6, 453–458.

Dersch, G., Böhm, K., 2001. Effects of agronomic practices on the soil carbon storage potential in arable farming in Austria. Nutr. Cycl. Agroecosystems 60, 49–55.

Dhaouadi, A.K., 2014. Insertion des Produits Résiduaires Organiques dans les systèmes de culture : Cas des systèmes céréaliers de la Plaine de Versailles et du Plateau des Alluets. Paris, AgroParisTech.

Diacono, M., Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility. A review. Agron. Sustain. Dev. 30, 401–422.

Dogliotti, S., van Ittersum, M.K., Rossing, W.A.H., 2005. A method for exploring sustainable development options at farm scale: a case study for vegetable farms in South Uruguay. Agric. Syst. 86, 29–51.

Don, A., Schumacher, J., Scherer-Lorenzen, M., Scholten, T., Schulze, E.-D., 2007. Spatial and vertical variation of soil carbon at two grassland sites — Implications for measuring soil carbon stocks. Geoderma 141, 272–282.

Dou, F., Yu, X., Ping, C.-L., Michaelson, G., Guo, L., Jorgenson, T., 2010. Spatial variation of tundra soil organic carbon along the coastline of northern Alaska. Geoderma 154, 328–335.

Drouet, J.-L., Duretz, S., Durand, P., Cellier, P., 2012. Modelling the contribution of short-range atmospheric and hydrological transfers to nitrogen fluxes, budgets and indirect emissions in rural landscapes. Biogeosciences 9, 1647–1660.

Duretz, S., Drouet, J.L., Durand, P., Hutchings, N.J., Theobald, M.R., Salmon-Monviola, J., Dragosits, U., Maury, O., Sutton, M.A., Cellier, P., 2011. NitroScape: A model to integrate nitrogen transfers and transformations in rural landscapes. Environ. Pollut., Assessment of Nitrogen Fluxes to Air and Water from Site Scale to Continental Scale 159, 3162–3170.

Ellert, B.H., Janzen, H.H., Entz, T., 2002. Assessment of a method to measure temporal change in soil carbon storage. Soil Sci. Soc. Am. J. 66, 1687–1695.

Ellert, B.H., Janzen, H.H., McConkey, B.G., 2001. Measuring and comparing soil carbon storage. In: Assessment Methods for Soil Carbon. CRC Press.

Enckell, C., 2013. Recyclage des déchets dangereux: le juge européen et Reach au secours du statut de produit. Environ. Tech. 52–55.

European Commission, 2006a. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions - Thematic Strategy for Soil Protection [SEC(2006)620][SEC(2006)1165].

European Commission, 2006b. Proposal for a Directive of the European Parliament and of the Council establishing a framework for the protection of soil and amending Directive 2004/35/EC.

Fischer, G., Velthuizen, H.V., Nachtergaele, F.O., Jernelöv, A., 2000. Global Agro-Ecological Zones Assessment: Methodology and Results. International Institute for Applied Systems Analysis, Food and Agriculturale Organization of the United Nations, Laxenburg, Austria.

FranceAgriMer, 2012. L'observatoire national des ressources en biomasse. Evaluation des ressources disponibles en France. Les études de FranceAgriMer, Montreuil-sous- Bois cedex: FranceAgriMer.

Franko, U., Oelschlägel, B., Schenk, S., 1995. Simulation of temperature-, water- and nitrogen dynamics using the model CANDY. Ecol. Model., Modelling of Geo-Biosphere Processes 81, 213–222.

Frogbrook, Z.L., Bell, J., Bradley, R.I., Evans, C., Lark, R.M., Reynolds, B., Smith, P., Towers, W., 2009. Quantifying terrestrial carbon stocks: examining the spatial variation in two upland areas in the UK and a comparison to mapped estimates of soil carbon. Soil Use Manag. 25, 320–332.

Gabrielle, B., Da-Silveira, J., Houot, S., Francou, C., 2004. Simulating Urban Waste Compost Effects on Carbon and Nitrogen Dynamics Using a Biochemical Index. J. Environ. Qual. 33, 2333.

Gabrielle, B., Da-Silveira, J., Houot, S., Michelin, J., 2005. Field-scale modelling of carbon and nitrogen dynamics in soils amended with urban waste composts. Agric. Ecosyst. Environ. 110, 289–299.

Gabrielle, B., Laville, P., Duval, O., Nicoullaud, B., Germon, J.C., Henault, C., 2006. Process-based modeling of nitrous oxide emissions from wheat-cropped soils at the subregional scale. Glob. Biogeochem. Cycles 20, 1–13.

Gabrielle, B., Mary, B., Roche, R., Smith, P., Gosse, G., 2002. Simulation of carbon and nitrogen dynamics in arable soils: a comparison of approaches. Eur. J. Agron., Process Simulation and Application of Cropping System Models 18, 107–120.

Gabrielle, B., Menasseri, S., Houot, S., 1995. Analysis and Field Evaluation of the Ceres Models Water Balance Component. Soil Sci. Soc. Am. J. 59, 1403–1412.

Garnier, P., Néel, C., Aita, C., Recous, S., Lafolie, F., Mary, B., 2003. Modelling carbon and nitrogen dynamics in a bare soil with and without straw incorporation. Eur. J. Soil Sci. 54, 555–568.

Gerke, H.H., Arning, M., Stöppler-Zimmer, H., 1999. Modeling long-term compost application effects on nitrate leaching. Plant Soil 213, 75–92.

Giacomini, S.J., Recous, S., Mary, B., Aita, C., 2007. Simulating the effects of N availability, straw particle size and location in soil on C and N mineralization. Plant Soil 301, 289–301.

Gifford, R.M., Roderick, M.L., 2003. Soil carbon stocks and bulk density: spatial or cumulative mass coordinates as a basis of expression? Glob. Change Biol. 9, 1507–1514.

Gnanavelrajah, N., Shrestha, R.P., Schmidt-Vogt, D., Samarakoon, L., 2008. Carbon stock assessment and soil carbon management in agricultural land-uses in Thailand. Land Degrad. Dev. 19, 242–256.

Goidts, E., Van Wesemael, B., Crucifix, M., 2009. Magnitude and sources of uncertainties in soil organic carbon (SOC) stock assessments at various scales. Eur. J. Soil Sci. 60, 723–739.

Gomez, C., Viscarra Rossel, R.A., McBratney, A.B., 2008. Soil organic carbon prediction by hyperspectral remote sensing and field vis-NIR spectroscopy: An Australian case study. Geoderma 146, 403–411.

Grimm, R., Behrens, T., Märker, M., Elsenbeer, H., 2008. Soil organic carbon concentrations and stocks on Barro Colorado Island — Digital soil mapping using Random Forests analysis. Geoderma 146, 102–113.

Groot, J.C.J., Oomen, G.J.M., Rossing, W.A.H., 2012. Multi-objective optimization and design of farming systems. Agric. Syst. 110, 63–77.

Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Glob. Change Biol. 8, 345–360.

Guo, Y., Amundson, R., Gong, P., Yu, Q., 2006. Quantity and Spatial Variability of Soil Carbon in the Conterminous United States. Soil Sci. Soc. Am. J. 70, 590.

Gutser, R., Ebertseder, T., Weber, A., Schraml, M., Schmidhalter, U., 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.

Hadas, A., Molina, J. a. E., 1993. Simulation of nitrogen assimilation by heterotrophic soil microbial biomass. Physiol. Plant. 89, 664–668.

Hansen, S., Jensen, H.E., Nielsen, N.E., Svendsen, H., 1991. Simulation of nitrogen dynamics and biomass production in winter wheat using the Danish simulation model DAISY. Fertil. Res. 27, 245–259.

Henin, S., Dupuis, M., 1945. Essai de bilan de la matière organique du sol. Ann. Agron. 17–29.

Holmes, K.W., Wherrett, A., Keating, A., Murphy, D.V., 2012. Meeting bulk density sampling requirements efficiently to estimate soil carbon stocks. Soil Res. 49, 680–695.

Homann, P.S., Sollins, P., Fiorella, M., Thorson, T., Kern, J.S., 1998. Regional soil organic carbon storage estimates for western Oregon by multiple approaches. Soil Sci. Soc. Am. J. 62, 789–796.

Houot, S., Pons, M.-N., Pradel, M., Caillaud, M.-A., Savini, I., Tibi, A., 2014. Valorisation des matières fertilisantes d'origine résiduaire sur les sols à usage agricole ou forestier. Impacts agronomiques, environnementaux, socio-économiques. Expertise scientifique collective. INRA-CNRS-Irstea. Synthèse 113pp (http://institut.inra.fr/Missions/Eclairer-les-decisions/Expertises/Toutes-les-actualites/Expertise-Mafor-effluents-boues-et-dechets-organiques#).

Howard, P. j. a., Loveland, P. j., Bradley, R. i., Dry, F. t., Howard, D. m., Howard, D. c., 1995. The carbon content of soil and its geographical distribution in Great Britain. Soil Use Manag. 11, 9–15.

Huang, B., Sun, W., Zhao, Y., Zhu, J., Yang, R., Zou, Z., Ding, F., Su, J., 2007. Temporal and spatial variability of soil organic matter and total nitrogen in an agricultural ecosystem as affected by farming practices. Geoderma 139, 336–345.

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA,.

Jemison, J.M., Fox, R.H., 1992. Estimation of zero-tension pan lysimeter collection efficiency. Soil Sci., Williams & Wilkins 1992 154.

Jemison, J.M., Fox, R.H., 1994. Nitrate Leaching from Nitrogen-Fertilized and Manured Corn Measured with Zero-Tension Pan Lysimeters. J. Environ. Qual. 23, 337–343.

Jenkinson, D.S., Andrew, S.P.S., Lynch, J.M., Goss, M.J., Tinker, P.B., 1990. The Turnover of Organic Carbon and Nitrogen in Soil [and Discussion]. Philos. Trans. R. Soc. Lond. B Biol. Sci. 329, 361–368.

Jenkinson, D.S., Jenkinson, P.B.S., Hart, J.H., Rayner, J.H., Parry, L.C., 1987. Modelling the turnover of organic matter in long term experiments at Rothamsted. INTECOL Bull. 15, 1–8.

Jobbágy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecol. Appl. 10, 423–436.

Jones, C.A., Kiniry, J.R., Dyke, P.T., 1986. CERES-Maize: a simulation model of maize growth and development. Tex. AM Univ. Press Coll. Stn. 194.

Kempen, B., Heuvelink, G.B.M., Brus, D.J., Stoorvogel, J.J., 2010. Pedometric mapping of soil organic matter using a soil map with quantified uncertainty. Eur. J. Soil Sci. 61, 333–347.

Kern, J.S., 1994. Spatial Patterns of Soil Organic Carbon in the Contiguous United States. Soil Sci. Soc. Am. J. 58, 439–455.

Kirkby, C.A., Kirkegaard, J.A., Richardson, A.E., Wade, L.J., Blanchard, C., Batten, G., 2011. Stable soil organic matter: A comparison of C:N:P:S ratios in Australian and other world soils. Geoderma 163, 197–208.

Kooistra, L., Wanders, J., Epema, G.F., Leuven, R.S.E.W., Wehrens, R., Buydens, L.M.C., 2003. The potential of field spectroscopy for the assessment of sediment properties in river floodplains. Anal. Chim. Acta 484, 189–200.

Krige, D.G., 1951. A statistical approach to some basic mine valuation problems on the Witwatersrand. • J. Chem. Metall. Min. Soc. South Afr. 52, 119–139.

Kumar, S., Lal, R., Liu, D., 2012. A geographically weighted regression kriging approach for mapping soil organic carbon stock. Geoderma 189–190, 627–634.

Kundu, S., Bhattacharyya, R., Prakash, V., Ghosh, B.N., Gupta, H.S., 2007. Carbon sequestration and relationship between carbon addition and storage under rainfed soybean–wheat rotation in a sandy loam soil of the Indian Himalayas. Soil Tillage Res. 92, 87–95.

Lacoste, M., Minasny, B., McBratney, A., Michot, D., Viaud, V., Walter, C., 2014. High resolution 3D mapping of soil organic carbon in a heterogeneous agricultural landscape. Geoderma 213, 296–311.

Lal, R., 2004. Soil carbon sequestration to mitigate climate change. Geoderma 123, 1–22.

Lal, R., Follett, F., Stewart, B.A., Kimble, J.M., 2007. Soil carbon sequestration to mitigate climate change and advance food security. Soil Sci. 172, 943–956.

Lal, R., Griffin, M., Apt, J., Lave, L., Morgan, M., 2004. Managing soil carbon. Science 304 : 393.

Lashermes, G., 2005. Etude et modélisation de la décomposition dans le sol des produits résiduaires organiques (Rapport de stage M2).

Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuries, L., Chaussod, R., Guillotin, M.L., Lineres, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S., 2009. Indicator of

potential residual carbon in soils after exogenous organic matter application. Eur. J. Soil Sci. 60, 297–310.

Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuriès, L., Chaussod, R., Guillotin, M.L., Linères, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S., 2010. Typology of exogenous organic matters based on chemical and biochemical composition to predict potential nitrogen mineralization. Bioresour. Technol. 101, 157–164.

Laville, P., Michelin, J., Houot, S., Gueudet, J.-C., Rampon, J.N., Labat, C., Vaudour, E., 2014. Soil N2O Emissions from Recovered Organic Waste Application in Versailles Plain (France): A Laboratory Approach. Waste Biomass Valorization 5, 515–527.

Lee, J., Hopmans, J.W., Rolston, D.E., Baer, S.G., Six, J., 2009. Determining soil carbon stock changes: Simple bulk density corrections fail. Agric. Ecosyst. Environ. 134, 251–256.

Leirós, M.C., Trasar-Cepeda, C., Seoane, S., Gil-Sotres, F., 1999. Dependence of mineralization of soil organic matter on temperature and moisture. Soil Biol. Biochem. 31, 327–335.

Le Quéré, C., Moriarty, R., Andrew, R.M., Peters, G.P., Ciais, P., Friedlingstein, P., Jones, S.D., Sitch, S., Tans, P., Arneth, A., Boden, T.A., Bopp, L., Bozec, Y., Canadell, J.G., Chevallier, F., Cosca, C.E., Harris, I., Hoppema, M., Houghton, R.A., House, J.I., Jain, A., Johannessen, T., Kato, E., Keeling, R.F., Kitidis, V., Klein Goldewijk, K., Koven, C., Landa, C.S., Landschützer, P., Lenton, A., Lima, I.D., Marland, G., Mathis, J.T., Metzl, N., Nojiri, Y., Olsen, A., Ono, T., Peters, W., Pfeil, B., Poulter, B., Raupach, M.R., Regnier, P., Rödenbeck, C., Saito, S., Salisbury, J.E., Schuster, U., Schwinger, J., Séférian, R., Segschneider, J., Steinhoff, T., Stocker, B.D., Sutton, A.J., Takahashi, T., Tilbrook, B., Werf, V.D., R, G., Viovy, N., Wang, Y.-P., Wanninkhof, R., Wiltshire, A., Zeng, N., 2014. Global carbon budget 2014. Earth Syst. Sci. Data Discuss. 7, 521–610.

Lessard, R., Rochette, P., Gregorich, E.G., Pattey, E., Desjardins, R.L., 1996. Nitrous Oxide Fluxes from Manure-Amended Soil under Maize. J. Environ. Qual. 25, 1371–1377.

Lettens, S., Van Orshoven, J., van Wesemael, B., De Vos, B., Muys, B., 2005a. Stocks and fluxes of soil organic carbon for landscape units in Belgium derived from heterogeneous data sets for 1990 and 2000. Geoderma 127, 11–23.

Lettens, S., Van Orshoven, J., van Wesemael, B., Muys, B., 2004. Soil organic and inorganic carbon contents of landscape units in Belgium derived using data from 1950 to 1970. Soil Use Manag. 20, 40–47.

Lettens, S., Van Orshoven, J., Van Wesemael, B., Muys, B., Perrin, D., 2005b. Soil organic carbon changes in landscape units of Belgium between 1960 and 2000 with reference to 1990. Glob. Change Biol. 11, 2128–2140.

Le villio, M., Arrouays, D., Deslais, W., Daroussin, J., Le bissonnais, Y., Clergeot, D., 2001. Estimation des quantités de matière organique exogène nécessaires pour restaurer et entretenir les sols limonuex français à un niveau organique donné. Etude Gest. Sols 8, 47–63.

Li, C., Frolking, S., Crocker, G.J., Grace, P.R., Klír, J., Körchens, M., Poulton, P.R., 1997. Simulating trends in soil organic carbon in long-term experiments using the DNDC model. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 45–60.

Li, C., Frolking, S., Frolking, T.A., 1992. A model of nitrous oxide evolution from soil driven by rainfall events. J. Geophys. Res. Atmospheres 97, 9759–9783.

Linères, M., Djakovitch, J., 1993. Caractérisation de la stabilité biologique des apports organiques par l'analyse biochimique. Decroux J Ignazi JC Eds Quatr. Journ. L'analyse Terre Cinquième Forum Fertil. Raison. GEMAS-COMIFER Blois 159–168.

Liu, D., Wang, Z., Zhang, B., Song, K., Li, X., Li, J., Li, F., Duan, H., 2006. Spatial distribution of soil organic carbon and analysis of related factors in croplands of the black soil region, Northeast China. Agric. Ecosyst. Environ. 113, 73–81.

Liu, S., Wei, Y., Post, W.M., Cook, R.B., Schaefer, K., Thornton, M.M., 2013. The Unified North American Soil Map and its implication on the soil organic carbon stock in North America. Biogeosciences 10, 2915–2930.

Liu, Z., Shao, M. 'an, Wang, Y., 2011. Effect of environmental factors on regional soil organic carbon stocks across the Loess Plateau region, China. Agric. Ecosyst. Environ. 142, 184–194.

Loveland, P., Webb, J., 2003. Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. Soil Tillage Res. 70, 1–18.

Luo, Z., Wang, E., Sun, O.J., 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agric. Ecosyst. Environ. 139, 224–231.

Malhi, Y., Baldocchi, D.D., Jarvis, P.G., 1999. The carbon balance of tropical, temperate and boreal forests. Plant Cell Environ. 22, 715–740.

Malone, B.P., McBratney, A.B., Minasny, B., 2011. Empirical estimates of uncertainty for mapping continuous depth functions of soil attributes. Geoderma 160, 614–626.

Malone, B.P., McBratney, A.B., Minasny, B., Laslett, G.M., 2009. Mapping continuous depth functions of soil carbon storage and available water capacity. Geoderma 154, 138–152.

Mamo, M., Rosen, C.J., Halbach, T.R., 1999. Nitrogen Availability and Leaching from Soil Amended with Municipal Solid Waste Compost. J. Environ. Qual. 28, 1074.

Marmo, L., Feix, I., Bourmeau, E., Amlinger, F., Bannick, C.G., De Neve, S., Favoino, E., Gendebien, A., Gibert, J., Givelet, M., Leifert, I., Morris, R., Rodriguez Cruz, A., Ruck, F., Siebert, S., Tittarelli, F., 2004. Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection, vol. 3, taskegroup 4 Exogenous Organic Matter.

Martin, M.P., Wattenbach, M., Smith, P., Meersmans, J., Jolivet, C., Boulonne, L., Arrouays, D., 2011. Spatial distribution of soil organic carbon stocks in France. Biogeosciences 8, 1053–1065.

Matheron, G., 1963. Principles of geostatistics. Econ. Geol. 58, 1246–1266.

Matsuura, S., Sasaki, H., Kohyama, K., 2012. Organic carbon stocks in grassland soils and their spatial distribution in Japan. Grassl. Sci. 58, 79–93.

McGrath, D., Zhang, C., 2003. Spatial distribution of soil organic carbon concentrations in grassland of Ireland. Appl. Geochem. 18, 1629–1639.

Meersmans, J., De Ridder, F., Canters, F., De Baets, S., Van Molle, M., 2008. A multiple regression approach to assess the spatial distribution of Soil Organic Carbon (SOC) at the regional scale (Flanders, Belgium). Geoderma 143, 1–13.

Meersmans, J., Martin, M.P., Lacarce, E., Baets, S.D., Jolivet, C., Boulonne, L., Lehmann, S., Saby, N.P.A., Bispo, A., Arrouays, D., 2012. A high resolution map of French soil organic carbon. Agron. Sustain. Dev. 32, 841–851.

Meersmans, J., van Wesemael, B., De Ridder, F., Van Molle, M., 2009. Modelling the three-dimensional spatial distribution of soil organic carbon (SOC) at the regional scale (Flanders, Belgium). Geoderma 152, 43–52.

Meersmans, J., Van WESEMAEL, B., Goidts, E., Van MOLLE, M., De BAETS, S., De RIDDER, F., 2011. Spatial analysis of soil organic carbon evolution in Belgian croplands and grasslands, 1960–2006. Glob. Change Biol. 17, 466–479.

Minasny, B., McBratney, A.B., Malone, B.P., Wheeler, I., 2013. Digital Mapping of Soil Carbon. In: Advances in Agronomy. Academic Press, pp. 1–47.

Mishra, U., Lal, R., Liu, D., Van Meirvenne, M., 2010. Predicting the Spatial Variation of the Soil Organic Carbon Pool at a Regional Scale. Soil Sci. Soc. Am. J. 74, 906.

Mishra, U., Lal, R., Slater, B., Calhoun, F., Liu, D., Van Meirvenne, M., 2009. Predicting Soil Organic Carbon Stock Using Profile Depth Distribution Functions and Ordinary Kriging. Soil Sci. Soc. Am. J. 73, 614–621.

Mishra, U., Riley, W.J., 2012. Alaskan soil carbon stocks: spatial variability and dependence on environmental factors. Biogeosciences 9, 3637–3645.

Molina, J. a. E., 1996. Description of the model NCSOIL. Springer Berlin Heidelberg.

Molina, J. a. E., Clapp, C.E., Shaffer, M.J., Chichester, F.W., Larson, W.E., 1983. NCSOIL, A Model of Nitrogen and Carbon Transformations in Soil: Description, Calibration, and Behavior. Soil Sci. Soc. Am. J. 47, 85–91.

Mondini, C., Coleman, K., Whitmore, A.P., 2012. Spatially explicit modelling of changes in soil organic C in agricultural soils in Italy, 2001-2100: Potential for compost amendment. Agric. Ecosyst. Environ. 153, 24–32.

Mora-Vallejo, A., Claessens, L., Stoorvogel, J., Heuvelink, G.B.M., 2008. Small scale digital soil mapping in Southeastern Kenya. CATENA 76, 44–53.

Morvan, T., Nicolardot, B., Péan, L., 2006. Biochemical composition and kinetics of C and N mineralization of animal wastes: a typological approach. Biol. Fertil. Soils 42, 513–522.

Mueller, T., Jensen, L.S., Hansen, S., Nielsen, N.E., 1996. Simulating soil carbon and nitrogen dynamics with the soil-plant-atmosphere system model DAISY. Springer Berlin Heidelberg.

Murgue, C., Therond, O., Leenhardt, D., 2015. Toward integrated water and agricultural land management: Participatory design of agricultural landscapes. Land Use Policy 45, 52–63.

Muukkonen, P., Häkkinen, M., Mäkipää, R., 2008. Spatial variation in soil carbon in the organic layer of managed boreal forest soil—implications for sampling design. Environ. Monit. Assess. 158, 67–76.

Nelson, D.W., Sommers, L.E., 1982. Total Carbon, Organic Carbon, and Organic Matter. Methods Soil Anal. Part 2 Chem. Microbiol. Prop. agronomymonogra, 539–579.

Nicolardot, B., Molina, J.A.E., Allard, M.R., 1994. C and N fluxes between pools of soil organic matter: Model calibration with long-term incubation data. Soil Biol. Biochem. 26, 235–243.

Nicolardot, B., Recous, S., Mary, B., 2001. Simulation of C and N mineralisation during crop residue decomposition: A simple dynamic model based on the C:N ratio of the residues. Plant Soil 228, 83–103.

Novák, V., Kňava, K., Šimůnek, J., 2011. Determining the influence of stones on hydraulic conductivity of saturated soils using numerical method. Geoderma 161, 177–181.

Nyssen, J., Temesgen, H., Lemenih, M., Zenebe, A., Haregeweyn, N., Haile, M., 2008. Spatial and temporal variation of soil organic carbon stocks in a lake retreat area of the Ethiopian Rift Valley. Geoderma 146, 261–268.

Osunbitan, J.A., Oyedele, D.J., Adekalu, K.O., 2005. Tillage effects on bulk density, hydraulic conductivity and strength of a loamy sand soil in southwestern Nigeria. Soil Tillage Res. 82, 57–64.

Parnaudeau, V., Génermont, S., Henault, C., Farrugia, A., Robert, P., Nicolardot, B., 2009. Measured and Simulated Nitrogen Fluxes after Field Application of Food-Processing and Municipal Organic Wastes. J. Environ. Qual. 38, 268–280.

Parton, W.J., 1996. The CENTURY model. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of Soil Organic Matter Models, NATO ASI Series. Springer Berlin Heidelberg, pp. 283–291.

Parton, W.J., Schimel, D.S., Cole, C.V., Ojima, D.S., 1987. Analysis of Factors Controlling Soil Organic Matter Levels in Great Plains Grasslands. Soil Sci. Soc. Am. J. 51, 1173–1179.

Paul, E.A., Follett, R.F., Leavitt, S.W., Halvorson, A., Peterson, G.A., Lyon, D.J., 1997. Radiocarbon Dating for Determination of Soil Organic Matter Pool Sizes and Dynamics. Soil Sci. Soc. Am. J. 61, 1058–1067.

Peltre, C., 2010. Potentialité de stockage de carbone dans les sols par apport de matières organiques exogènes. AgroParisTech, Paris.

Peltre, C., Christensen, B.T., Dragon, S., Icard, C., Katterer, T., Houot, S., 2012. RothC simulation of carbon accumulation in soil after repeated application of widely different organic amendments. Soil Biol. Biochem. 52, 49–60.

Piccolo, A., 1996. Humic Substances in Terrestrial Ecosystems. Elsevier.

Poggio, L., Gimona, A., 2014. National scale 3D modelling of soil organic carbon stocks with uncertainty propagation — An example from Scotland. Geoderma 232–234, 284–299.

Prechtel, A., von Luetzow, M., Schneider, B.U., Bens, O., Bannick, C.G., Koegel-Knabner, I., Huettl, R.F., 2009. Organic carbon in soils of Germany: Status quo and the need for new data to evaluate potentials and trends of soil carbon sequestration. J. Plant Nutr. Soil Sci. 172, 601–614.

Rasool, R., Kukal, S.S., Hira, G.S., 2008. Soil organic carbon and physical properties as affected by long-term application of FYM and inorganic fertilizers in maize–wheat system. Soil Tillage Res. 101, 31–36.

Rasse, D.P., Rumpel, C., Dignac, M.-F., 2005. Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. Plant Soil 269, 341–356.

Redin, M., Recous, S., Aita, C., Dietrich, G., Skolaude, A.C., Ludke, W.H., Schmatz, R., Giacomini, S.J., 2014. How the chemical composition and heterogeneity of crop residue mixtures decomposing at the soil surface affects C and N mineralization. Soil Biol. Biochem. 78, 65–75.

Rochette, P., Angers, D.A., Chantigny, M.H., Bertrand, N., Côté, D., 2004. Carbon Dioxide and Nitrous Oxide Emissions following Fall and Spring Applications of Pig Slurry to an Agricultural Soil. Soil Sci. Soc. Am. J. 68, 1410–1420.

Rochette, P., van Bochove, E., Prévost, D., Angers, D.A., C o ˆ té, D., Bertrand, N., 2000. Soil Carbon and Nitrogen Dynamics Following Application of Pig Slurry for the 19th Consecutive Year II. Nitrous Oxide Fluxes and Mineral Nitrogen. Soil Sci. Soc. Am. J. 64, 1396–1403.

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.

Rousk, J., Brookes, P.C., Bååth, E., 2009. Contrasting Soil pH Effects on Fungal and Bacterial Growth Suggest Functional Redundancy in Carbon Mineralization. Appl. Environ. Microbiol. 75, 1589–1596.

Rytter, R.-M., 2012. Stone and gravel contents of arable soils influence estimates of C and N stocks. CATENA 95, 153–159.

Saby, N.P.A., Arrouays, D., Antoni, V., Lemercier, B., Follain, S., Walter, C., Schvartz, C., 2008. Changes in soil organic carbon in a mountainous French region, 1990–2004. Soil Use Manag. 24, 254–262.

Saffih-Hdadi, K., Mary, B., 2008. Modeling consequences of straw residues export on soil organic carbon. Soil Biol. Biochem. 40, 594–607.

Santra, P., Kumawat, R.N., Mertia, R.S., Mahla, H.R., Sinha, N.K., 2012. Spatial variation of soil organic carbon stock in a typical agricultural farm of hot arid ecosystem of India. Curr. Sci. 102, 1303–1309.

Schwartz, D., Namri, M., 2002. Mapping the total organic carbon in the soils of the Congo. Glob. Planet. Change 33, 77–93.

Senthilkumar, K., Nesme, T., Mollier, A., Pellerin, S., 2011. Regional-scale phosphorus flows and budgets within France: The importance of agricultural production systems. Nutr. Cycl. Agroecosystems 92, 145–159.

Senthilkumar, K., Nesme, T., Mollier, A., Pellerin, S., 2012. Conceptual design and quantification of phosphorus flows and balances at the country scale: The case of France. Glob. Biogeochem. Cycles 26, GB2008.

Sikora, L.J., Yakovchenko, V., 1996. Soil Organic Matter Mineralization after Compost Amendment. Soil Sci. Soc. Am. J. 60, 1401–1404.

Simbahan, G.C., Dobermann, A., Goovaerts, P., Ping, J., Haddix, M.L., 2006. Fine-resolution mapping of soil organic carbon based on multivariate secondary data. Geoderma 132, 471–489.

Simón, N., Montes, F., Díaz-Pinés, E., Benavides, R., Roig, S., Rubio, A., 2013. Spatial distribution of the soil organic carbon pool in a Holm oak dehesa in Spain. Plant Soil 366, 537–549.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. Plant Soil 241, 155–176.

Sleutel, S., De Neve, S., Beheydt, D., Li, C., Hofman, G., 2006. Regional simulation of long-term organic carbon stock changes in cropland soils using the DNDC model: 1. Large-scale model validation against a spatially explicit data set. Soil Use Manag. 22, 342–351.

Sleutel, S., De Neve, S., Hofman, G., Boeckx, P., Beheydt, D., Van Cleemput, O., Mestdagh, I., Lootens, P., Carlier, L., Van Camp, N., Verbeeck, H., Vande Walle, I., Samson, R., Lust, N., Lemeur, R., 2003. Carbon stock changes and carbon sequestration potential of Flemish cropland soils. Glob. Change Biol. 9, 1193–1203.

Sleutel, S., De Neve, S., Németh, T., Tóth, T., Hofman, G., 2006. Effect of manure and fertilizer application on the distribution of organic carbon in different soil fractions in long-term field experiments. Eur. J. Agron. 25, 280–288.

Smith, P., Smith, J.U., Powlson, D.S., McGill, W.B., Arah, J.R.M., Chertov, O.G., Coleman, K., Franko, U., Frolking, S., Jenkinson, D.S., Jensen, L.S., Kelly, R.H., Klein-Gunnewiek, H., Komarov, A.S., Li, C., Molina, J.A.E., Mueller, T., Parton, W.J., Thornley, J.H.M., Whitmore, A.P., 1997. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 153–225.

Spielvogel, S., Prietzel, J., Auerswald, K., Kögel-Knabner, I., 2009. Site-specific spatial patterns of soil organic carbon stocks in different landscape units of a high-elevation forest including a site with forest dieback. Geoderma 152, 218–230.

Stevens, A., Miralles, I., van Wesemael, B., 2012. Soil Organic Carbon Predictions by Airborne Imaging Spectroscopy: Comparing Cross-Validation and Validation. Soil Sci. Soc. Am. J. 76, 2174–2183.

Stevens, A., Udelhoven, T., Denis, A., Tychon, B., Lioy, R., Hoffmann, L., van Wesemael, B., 2010. Measuring soil organic carbon in croplands at regional scale using airborne imaging spectroscopy. Geoderma 158, 32–45.

Stevens, A., Van Wesemael, B., Vandenschrick, G., Toure, S., Tychon, B., 2006. Detection of carbon stock change in agricultural soils using spectroscopic techniques. Soil Sci. Soc. Am. J. 70, 844–850.

Stevenson, F.J., 1994. Humus Chemistry: Genesis, Composition, Reactions. John Wiley & Sons.

Tan, Z., Lal, R., Smeck, N.E., Calhoun, F.G., Slater, B.K., Parkinson, B., Gehring, R.M., 2004. Taxonomic and Geographic Distribution of Soil Organic Carbon Pools in Ohio. Soil Sci. Soc. Am. J. 68, 1896.

Therond, O., Hengsdijk, H., Casellas, E., Wallach, D., Adam, M., Belhouchette, H., Oomen, R., Russell, G., Ewert, F., Bergez, J.-E., Janssen, S., Wery, J., Van Ittersum, M.K., 2011. Using a cropping system model at regional scale: Low-data approaches for crop management information and model calibration. Agric. Ecosyst. Environ., Scaling methods in integrated assessment of agricultural systems 142, 85–94.

Thuriès, L., Pansu, M., Larré-Larrouy, M.-C., Feller, C., 2002. Biochemical composition and mineralization kinetics of organic inputs in a sandy soil. Soil Biol. Biochem. 34, 239–250.

Tornquist, C.G., Gassman, P.W., Mielniczuk, J., Giasson, E., Campbell, T., 2009. Spatially explicit simulations of soil C dynamics in Southern Brazil: Integrating century and GIS with i_Century. Geoderma 150, 404–414.

Torstensson, G., Aronsson, H., 2000. Nitrogen leaching and crop availability in manured catch crop systems in Sweden. Nutr. Cycl. Agroecosystems 56, 139–152.

Ungaro, F., Staffilani, F., Tarocco, P., 2010. Assessing and mapping topsoil organic carbon stock at regional scale: A scorpan kriging approach conditional on soil map delineations and land use. Land Degrad. Dev. 21, 565–581.

Vågen, T.-G., Winowiecki, L.A., 2013. Mapping of soil organic carbon stocks for spatially explicit assessments of climate change mitigation potential. Environ. Res. Lett. 8, 015011.

Valé, M., 2006. Quantification et prédiction de la minéralisation nette de l'azote du sol in situ, sous divers pédoclimats et systèmes de culture français [WWW Document]. URL http://ethesis.inp-toulouse.fr/archive/00000301/ (accessed 8.19.15).

Van Soest, P.J., 1963. Use of Detergents in the Analysis of Fibrous Feeds. II. A Rapid Method for the Determination of Fiber and Lignin. J Assoc Anal Chem 46, 829–835.

Van Soest, P.J., Robertson, J.B., Lewis, B.A., 1991. Methods for Dietary Fiber, Neutral Detergent Fiber, and Nonstarch Polysaccharides in Relation to Animal Nutrition. J. Dairy Sci. 74, 3583–3597.

Van Soest, P.J., Wine, R.H., 1967. Use of detergents in the analysis of fibrous feeds. IV. Determination of plant call-wall constituents. J. Assoc. Off. Anal. Chem. 50, 50–55.

Vaudour, E., Bel, L., Gilliot, J.M., Coquet, Y., Hadjar, D., Cambier, P., Michelin, J., Houot, S., 2013. Potential of SPOT Multispectral Satellite Images for Mapping Topsoil Organic Carbon Content over Peri-Urban Croplands. Soil Sci. Soc. Am. J. 77, 2122–2139.

Vaudour, E., Gilliot, J.M., Bel, L., Lefevre, J., Chehdi, 2016. Regional prediction of soil organic carbon content over temperate croplands using visible near-infrared airborne hyperspectral imagery and synchronous field spectra. Int. J. Appl. Earth Obs. Geoinformation.

Vaudour, E., Noirot-Cosson, P.E., Membrive, O., 2015. Early-season mapping of crops and cultural operations using very high spatial resolution Pléiades images. Int. J. Appl. Earth Obs. Geoinformation 42, 128–141.

Wang, S., Huang, M., Shao, X., Mickler, R.A., Li, K., Ji, J., 2004. Vertical Distribution of Soil Organic Carbon in China. Environ. Manage. 33, S200–S209.

Wang, Z., Liu, G.-B., Xu, M.-X., Zhang, J., Wang, Y., Tang, L., 2012. Temporal and spatial variations in soil organic carbon sequestration following revegetation in the hilly Loess Plateau, China. Catena 99, 26–33.

Webb, J., Pain, B., Bittman, S., Morgan, J., 2010. The impacts of manure application methods on emissions of ammonia, nitrous oxide and on crop response—A review. Agric. Ecosyst. Environ., Special section Harvested perennial grasslands: Ecological models for farming's perennial future 137, 39–46.

Wiesmeier, M., Barthold, F., Blank, B., Kögel-Knabner, I., 2011. Digital mapping of soil organic matter stocks using Random Forest modeling in a semi-arid steppe ecosystem. Plant Soil 340, 7–24.

Wilhelm, W.W., Johnson, J.M.F., Hatfield, J.L., Voorhees, W.B., Linden, D.R., 2004. Crop and Soil Productivity Response to Corn Residue Removal. Agron. J. 96, 1–17.

Xu, G.-C., Li, Z.-B., Li, P., Lu, K.-X., Wang, Y., 2013. Spatial variability of soil organic carbon in a typical watershed in the source area of the middle Dan River, China. Soil Res. 51, 41–49.

Xu, S., Shi, X., Zhao, Y., Yu, D., Wang, S., Tan, M., Sun, W., Li, C., 2012. Spatially explicit simulation of soil organic carbon dynamics in China's paddy soils. Catena 92, 113–121.

Xu, X., Liu, W., Zhang, C., Kiely, G., 2011. Estimation of soil organic carbon stock and its spatial distribution in the Republic of Ireland. Soil Use Manag. 27, 156–162.

Yokozawa, M., Shirato, Y., Sakamoto, T., Yonemura, S., Nakai, M., Ohkura, T., 2010. Use of the RothC model to estimate the carbon sequestration potential of organic matter application in Japanese arable soils. Soil Sci. Plant Nutr. 56, 168–176.

Yoo, K., Amundson, R., Heimsath, A.M., Dietrich, W.E., 2006. Spatial patterns of soil organic carbon on hillslopes: Integrating geomorphic processes and the biological C cycle. Geoderma 130, 47–65.

Yost, R.S., Uehara, G., Fox, R.L., 1982. Geostatistical Analysis of Soil Chemical Properties of Large Land Areas. II. Kriging. Soil Sci. Soc. Am. J. 46, 1033–1037.

Young, R.R., Wilson, B., Harden, S., Bernardi, A., 2009. Accumulation of soil carbon under zero tillage cropping and perennial vegetation on the Liverpool Plains, eastern Australia. Soil Res. 47, 273–285.

Yuan, Z., Gazol, A., Lin, F., Ye, J., Shi, S., Wang, X., Wang, M., Hao, Z., 2013. Soil organic carbon in an old-growth temperate forest: Spatial pattern, determinants and bias in its quantification. Geoderma 195–196, 48–55.

Yu, D.S., Shi, X.Z., Wang, H.J., Sun, W.X., Chen, J.M., Liu, Q.H., Zhao, Y.C., 2007. Regional patterns of soil organic carbon stocks in China. J. Environ. Manage. 85, 680–689.

Zhang, C., McGrath, D., 2004. Geostatistical and GIS analyses on soil organic carbon concentrations in grassland of southeastern Ireland from two different periods. Geoderma 119, 261–275.

Zhao, Y., Shi, X., Yu, D., Wang, H., Sun, W., 2005. Uncertainty assessment of spatial patterns of soil organic carbon density using sequential indicator simulation, a case study of Hebei province, China. Chemosphere 59, 1527–1535.

Zhou, C., Zhou, Q., Wang, S., 2003. Estimating and Analyzing the Spatial Distribution of Soil Organic Carbon in China. AMBIO J. Hum. Environ. 32, 6–12.

Présentation du territoire d'étude

Pour cette étude, nous avons retenu le territoire de la Plaine de Versailles, situé à 30 km à l'ouest de Paris (Fig. 1). Il couvre 221 km², dont environ 100 km² de surface agricole utile (coin NW : 48°56'32''N, 1°50'12''E ; coin SE : 48°46'53''N, 02°07'22''E). De par sa situation périurbaine, l'agriculture de ce territoire est au cœur des enjeux liant ville et agriculture et en particulier les enjeux concernant le recyclage des PRO urbains. Ces enjeux sont également accrus du fait du contexte de dégradation des teneurs en carbone organique des sols de grandes cultures du Bassin Parisien. Ce territoire a été déjà particulièrement étudié en cause notamment la présence du laboratoire INRA de Thiverval-Grignon. Il inclut géographiquement des sites bénéficiant d'une longue antériorité d'acquisition de données sur les sols, en particulier la Ferme expérimentale de Grignon et surtout le dispositif expérimental QualiAgro (48°53'47''N, 1°58'20''E, commune de Feucherolles), l'un des sites principaux du SOERE-PRO. Ce dispositif au champ, d'une superficie de 6 ha, est un dispositif de longue durée initié en 1998 par l'UMR EGC (aujourd'hui ECOSYS), en collaboration avec Véolia Environnement, afin de suivre les effets d'épandages successifs de différents PRO sur les propriétés physico-chimiques des sols. Des travaux ont été menés sur l'insertion des PRO dans les systèmes de cultures du territoire (Dhaouadi, 2014). Ils ont porté sur le potentiel de substitution des PRO à de la fertilisation azotée de synthèse dans quelques contextes de sols et de systèmes de cultures du territoire. Cette thèse antérieure a permis le recensement et l'étude des PRO disponibles sur le territoire, et la caractérisation des pratiques d'épandages de PRO par les agriculteurs du territoire.

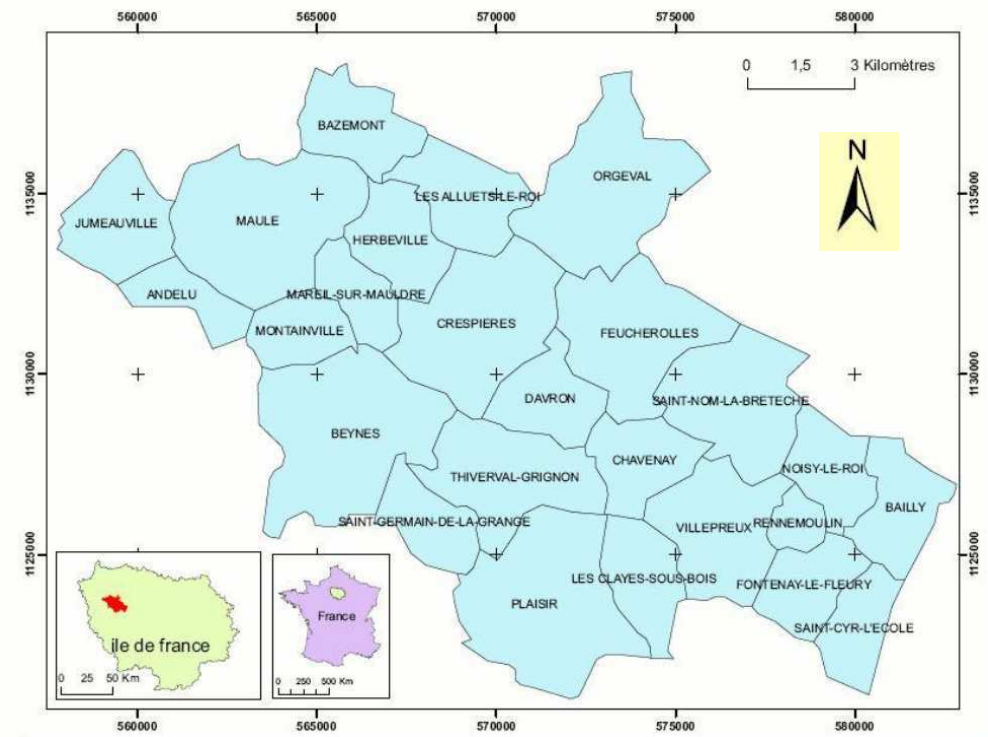


Figure 1. Localisation des 25 communes de l'association patrimoniale de la Plaine de Versailles et du Plateau des Alluets. Réalisation J.M. Gilliot.

Ce territoire est caractérisé par le regroupement de 25 communes à l'ouest de Versailles au sein de l'association Patrimoniale de la Plaine de Versailles et du Plateau des Alluets (APPVPA) aujourd'hui dénommée association patrimoniale de la Plaine de Versailles (http://www.plainedeversailles.fr) (Fig.1). Cette association a été créée en 2004, en réaction au classement de 2600 ha dans le prolongement de la perspective du château de Versailles, avec la volonté d'encourager une logique de développement autant que de protection. Elle a pour objectif de « créer un espace de communication pour faire se rencontrer, puis rassembler, les personnes physiques et morales représentatives des différents intérêts locaux, afin de réfléchir, étudier et formuler des propositions visant à l'établissement d'un projet de développement durable, commun aux agriculteurs et aux citadins, sur les territoires de la Plaine de Versailles et du Plateau des Alluets » (A.P.P.V.P.A, 2015).

L'agriculture occupe environ 9900 ha (Fig. 2), soit 45% du territoire, et comptait 82 exploitations agricoles en 2010. Elle est consacrée essentiellement aux grandes cultures (colza, blé, maïs, orge) avec 60 exploitations de grande taille (en moyenne 130 ha) représentant 90% de la SAU ; le 10% de la SAU restant est occupé par des exploitations maraîchères et/ou arboricoles et, dans une moindre mesure, d'exploitations d'élevage (3 seulement : un élevage de bovin-caprin-ovin, deux éleveurs de volailles). L'équitation y est particulièrement développée avec 23 centres équestres soit environ 910 chevaux (Dhaouadi, 2014). Son classement en zone vulnérable, au titre de la directive nitrate ayant pour but de limiter la pollution des eaux par les nitrates, encadre les pratiques de fertilisation et d'épandage de son

agriculture. Les agriculteurs doivent respecter l'équilibre de la fertilisation azotée et ne peuvent apporter plus de 170 kg N ha^{-1} via l'épandage de PRO.

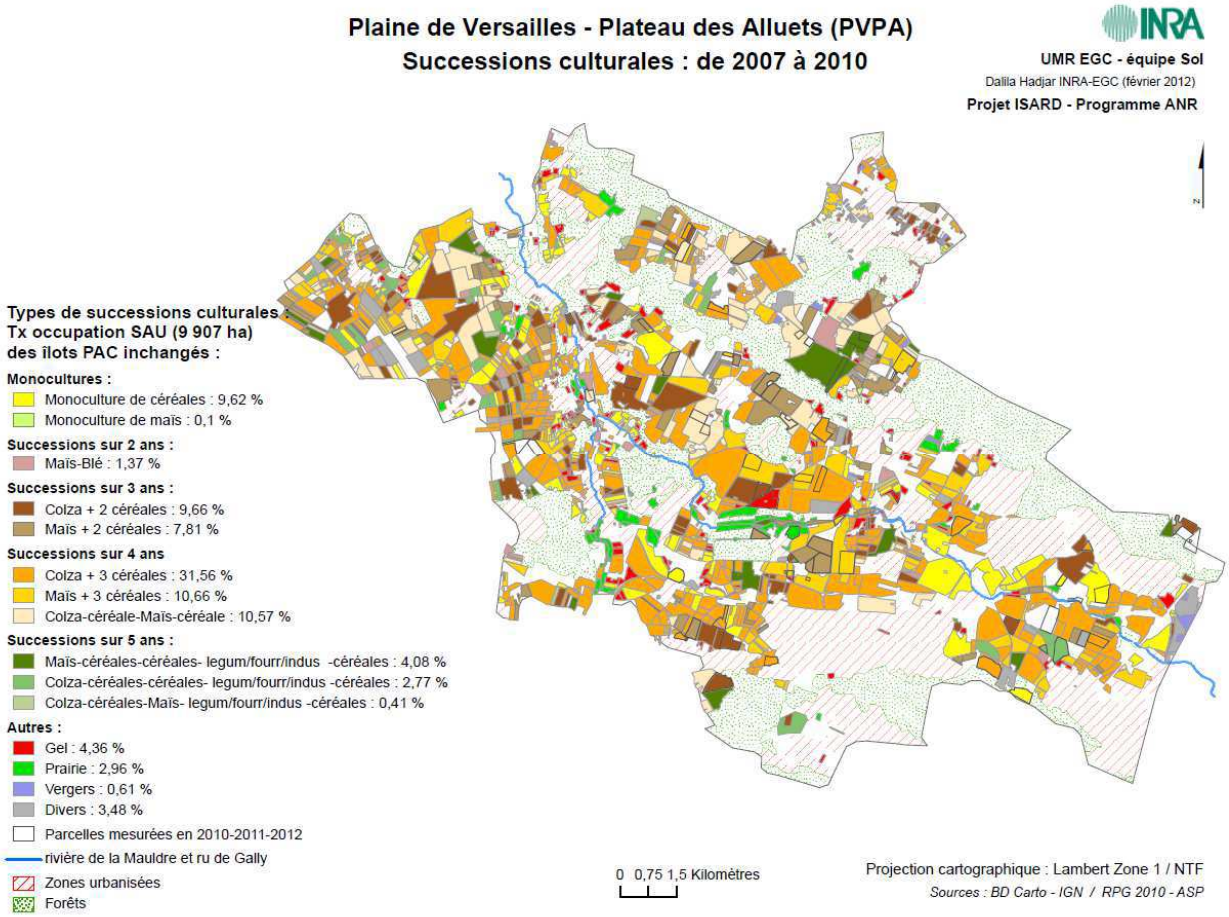


Figure 2. Carte des successions culturales de la Plaine de Versailles et du Plateau des Alluets (auteurs Dalila Hadjar et Christine Aubry).

Ce territoire est formé par deux niveaux juxtaposés de plateaux sédimentaires (120-180 m), de direction générale sud-est/nord-est, légèrement disséqués par des vallées : la vallée du Ru de Gally qui prend sa source au Grand Canal du château de Versailles et se jette dans la Mauldre, affluent de la Seine ; la vallée de la Mauldre qui parcourt l'ouest du territoire du sud au nord. Ces plateaux reposent sur un substratum de craie du Crétacé supérieur, que surmontent successivement des substrats tertiaires : marnes, puis calcaires, sables siliceux et enfin argile à meulière au sommet de la lithoséquence (Vaudour et al., 2014). Les sols dominants étudiés dans le cadre de cette thèse sont les Luvisols des plateaux supérieurs (Haplic luvisol retic), les luvisols des plateaux inférieurs (Haplic luvisol), des calcosols (calcaric cambisol) et des colluviosols sur sables (arenic cambisol)(Crahet, 1992; Vaudour et al., 2014) (Fig. 3).

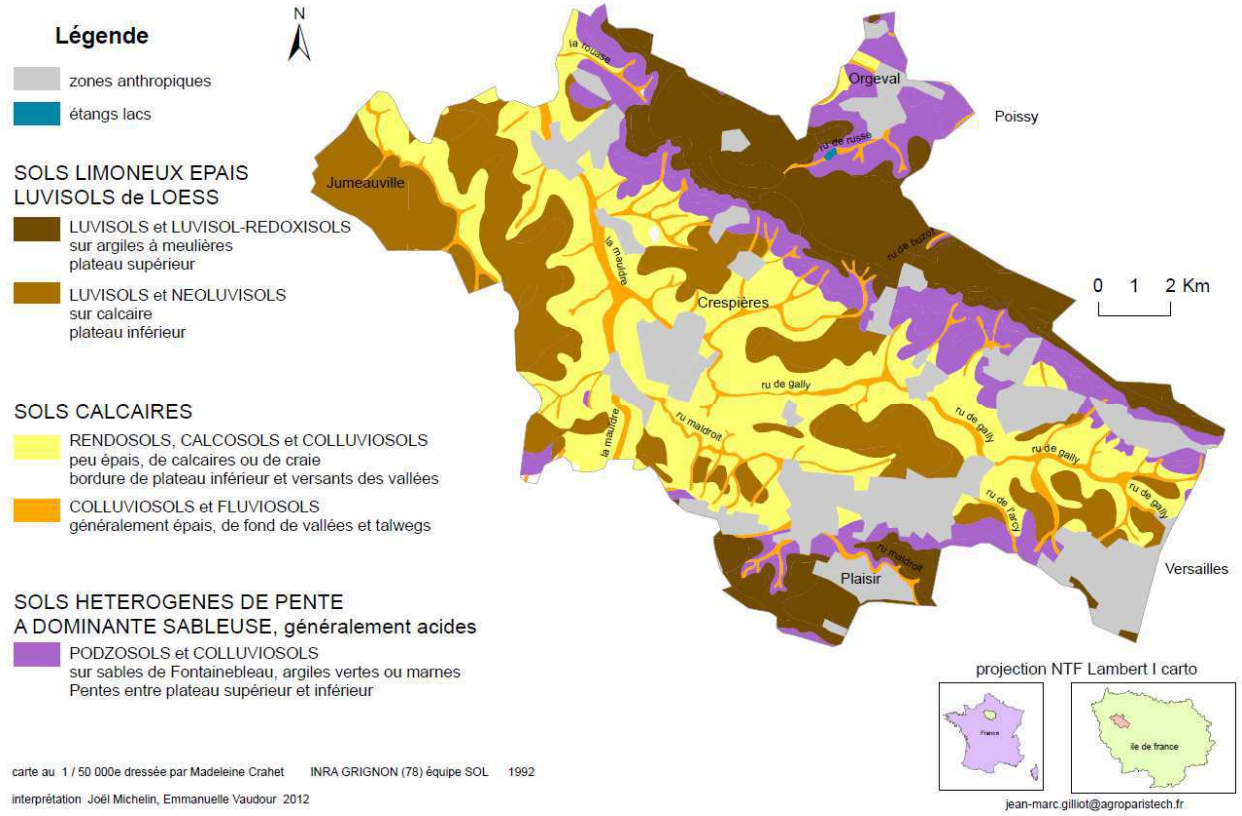


Figure 3. Carte simplifiée des sols de la Plaine de Versailles et du Plateau des Alluets (auteurs J. Michelin, E. Vaudour, 2012 ; d'après Crahet , 1992)

Les teneurs en carbone organique des sols agricoles ont été étudiées antérieurement : elles ont fait l'objet de nombreuses mesures permettant d'ailleurs leur spatialisation via des modèles géostatistiques (Hamiache et al., 2012; Fig. 4) et ou via des modèles de spectrotransfert grâce à l'acquisition d'images aériennes et satellitaires couplées à des mesures de réflectance de terrain (Vaudour et al., 2013, 2016).

Ses faibles teneurs comprises entre 6 et 36 g kg^{-1} (Vaudour et al., 2016) sont d'ailleurs représentatives ndes sols agricoles du Bassin Parisien (Arrouays et al., 2001).

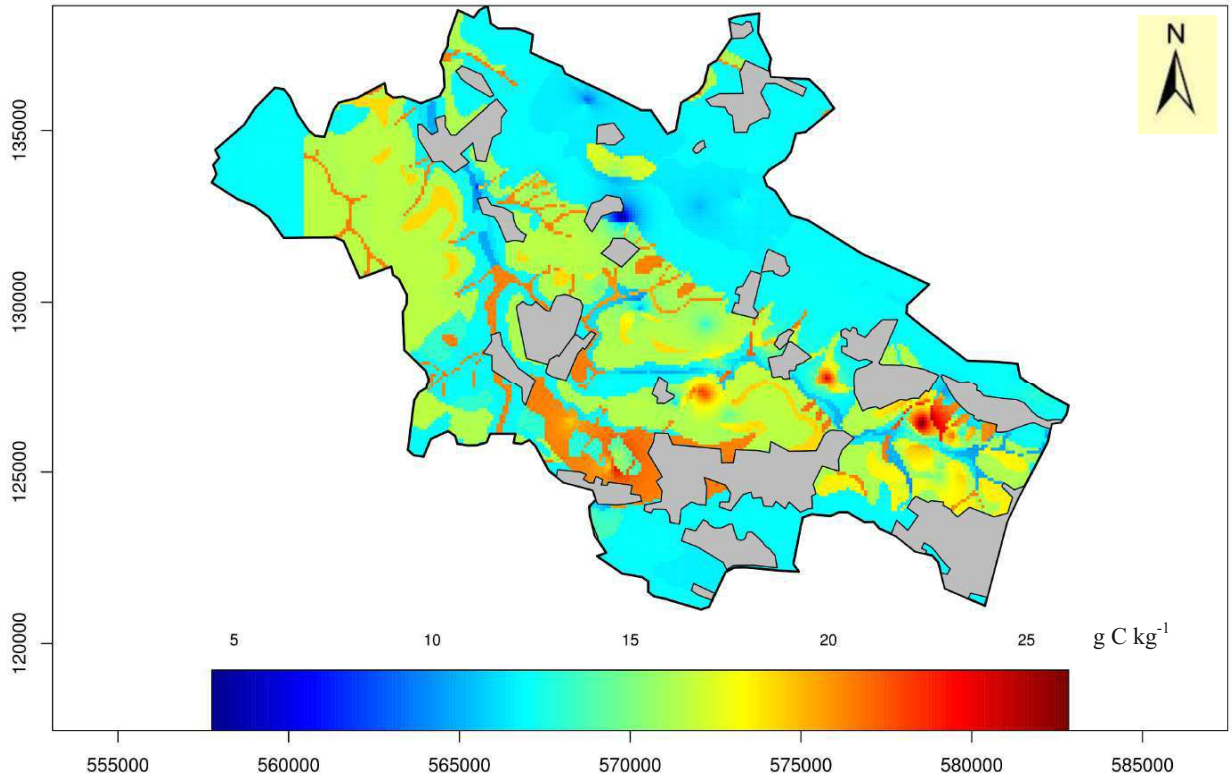


Figure. 4 Cartographie des teneurs en carbone organique des sols de la Plaine de Versailles (Hamiache et al., 2012)

Références

A.P.P.V.P.A, 2015. http://www.plainedeversailles.fr/association.php.

Arrouays, D., Deslais, W., Badeau, V., 2001. The carbon content of topsoil and its geographical distribution in France. Soil Use and Management 17, 7–11. doi:10.1111/j.1475-2743.2001.tb00002.x

Crahet, M., 1992. Soil Map of the Versailles Plain. Scale 1:50 000 (Internal report). Institut National Agronomique Paris-Grignon.

Dhaouadi, A.K., 2014. Insertion des Produits Résiduaires Organiques dans les systèmes de culture : Cas des systèmes céréaliers de la Plaine de Versailles et du Plateau des Alluets. Paris, AgroParisTech.

Hamiache, J., Bel, L., Vaudour, E., Gilliot, J.M., 2012. Spatial stochastic modeling of topsoil organic carbon content over a cultivated peri-urban region, using soil properties, soil types and a digital elevation model. Digital Soil Assessments and Beyond 161–164.

Vaudour, E., Bel, L., Gilliot, J.M., Coquet, Y., Hadjar, D., Cambier, P., Michelin, J., Houot, S., 2013. Potential of SPOT Multispectral Satellite Images for Mapping Topsoil Organic Carbon Content over Peri-Urban Croplands. Soil Science Society of America Journal 77, 2122–2139. doi:10.2136/sssaj2013.02.0062

Vaudour, E., Gilliot, J.M., Bel, L., Bréchet, L., Hamiache, J., Hadjar, D., Lemonnier, Y., 2014. Uncertainty of soil reflectance retrieval from SPOT and RapidEye multispectral satellite images using a per-pixel bootstrapped empirical line atmospheric correction over an agricultural region. International Journal of Applied Earth Observation and Geoinformation 26, 217–234. doi:10.1016/j.jag.2013.07.003

Vaudour, E., Gilliot, J.M., Bel, L., Lefevre, J., Chehdi, 2016. Regional prediction of soil organic carbon content over temperate croplands using visible near-infrared airborne hyperspectral imagery and synchronous field spectra. International Journal of Applied Earth Observation and Geoinformation,.

Chapter 2. Modelling the long-term effect of urban waste compost applications on carbon and nitrogen dynamics in temperate cropland

P.E. NOIROT-COSSON[1], VAUDOUR E.[1], GILLIOT J.M.[1], GABRIELLE B.[1], HOUOT S.[1*]

[1] UMR ECOSYS, AgroParisTech, INRA, Université Paris-Saclay, 78850, Thiverval-Grignon,

Article published in Soil Biology & Biochemistry

1. Abstract

The recycling in agriculture of Exogenous Organic Matter (EOM) issued from organic waste treatment is a promising way to restore soil organic matter (SOM) content in intensively managed soils. EOM applications to crop fields may also be used as substitute to synthetic fertilisers. The CERES-EGC mechanistic model was used to simulate the effects of repeated applications of urban waste composts and manure over 13 years on both soil carbon (C) and nitrogen (N) dynamics in the soil-crop-water-air system of the long-term field experiment QualiAgro. Several EOM were considered: a farmyard manure, FYM; a municipal solid waste compost, MSW; a bio-waste compost, BIO; a co-compost of green waste and sewage sludge, GWS. Each EOM application brought the equivalent of 220-400 kg N ha^{-1}. The sub-model NCSOIL was parameterised from C and N mineralisation kinetics of EOM measured during incubations of soil-EOM mixtures in controlled conditions. The simulation correctly reproduced the experimental kinetics. When transposing these parameters into the CERES-EGC model, C storage at the field scale was well simulated, together with crop N uptake and yields, as well as soil mineral N contents despite a slight overestimation. The GWS compost generated the highest C storage over the 13y-period and MSW the lowest with 65% and 36% of the Exogenous Organic Carbon (EOC) applied incorporated into the soil organic C, respectively. The GWS and MSW had the highest potential of N loss because of high mineral N content and a high potential of N mineralisation, respectively in contrast to FYM and BIO. MSW had also the highest apparent N use efficiency (48.8%) thanks to a high potential of mineralisation (76.3% of organic N applied). The achieved CERES-EGC parameterisation offers promising prospects for predicting the effects of a larger panel of EOM, and for further using this soil-plant-water-atmosphere model to manage EOM application practices at the regional scale in compliance with crop production and environmental aims.

Key-words: Compost, NCSOIL, CERES-EGC, C and N dynamics, long-term experiment, parameterisation

2. Introduction

Low level of soil organic matter (SOM) in cropped soils has become a worldwide threat for soil sustainability, notably in Europe (Ciais et al., 2010). Increasing soil organic carbon (SOC) content may contribute to the carbon (C) storage into soils and the mitigation of climate change (Lal et al., 2004b) while improving soil chemical, physical and biological properties (Kundu et al., 2007), including increased nutrient availability and soil water retention, reduced risk of crust formation and soil erosion, therefore enhanced crop productivity (Lal et al., 2004a). One strategy to increase SOC is amending soils with exogenous organic matter (EOM) defined as organic residues issued from agriculture (manures, litters, slurries), from urban activities (sewage sludge, biowaste composts, green waste composts), or from industries (Marmo et al., 2004). These EOM have variable efficiencies at increasing SOC and enhancing N availability, depending on their biochemical characteristics and C and N contents (Peltre et al., 2012) and on the pedo-climatic context of their application. Potential trade-off may also take place through mineral N leaching, NH_3 volatilisation or greenhouse gas emission (Parnaudeau et al., 2009). EOM applications may also lead to soil, water, atmosphere or plant pollutions with various xenobiotic compounds (Houot et al., 2014).

Mechanistic models make possible to simultaneously simulate all C and N fluxes in a soil-plant-water-air system after repeated EOM applications while considering a wide range of soils and EOM types (Smith et al., 1997; Gabrielle et al., 2004). Process-based models were parameterised for the purpose of simulating SOC stocks after EOM application: Roth-C (Guo et al., 2007; Peltre et al., 2012), Century (Falloon et al., 2002), DAISY (Bruun et al., 2003). Modelling was also used to simulate the N behaviour after applications of EOM (Chalhoub et al., 2013). The effects of EOM application on N dynamics were often studied and simulated on a short term basis to determine potential N availability for crops. In case of repeated applications, simulation models make possible to distinguish the increase in mineral N availability related to the last application from the enhanced N availability due to mineralisation of the accumulated OM in soils (Chalhoub et al., 2013). The QualiAgro long term experiment was set up in 1998 in Ile de France to study the effects of repeated EOM amendments in a cropped soil on SOM stocks, nutrients availability for crops and potential accumulation of contaminants in soil. Gabrielle et al. (2004, 2005) used CERES-EGC, a modified version of the crop model CERES (Jones et al., 1986) in which the soil C–N module was substituted with the NCSOIL model (Molina et al., 1983) to simulate the first years of the experiment (4 years).

In most models, OM is split into several compartments or “pools” characterised with different parameters. The parameterisation of EOM C and N behaviour is quite challenging as they may be composed of a broad variety of biodegradable materials, and calibration may highly vary among EOM with different constituents (Gabrielle et al., 2004). Recently, Lashermes et al. (2009) elaborated the indicator of residual organic carbon in soil (I_{ROC} indicator) aiming to predict exogenous organic carbon

(EOC) potential contribution to soil organic C and an I_{ROC} database including many EOM categories was constructed. Peltre et al. (2012) used the I_{ROC} indicator to parameterise the partition of EOM organic C into the entry pools of Roth-C thus making possible to predict SOC stocks evolution based on scenarios of EOM applications. The N availability from EOM is conditioned by its content in mineral N, its mineralisation rate related with the biochemical composition and C/N ratio of the EOM (Janssen, 1996). Additionally, mineral N availability is also increased in amended soils because of the increased SOM stocks after repeated EOM applications. Many authors have used optimisation procedure to determine EOM unknown parameters in the NCSOIL model such as the C/N ratio for the different EOM pools (Beraud et al., 2005; Burger and Venterea, 2008; Antil et al., 2011). However, optimisation procedures may not be generic as they rely on specific datasets and often require time-consuming experimental data such as kinetics of C and N mineralisation in soil. Under given climatic conditions, the rates of decomposition of organic materials in soil primarily depend on their composition, and are not clearly dependent of the soil type (Hadas and Portnoy, 1994), or vary inconsistently with soil properties (Smith et al., 2007).

Using the CERES-EGC model, and revisiting the parameterisation of the OM sub-model NCSOIL, the effect of 13 years of different EOM applications on C and N dynamics was studied in the QualiAgro experiment, in order to predict the C storage, the increase in N availability for crops and plant response and finally the N leaching risk. The objectives of this paper were: (i) to evaluate a generic parameterisation method to be able to simulate the C and N behaviours of EOM pools based on both I_{ROC} indicator and kinetics of C and N mineralisation during laboratory incubations in soil, (ii) to evaluate the capacity of NCSOIL-parameterisation based on laboratory data to be transposed into the CERES-EGC model accounting for long-term plant response to N mineralisation, C storage and nitrate leaching in field condition, (iii) to compare the EOM through their effects on C and N dynamics in cropped soil.

3. Materials and methods

3.1. The QualiAgro long-term experiment

The QualiAgro experiment (INRA-Veolia collaboration), located at Feucherolles 50 km west of Paris, France (48°53’ N, 1°58’ E) was set up in 1998 on a hortic glossic Luvisol (IUSS Working Group WRB, 2014) with a silt-loam texture (15.0% clay, 78.3% silt) and a pH of 6.9 in the surface horizon (0-30 cm). The crop rotation was maize (*Zea mays L.*) - wheat (*Triticum aestivum L.*) but barley was cropped in 2006-2007 due to prognosis of regional attack of *Diabrotica virgifera* to maize that year. Sowings occurred early May for maize, early November for wheat and mid-October for barley. Mineral fertiliser applications occurred once a year for maize (early May) and barley (early March), twice for wheat (early March, early April). Crop yields and biomass of residues at harvest time were

determined by sampling 8 surfaces of 1 m² for maize and 0.5 m² for wheat and barley in each plot. Dry matter and N content (elemental analysis) were analysed in grains and aerial crop residues.

The 6-ha experiment followed a split-plot design comprising 40 plots: 4 replication blocks for each treatment including (i) 4 EOM applications and a control treatment without EOM application, crossed with (ii) 2 mineral N treatments corresponding to either "optimal" (N_{opt}) or "minimal" (N_{min}) dose of a N solution (50% urea-50% ammo-nitrate). N_{opt} fertilisation corresponded to applications of 140 (±37) kg N ha^{-1} for wheat depending on the year, 82 (±34) kg N ha^{-1} for maize and 82 kg N ha^{-1} for barley, whereas for N_{min} it was 61 (±31) kg N ha^{-1} for wheat, 0 (±0) kg N ha^{-1} for maize and 31 kg N ha^{-1} for barley. The EOM applications started in September 1998 and occurred every 2 years in September on wheat stubble at a rate of 4 Mg C ha^{-1}. An additional EOM spreading occurred in 2007 after barley harvest. The four block-lines were separated by a 25 m wide buffer strip and within each block, plots (10x45m each) were 6 m apart from each other. The 4 EOM described in Annabi et al., (2007) included a dairy farmyard manure (FYM), a municipal solid waste compost (MSW) issued of the composting of mechanically separated organic fractions from residual waste after selective collection of dry and clean packaging, a bio-waste compost (BIO) obtained by composting green wastes and a source-separated organic fraction of municipal solid waste, and a co-compost of green waste (70%) and sewage sludge (30%) (GWS), the green wastes being collected from private and public gardens.

Mineral N content in the soil was determined in each plot at 3 depths (0-30, 30-60 and 60-90 cm) for 3 soil cores, each pooled from 3 separate samplings with an automatic auger. Mineral N was then extracted with KCl (1 $mol.L^{-1}$) and analysed on a continuous flow analyser (Skalar, the Netherlands) using the Berthelot method for $N\text{-}NH_4$ and the Griess method for $N\text{-}NO_3$. The sampling occurred 3 times a year: at the end of the drainage period early February, after harvest in August and at the beginning of the drainage period in November for the wheat years; early February, before sowing in April and after harvest in November for the maize years. To quantify water and N leaching, glass fibre wick and stainless steel plate lysimeters were installed at 45 cm in 2004 in one plot of each treatment receiving optimum mineral N. Waters were collected up to 8 times between December and June, depending on the year rainfall frequency and intensity.

The soils were sampled in the ploughed layer (0-30 cm) in each plot (10 pooled samples) before each EOM application (1998-2000-2002-2004-2006-2007-2009-2011) at the end of August for analysis of total organic carbon (TOC) and total nitrogen contents by elemental analysis of air-dried 200-µm ground samples (AFNOR, 1995 for C and AFNOR, 1998 for N). For TOC analysis, carbonates were removed with hydrochloric acid treatment (Bisutti et al., 2004).

Before starting the field experiment, the potential mineralisation of total soil organic C and N was assessed during 91-day incubations under controlled conditions (temperature of 28°C and gravimetric water content 23%) following a standard method (AFNOR, 2009b). Fresh 4mm-seived soil samples (equivalent to 25 g of dry soil) representative of the initial conditions, were incubated in air-tight 500 ml jars, with four replicates, for either the organic C or N mineralisation measurements during

separate parallel experiments. The C-CO_2 was trapped in 10 ml of NaOH (0.5 mol.L^{-1}), which was periodically sampled and replaced. The C-CO_2 trapped in NaOH was analysed by colorimetry on a continuous flow analyser (Skalar, The Netherlands). The opening renewed the jar atmosphere and maintained aerobic conditions. On the same dates, the mineral N (N-NH_4^+ and N-NO_3^-) was extracted with 100 ml of 1 mol.L^{-1} KCl and then analysed by colorimetry as previously described.

3.2. EOM characterisation

3.2.1. Physico-chemical analysis and biochemical fractionation

The EOM were sampled for physico-chemical characterisation at the time of each application. The FYM always came from the same farm and the composts came from 2 to 3 plants. Process information and average physico-chemical characteristics are summarised for each EOM in Table 1.

Table 1. Characteristics of the Exogenous Organic Matters (EOM) and application rates.

	BIO[a]	GWS	FYM	MSW
Composting process				
Fermentation phase	1-3 months	1-3 months		3-4 weeks
Maturation phase	2-6 months	1-4 months		5 days
EOM characteristics				
DM[b] (%RM)	70.1 (±9.1)[c]	63.3 (±8.7)	41.1 (±9.7)	66.6 (±15.5)
Organic C (g C kg^{-1} DM)	202.6 (±39.8)	263.5 (±68.3)	317.8 (±62.8)	300.1(±93.5)
Total N (g N kg^{-1} DM)	17.3 (±3.4)	24.2 (±3.0)	23.2 (±6.0)	18.5 (±6.0)
Mineral N (mg N kg^{-1} DM)	989 (±843)	3790 (±851)	1749 (±1688)	1676 (±1379)
Organic C/N ratio	12.5 (±1.6)	12.9 (±2.7)	15.1 (±2.8)	18.5 (±4.3)
Van Soest fractions (% organic matter)				
Total soluble fraction (SOL)	44.9 (±6.6)	45.5 (±7.9)	39.1 (±6.8)	41.5 (±7.9)
Hemicellulose (HEM)	4.3 (±2.4)	5.6 (±2.1)	12.1 (±3.3)	6.8 (±2.6)
Cellulose (CEL)	21.5 (±2.9)	21.1 (±8.7)	27.0 (±7.6)	33.7 (±9.6)
Lignine (LIG)	29.3 (±8.3)	27.8 (±10.2)	21.8 (±3.7)	18.0 (±5.7)
I_{ROC} (% organic C)	75.5 (±6.7)	77.6 (±9.3)	66.6 (±7.5)	49.0 (±13.9)
Application rate (Mg DM ha^{-1})	19.1 (±4.4)	16.5 (±2.8)	13.2 (±2.1)	12.0 (±3.4)

[a]BIO: bio-waste compost, GWS: green waste and sewage sludge co-compost, FYM: dairy farmyard manure, MSW: municipal solid waste compost.
[b]DM: dry matter; RM: raw matter.
[c]Mean of 8 applications and standard deviation.

Mineral N was extracted with 200 mL of 1 mol.L^{-1} KCl from 50 g fresh samples and analysed as described previously. The EOM were then dried and ground to 1 mm before further analysis. Since mineral N is partly lost during the drying, it was also measured on the dried samples. Total organic C (C_{EOM}) and total N contents were measured on 200 µm-ground dry samples by elemental analysis for all EOM (AFNOR 1995 for C and AFNOR 2002 for N). Organic N (N_{EOM}) was calculated by subtracting mineral N from total N, both measured on dry samples, and corresponding organic C/N

ratios (CN_{EOM}) were then calculated. Total N of fresh samples was calculated by adding organic N and mineral N measured on fresh samples. Total OM was measured by dry mass loss on ignition at 550°C. The biochemical fractionation was performed using a modified Van Soest method (AFNOR, 2009a). Four biochemical fractions were isolated: “soluble” (SOL), "hemicellulose-like” (HCE), “cellulose-like” (CEL) and “lignin-like" (LIG). The indicator of residual organic carbon, I_{ROC}, was then calculated (Lashermes et al., 2009). It represents the proportion of the C_{EOM} that would be retained in soil for decades and is calculated as:

$$I_{ROC} = 44.5 + 0.5 \times SOL - 0.2 \times CEL + 0.7 \times LIC - 2.3 \times C_{3d} \qquad [1]$$

Where I_{ROC} is expressed as % of C_{EOM}; *SOL*, *CEL* and *LIC* are expressed in % total organic matter; C_{3d}, also expressed in % of C_{EOM}, the proportion of the C_{EOM} mineralised after 3 days of soil-EOM mixtures incubation (AFNOR, 2009a).

3.2.2. C and N mineralisation under laboratory controlled conditions

The kinetics of C and N mineralisation were measured for all applied organic amendments during 91-day incubations of soil-organic amendment mixtures under similar conditions and as previously described for soil alone. The soil used for incubations was sampled in the plough layer (0-30 cm) from a control plot of the field experiment. A quantity of 50 g (for 1998) then 25 g (for all other years) of soil was mixed to dry and 1mm-ground EOM (masses equivalent to 100 then 50 mg of organic C, respectively). Mineral N was added at the beginning of the incubation (25 mg $N\text{-}NO_3$ kg^{-1} soil) to prevent a mineral N stress affecting EOM organic matter degradation.

3.3. Model description

3.3.1. CERES-EGC model

The CERES-EGC model (Gabrielle et al., 1995, 2002, 2005, 2006) was adapted from the mechanistic soil crop model CERES (Jones et al., 1986). It includes three main modules: one physical module for heat, water and solutes transfers in soil as well as plant water uptake and evapo-transpiration led by climatic demand. The CERES-EGC model uses Darcy’s law to simulate water transfers in soil adapted for fine textured soil (Gabrielle et al., 1995). The second module, NCSOIL (described below), simulates soil OM dynamics in the ploughed layer. The third module describes crop growth thanks to crop net photosynthesis linearly related to intercepted radiation according to Monteith approach depending on leaf area index based on Beer’s law of diffusion in a turbid media. Crop N uptake is computed through a supply/demand scheme, in which soil supply depends on nitrate and ammonium soil concentrations and root length density (Gabrielle et al., 2005). Additional environmental output calculations such as nitrate leaching and gaseous emissions of N_2O, ammonia and nitrous oxides can also be calculated with additional modules (Gabrielle et al., 2006).

CERES-EGC runs on a daily time step using daily climatic data including solar radiation, rainfall, mean air temperature, potential evapotranspiration. It also requires soil parameters for heat and water transfers, parameters describing SOM characteristics as defined for the NCSOIL module, including added EOM characteristics when necessary and crop management information (sowing date and depth, fertiliser application…).

3.3.2. NCSOIL model

The NCSOIL model (Molina et al., 1983; Barak et al., 1990) was used to simulate mineralisation of C and N from SOM and from EOM during the lab incubations, then the defined parameterisation was used to simulate C and N behaviour in the soil-crop-water-air system with the coupled models CERES-EGC-NCSOIL (Gabrielle et al., 1995).

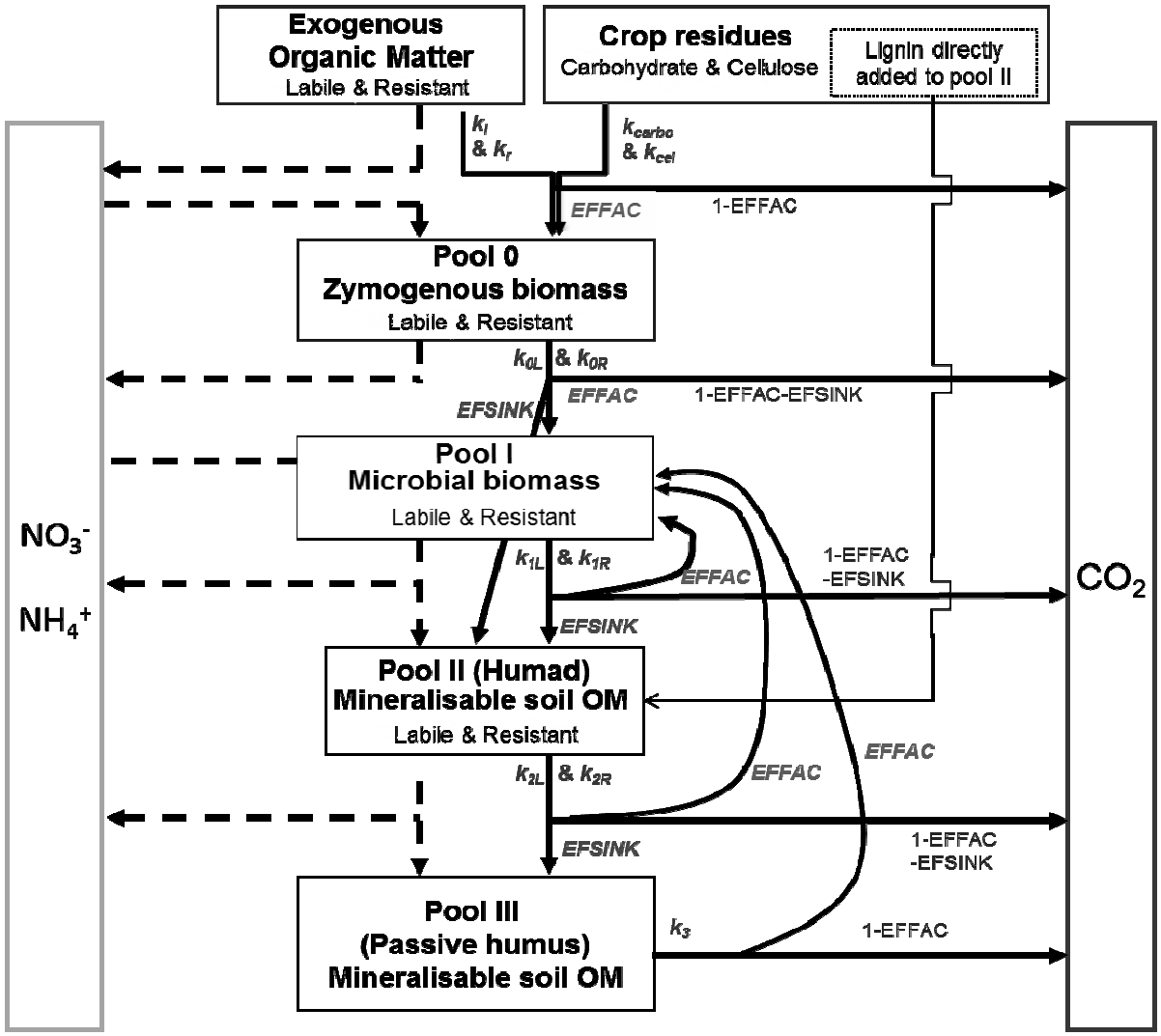


Figure 1. Structure of the NCSOIL model.
k stands for decomposition rates: k_l and k_r for the labile and resistant EOM fractions, k_{carbo} and k_{cel} for the carbohydrate and cellulose pools of crop residues, k_{0R} and k_{0L} for the resistant and labile fractions of pool 0, k_{1R} and k_{1L} idem for pool I, k_{2R} and k_{2L} idem for pool II, k_3 for pool 3, EFFAC for the proportion of decomposed OM recycled in pool I or in pool 0 for residues and EOM, EFSINK for the proportion of the decomposed OM recycled in the following mineralisable soil pool.

NCSOIL simulates crop residue and EOM decomposition after incorporation into soil, nitrification, immobilisation and mineralisation of N, along with SOM decomposition and transformation, by means of a system of controlling rates and decisional switches amongst a series of pools or compartments (Haskett et al., 1986). The SOM flow series implies 4 pools involved in N and C turnover and represented in Fig. 1: pool 0, the "zymogenous biomass", pool I, representing the endogenous microbial biomass, pool II, a fraction of humified OM termed "mineralisable SOM", pool III, corresponding to highly-humified OM with a very low decomposition rate. Each pool p*i*, is defined by a C content (or "pool size"), Cp_i, a C to N ratio, $CNp_i = Cp_i/Np_i$ (where Np_i is the N content), a decomposition rate, kp_i, a microbial use efficiency factor (EFFAC(i)) and a humification factor (EFSINK(i)). EFFAC represents the proportion of decomposed OM recycled in pool I or in pool 0 for residues and EOM and EFSINK the proportion of the decomposed OM recycled in the following mineralisable soil pool. The complement of degraded fraction (1-EFFAC-EFSINK) of each pool is evolved as CO_2. All pools decompose following first-order kinetics:

$$-(dCp_i / dt) = kp_i \times Cp_i \qquad [2]$$

The N flows follow the C flows respecting the C/N ratio of the pool being decomposed, mineral N being added or withdrawn by the following pool to maintain its C/N ratio.

EOM and soil pools 0 and I are divided into labile and resistant "fractions" with different decomposition rates. To take into account the effect of soil sieving on mineralisation in lab condition, a labile pool II can also be created with the "till" parameter, representing the proportion of pool II directed into the labile fraction. Crop residues are divided into 3 fractions, carbohydrates, cellulose and lignin with different decomposition rates. In this study, the lignin was directly incorporated into pool II, along with attributing to pool II the same decomposition rate as lignin, which allowed to maintain pool II size and to avoid a constant lignin accumulation over many years of simulation. The EOM were also divided into labile and resistant fractions with different decomposition rates (k_l and k_r). Parameters are summarised in Table 2.

Table 2. Sources and setting methods for soil organic matter and Exogenous Organic Matter (EOM) parameters in NCSOIL.

NCSOIL pools	Pool 0[a]	Pool I	Pool II	Pool III	Residues[b]	EOM
Initial Size (mg C.kg^{-1} soil)	0	2.6%[f] of TOC[c]	Fitted (see fig 2.)	TOC-C_{PII}-C_{PI}	Calculated for each crop in CERES-EGC	Organic C from EOM[k]
-Labile fraction (% total C of pool)	56%	56%[f]	Till[d,h]	-	20%[j]	100-I_{ROC}
-Resistant fraction (% total C of pool)	44%	44%[f]	100%-Till	-	70%[j]	I_{ROC}
C decay rate constants (k in day^{-1})				0.000055[f]		
-Labile	0.332	0.332[f;g]	0.16[g]	-	0.8[g]	From NLR
-Resistant	0.0404	0.0404[f;g]	0.002 (see in text)[e]	-	0.05[i]	
Fraction of decomposed C recycled into pool I (see fig. 1) (EFFAC)				0.25[g]		
-Labile	0.6	0.6[f,g]	0.4[g]	-	0.6[g]	0.6[a]
-Resistant	0.6	0.6[f,g]	0.4[g]	-	0.4[g]	0.6[a]
Fraction of decomposed C incorporated into the following mineralisable soil pool (see fig. 1) (EFSINK)	0.2	0.2[f;g]	0.2[f;g]	-	-	-
C/N ratio	6.0	6.0[f;g]	Optimised (see fig 2.)	CN_{PIII}= C_{PIII} / [(TOC/CN_{TOC})-(C_{PI}/CN_{PI})-(C_{PII}/CN_{PII})]	Calculated in CERES-EGC	CN_{EOM}

[a]Pool 0 parameters and EOM EFFACs taken as for Pool I.
[b]For crop residues, the labile fraction corresponds to carbohydrate and the resistant fraction to cellulose. The lignin fraction of the residues is directly incorporated into the Pool II.
[c]TOC, C_{PI}, C_{PII}, C_{PIII} stand for Total Organic Carbon, pool I, pool II, pool III contents (mg C kg^{-1} soil), CN_{TOC}, CN_{PI}, CN_{PII}, CN_{PIII} their associated organic C/N ratio.
[d]Till set to 0 in field condition and to 16% for lab simulations.
[e]C decay rate constant of resistant fraction of pool II taken as lignin in Corbeels et al. (1999).
[f]Nicolardot et al. (1994a), [g]Corbeels et al. (1999), [h]Molina et al. (1980), [i]Hadas et al. (2004), [j]Godwin and Jones. (1991).
[k]Organic C from EOM (mg C kg^{-1} soil) = organic C content in EOM (C_{EOM}; mg C g^{-1} DM)*mass of EOM (g DM)/ mass of soil (kg DM).

3.4. Model parameterisation

A schematic overview of our parameterisation methodology is given in Fig. 2. The overall parameterisation strategy consisted in using the CERES-EGC model for performing simulations at the field scale, combining a set of measured parameters (mainly referring to soil physico-chemical properties) and a set of parameters (mainly referring to the mineralisation of EOM) extrapolated from laboratory incubations using the NCSOIL model.

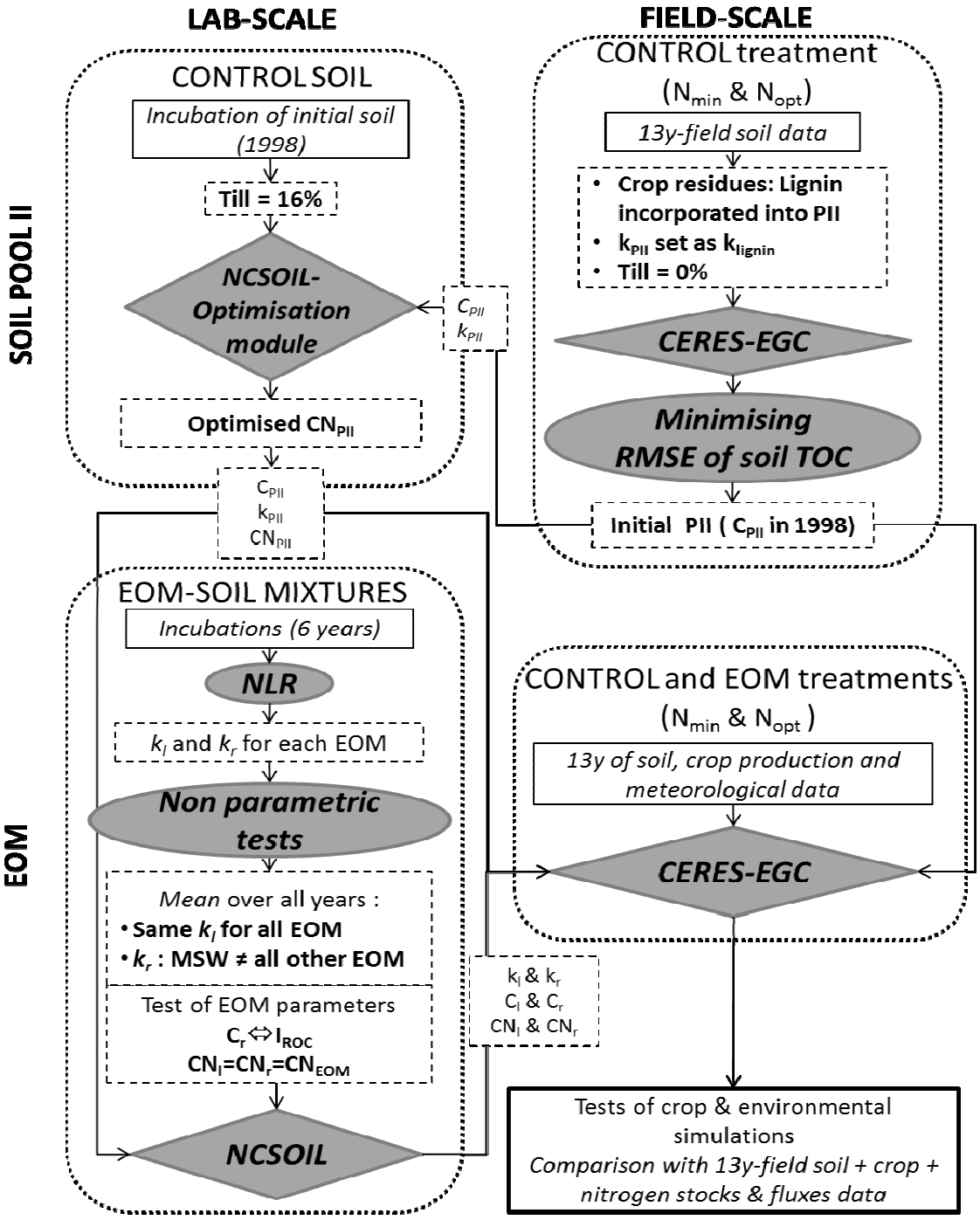


Figure 2. Schematic overview of the parameterisation methodology

NLR stands for Non-Linear-Regressions. N_{min} and N_{opt} correspond to the minimal and optimal level of mineral fertilisation in the field experiment. PII stands for soil pool II, r and l for resistant and labile EOM pool. C, CN and k, the size, C/N ratio and decomposition rate of these pools, TOC the soil Total Organic Carbon content and CN_{TOC} its associated C/N ratio. Till was set to 0 for field-scale and to 16% for lab-scale simulations.

3.4.1. CERES-EGC parameterisation for the QualiAgro experiment

Simulations were performed on the 1998-2011 period. Climatic data came from the Grignon (1998-2004) then Feucherolles (2005-2011) nearby meteorological stations and included daily minimum and maximum temperature, rainfall and potential evapotranspiration (ETP Penman).

Soil physico-chemical parameters relied on the description of 5 pits evenly spread over the QualiAgro experiment: the soil profile was divided into 6 horizons described with their thickness, bulk density, pH (AFNOR, 2005), carbonate contents (AFNOR, 2014) and granulometry (AFNOR, 2003) (Table 1S, from Chalhoub et al., 2013). A pedotransfer function was then used to calculate saturated hydraulic conductivity (Gabrielle et al., 2002). Soil water content at wilting point (*LL*), field capacity (*CC*) and saturation was determined by laboratory measurements of soil retention curves performed on pressure plate apparatuses: *LL* and *CC* were taken at pF -1513 and -33.9 kPa respectively. The soil characteristics were identical in all treatments except for initial SOC and mineral N contents. The initial measured nitrate and ammonium contents in all plots for 3 depths (0-30, 30-60 and 60-90 cm) were averaged for each treatment (4 plots).

3.4.2. NCSOIL parameterisation

Initial total SOC content and C/N ratio (TOC and $CN_{TOC,}$ respectively) were measured in soil sampled in the ploughed layer in 1998 in all 40 plots then averaged for each treatment.

C_{EOM} and CN_{EOM} were measured in each EOM for each application. Both SOC and EOM parameterisation were completed using parameters from Nicolardot et al. (1994), Corbeels et al. (1999) and Hadas et al., (2004) (Table 2). Other parameters including the sizes, C/N ratio and decomposition rates of soil pool II; EOM labile and resistant pools were adjusted. The till parameter was activated and set to 16% only during the lab-scale incubations to take into consideration soil sieving before incubations (Molina et al., 1983).

Initial characteristics of pool II

First, the initial pool II size was determined in order to fit the CERES-EGC simulation of TOC content evolution to the TOC measurements in the 2 control mineral N fertiliser treatments based on minimisation of the Root Mean Square Error (RMSE, see §2.4.3) calculated from their 8 control plots over 13 years: (i) RMSE calculated for the evolution of TOC during the 1998-2013 period; (ii) RMSE calculated for the final TOC of the considered period i.e. the year 2011.

Then, the C/N ratio of soil pool II was determined using an optimisation module added to NCSOIL by Barak et al (1990) along with lab C and N mineralisation measurements of the initial soil sampled in 1998. The optimisation module adjusts unknown parameters, by fitting simulations against measurements of CO_2 evolution and mineral N accumulation, using the Marquardt algorithm which minimises a merit function χ^2 (see §2.4.3).

EOM parameters

The kinetics of C and N mineralisation during incubation of soil-EOM mixtures corresponding to 6 sets of EOM application were used to adjust EOM decomposition rates using Non-Linear-Regression (NLR) as described in Gabrielle et al. (2004). For NLR adjustments, a two-compartment, exponential C-decay model was fitted against data of CO_2 release from the incubated soil-EOM mixtures, after subtracting the contribution of endogenous soil OM, assuming that the mineralisation of endogenous SOM did not interfere with the application of EOM (Mamo et al. 1999; Gabrielle et al. 2004). This model can be written as:

$$C_{release} = C_l\left[1-\exp(-k_l t)\right] + C_r k_r t \qquad [3]$$

Where $C_{release}$ is in mg C-CO_2 kg^{-1} dry soil; t is time (d); C_l and C_r are the sizes of the labile and resistant fractions, respectively (mg C added kg^{-1} dry soil) with C_l+C_r=organic C from EOM; k_l and k_r the decomposition rates of the labile and resistant fractions, respectively (d^{-1}).

The EOM decomposition rates k_l and k_r were multiplied by a 1.5 factor as NLR does not take into account microbial recycling (Gabrielle et al., 2004). A Kruskal-Wallis test was performed to study differences between k_r and k_l values obtained. In order to simplify future EOM parameterisation, the proportion of more resistant organic C was set using the I_{ROC} indicator previously calculated for each EOM applied. For the same purpose, labile and resistant C/N ratios were assumed to be equal to the total CN_{EOM} measured for each EOM and for each year. Indeed, we first tried to optimise the C/N ratios of both labile and resistant fractions; however it did not improve the N mineralisation simulations.

3.4.3. Evaluation of model goodness of fitting

These SOM and EOM parameters were finally tested through NCSOIL simulations of C and N mineralisation measured during lab incubations, and CERES-EGC simulations against 13-y of field measurements of crop yields and N exports, soil TOC, soil mineral N contents and nitrate leaching. NCSOIL and CERES-EGC simulations were compared to observed data both by visual appraisal from graphs and using statistical indicators.

The Root Mean Square Error (RMSE) was calculated to quantify the scatter between observed and predicted data.

$$RMSE = \{E[(S_i - O_i)^2]\}^{1/2} \qquad [4]$$

Where E is the expectancy, S_i and O_i are the simulated and observed values for the ith observation.

The RMSE is often given as a % of the mean measured values and can be also called Coefficient of Variation (CV)

$$CV = \frac{\{E[(S_i - O_i)^2]\}^{\frac{1}{2}}}{E(O_i)} \qquad [5]$$

Additionally, for NCSOIL, EOM parameterisation was tested through comparison of measured and simulated C and N mineralisation kinetics during soil-EOM mixtures incubations. To evaluate the goodness of simulation fits to measurement the weighted sum of squares of residuals was calculated represented by the chi-square value χ^2:

$$\chi^2 = \frac{\sum_{j=1}^{J}\sum_{n=1}^{N}\left[y_{jn} - \frac{y_j(n,a)}{\sigma_j}\right]^2}{(NJ-K)} \qquad [6]$$

$$\sigma_j = \left[\left(\sum_{n=1}^{N} y_{jn}{}^2 - \frac{\left(\sum_{n=1}^{N} y_{jn}\right)^2}{N}\right)/(N-1)\right]^{1/2} \qquad [7]$$

Where j, the index for the number of observed variables used for the optimisation of unknown parameters j=1, 2, 3…J (J= 2 for mineralised C and N); n, the index for the number of simulation intervals per treatment n=1, 2, 3,…N (N=7 for seven days); K; the number of unknown parameters a and σ_j the standard deviation of y_j measurements.

4. Results

4.1. Initial parameterisation of Pool II

4.1.1. Estimation of initial Pool II size with CERES-EGC

Based on the TOC evolution in control soils throughout the entire simulated period, the initial proportions of pool II which resulted in the lowest RMSE values between measured and predicted TOC with CERES-EGC, corresponded to 24 and 32% of the initial TOC in 1998 for N_{opt} and N_{min} treatments, respectively (Table 2S). Setting the initial proportion of pool II to 28% was the best compromise to better fit TOC evolution for the two control treatments simultaneously.

4.1.2. Estimation of Pool II C/N ratio with NCSOIL

The optimisations performed well as evidenced by the low χ^2 (0.0719; table 3S). Despite overall good agreements between simulations and measurements, C mineralisation simulations showed an early overestimation but final underestimation, revealing the difficulty to correctly represent the C dynamics over all the incubation period with these sets of parameters (Table 3S). The optimised CN_{PII} was 10.4 which was quite similar to CN_{TOC} of 9.2.

4.2. EOM parameterisation based on lab-scale results and simulations with NCSOIL

4.2.1. Results obtained from the NLR procedure

The Fig. 3 shows the mean decomposition rates retrieved from the Non-Linear-Regressions (NLR) procedure for labile and resistant EOM fractions obtained for the 6 sets of incubations corresponding to the EOM application made in 1998, 2002, 2004, 2006, 2007 and 2009. The results of EOM incubation in 2000 were not used since the incubations only lasted 70 days. The Kruskal-Wallis test showed no significant differences between k_l values, and it only showed significant differences for k_r values between MSW and GWS, FYM, and BIO respectively (p-values equal to 0.04, 0.02, 0.06). Significant differences also existed in k_r values between MSW and the group of all other EOM (p-value equal to 0.0011).

Based on these results and to simplify the parameterisation, we then assumed that the k_l could be the same for all EOM and it was set to the average value of all k_l results. Similarly, k_r was considered as identical for GWS-BIO-FYM but different for MSW characterised by a higher k_r.

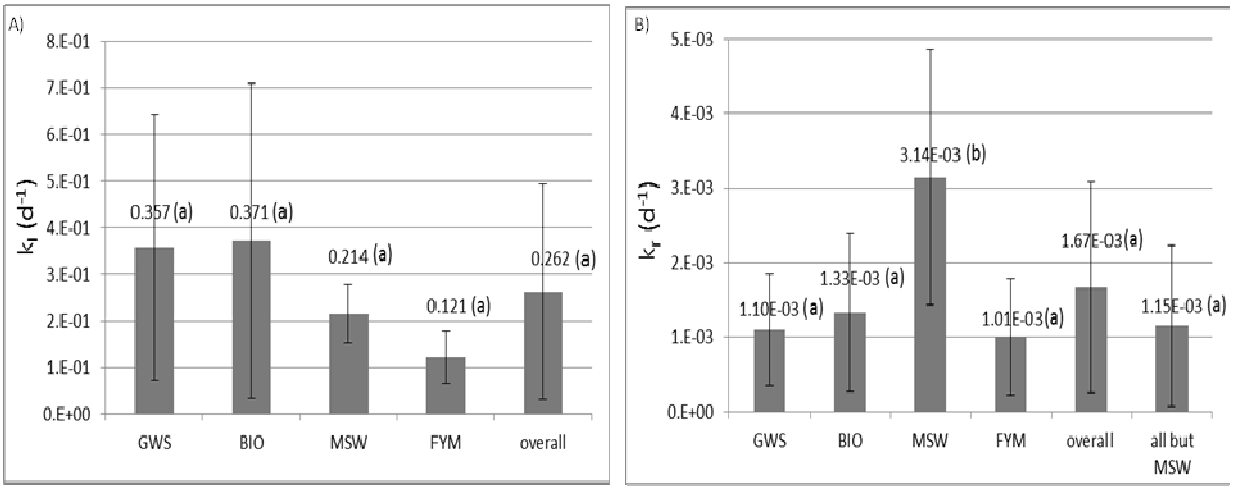


Figure. 3. Non-Linear-Regression (NLR) results. Mean of EOM decomposition rates, labile (A) and resistant (B). The NLR was fitted on C mineralisation measurements of 1998, 2002, 2004, 2006, 2007 and 2009. Results for GWS-2004 were not included because of procedure failure.

4.2.2. Variation of C and N mineralisation with the EOM origin

The average measured (dots) and NCSOIL-simulated (lines) kinetics of C and N mineralisation for all incubation years and each EOM are presented in Fig. 4. The results of C and N mineralisation are summarised in Table 3.

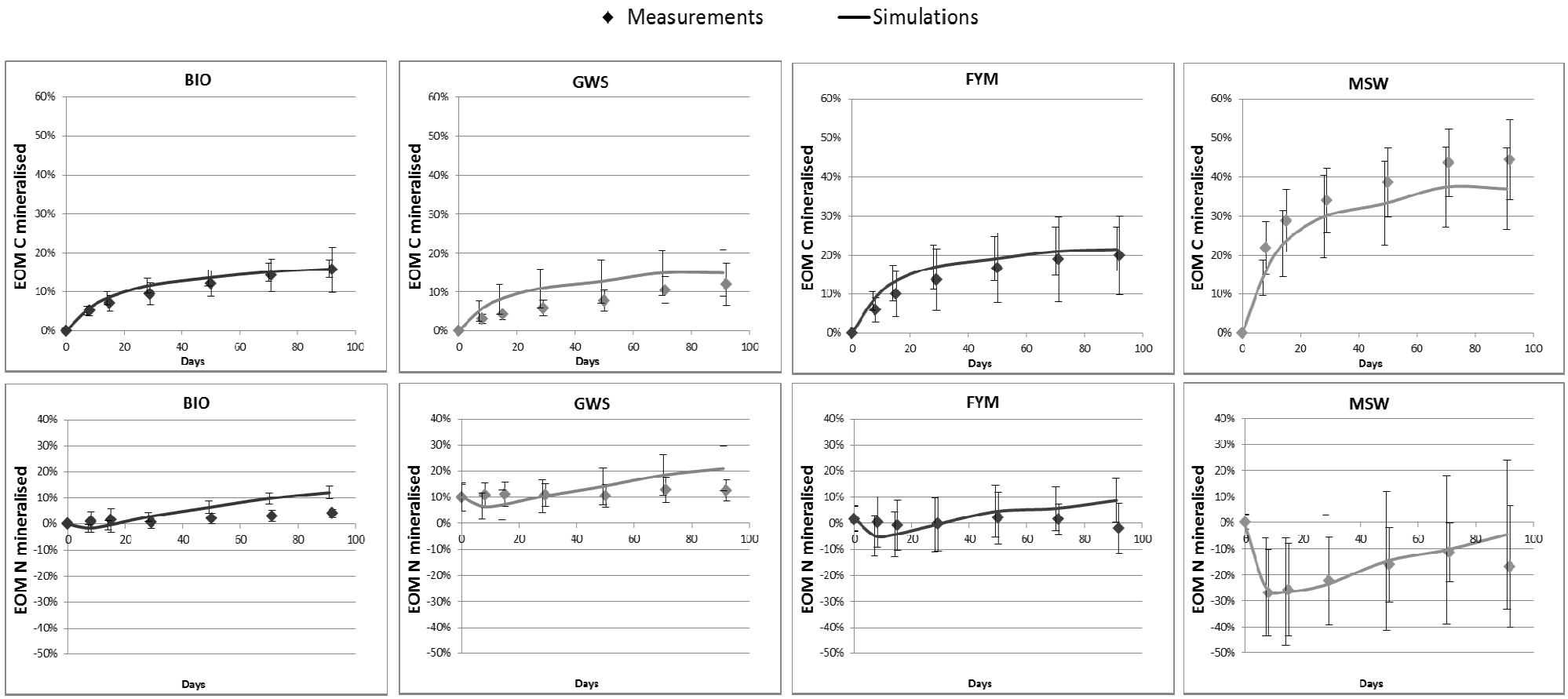


Figure 4. NCSOIL simulations (lines) against incubation measurements (dots) of CO_2-C release and mineral N content during soil-EOM incubations under laboratory controlled conditions (average of 6 sets of incubations corresponding to the EOM applied in 1998, 2002, 2004, 2006, 2007 and 2009). The results are expressed in percentage of organic C and N from EOM.

The proportion of mineralised organic C from EOM at the end of the incubations increased in the order GWS (12.7%) < BIO (16.2%) < FYM (20.9%) < MSW (45.3%) which varied inversely to their respective OM stability as shown by their I_{ROC}.

Referring to N mineralisation, the percentages of EOM organic N mineralised increased in the order: MSW (-13.2%) < FYM (-0.2%) < BIO (3.0%) < GWS (4.0%) which corresponded to increasing stability of EOM organic matter as revealed by I_{ROC} and almost to decreasing EOM C/N ratios: MSW (19.3) > FYM (14.6) > GWS (13.0) > BIO (12.0), but BIO and GWS had similar OM characteristics. Indeed, net N mineralisation depends on both C biodegradability and EOM C/N ratio. BIO and GWS did not show patterns of N immobilisation but FYM showed slight early immobilisation, while it was stronger for MSW. The amplitude of the early immobilisation was directly related to C/N ratios and inversely related to OM stability.

Table 3. Final Exogenous Organic Matter (EOM) parameters in NCSOIL and evaluation of the simulated compared to measured results.

	BIO[a]		GWS		FYM		MSW	
	Mean[b]	Sd	mean	Sd	mean	Sd	mean	Sd
k_r (day^{-1})	0.00115		0.00115		0.00115		0.00314	
k_l (day^{-1})	0.262		0.262		0.262		0.262	
I_{ROC}-derived *Cr* (% total C_{EOM}[c])	74.5	3.1	75.8	6.8	67.4	8.5	45.2	14.3
Both pools C/N ratio	12	1.2	13	3.3	14.6	2.3	19.3	4.7
Chi-square value χ^2	0.31	0.12	0.65	0.61	0.43	0.43	0.34	0.16
C mineralisation (% organic C from EOM[d])								
Final (day 91) measured C mineralised	16.2%	6.1%	12.7%	4.8%	20.9%	11.2%	45.3%	9.0%
Final (day 91) simulated C mineralised	16.9%	1.1%	15.8%	6.2%	22.5%	6.7%	38.8%	8.6%
Absolute bias of simulation on final day	0.6%	6.0%	3.0%	7.6%	1.6%	4.8%	-6.5%	6.0%
Absolute bias after inflexion (day 45)	2.1%	3.9%	5.1%	5.4%	3.3%	2.7%	-4.1%	4.7%
RMSE (% organic C from EOM)	3.7%	1.6%	5.7%	2.7%	3.5%	1.4%	6.5%	2.7%
N mineralisation (% organic N from EOM[e])								
Final (day 91) measured N mineralised	4.0%	1.9%	3.0%	5.1%	-0.2%	10.2%	-13.2%	18.0%
Final (day 91) simulated N mineralised	13.7%	2.1%	11.5%	5.3%	10.3%	4.6%	1.7%	25.0%
Absolute bias of simulation on final day	9.7%	3.5%	8.5%	6.8%	10.5%	11.1%	14.9%	12.7%
RMSE (% organic N from EOM)	6.0%	2.3%	5.7%	3.7%	6.2%	3.7%	11.4%	2.6%

[a]BIO: bio-waste compost, GWS: green waste and sewage sludge co-compost, FYM: dairy farmyard manure, MSW: municipal solid waste compost.
[b]Mean for years 1998-2002-2004-2006-2007-2009; Sd: Standard deviation.
[c]C_{EOM} and N_{EOM}: Exogenous Organic Carbon and Nitrogen contents.
[d]Organic C from EOM (mg C kg^{-1} soil) = C_{EOM} (mg C g^{-1} DM)*mass of EOM (g DM)/ mass of soil (kg DM).
[e]Organic C from EOM (mg C kg^{-1} soil) = N_{EOM} (mg N g^{-1} DM)*mass of EOM (g DM)/ mass of soil (kg DM).

4.2.3. Goodness of fitting of simulated EOM kinetics of mineralisation

Simulations globally well fitted to measurements as demonstrated by low average χ^2 values (Table 3). For C mineralisation, simulations preserved the same sorting as measurements in terms of organic C from EOM mineralised. On average, simulations slightly overestimated the measurements for GWS, FYM and BIO, while they underestimated measurements for MSW. This may be due to inappropriate EOM pool decomposition rate or pool C size parameterisation. For BIO, GWS and FYM, the overestimation of C mineralisation after the inflexion at day 45 (2.1%, 5.1% and 3.3%, respectively) were higher than the final bias (0.6%; 3.0% and 1.6%) suggesting the overestimation of the final mineralisation came mostly from an overestimation of C_l, thus an underestimation of C_r by I_{ROC}. On the contrary, for MSW, bias after inflexion was already negative (-4.1%) and increased for final bias (-6.5%) meaning the C_l was underestimated (i.e. C_r overestimated), but k_r as well. Nevertheless, the biases remained low and EOM parameters driving C mineralisation were in overall satisfying.

Simulations of N mineralisation maintained the same EOM ranking than measurements in terms of organic N mineralised except for MSW but globally overestimated the N mineralisation. Concerning MSW, N mineralisation showed high variability both in simulations and measurements, probably explaining this sorting discrepancy.

Simulations showed slight early immobilisation for BIO, GWS and FYM, which were not observed in measurements for BIO, GWS, and were lower for FYM. Underestimation of early N mineralisation was largely offset in the second period of mineralisation causing final overestimation of N mineralisation for all EOM. These biases could be explained both by C mineralisation biases and C/N inappropriate parameterisation. Overestimation of the I_{ROC}-derived labile fraction size may have resulted in overestimating C degradation and caused both early immobilisation and global N mineralisation overestimation. Using distinct C/N ratios for the two EOM pools, we actually observed that it could also improve the simulation of N mineralisation kinetics with a lower EOM labile pool C/N ratio and thus a higher resistant pool C/N ratio (not shown). However, we did not retain such parameterisation as it meant that these ratio values had to be optimised and were not actually accessible.

4.3. Field-scale CERES-EGC simulations over a 13 y-time series

4.3.1. Soil organic matter

The evolution of SOC contents are presented in Fig. 5 and Table 4. The TOC slightly decreased in control treatments with an annual rate of 0.067 and 0.017 g C kg^{-1} yr^{-1} for N_{opt} and N_{min} respectively whereas it increased for all EOM treatments (annual increase rate comprised between 0.146 g C kg^{-1} for MSW N_{min} and 0.408 g C kg^{-1} for GWS N_{opt}.

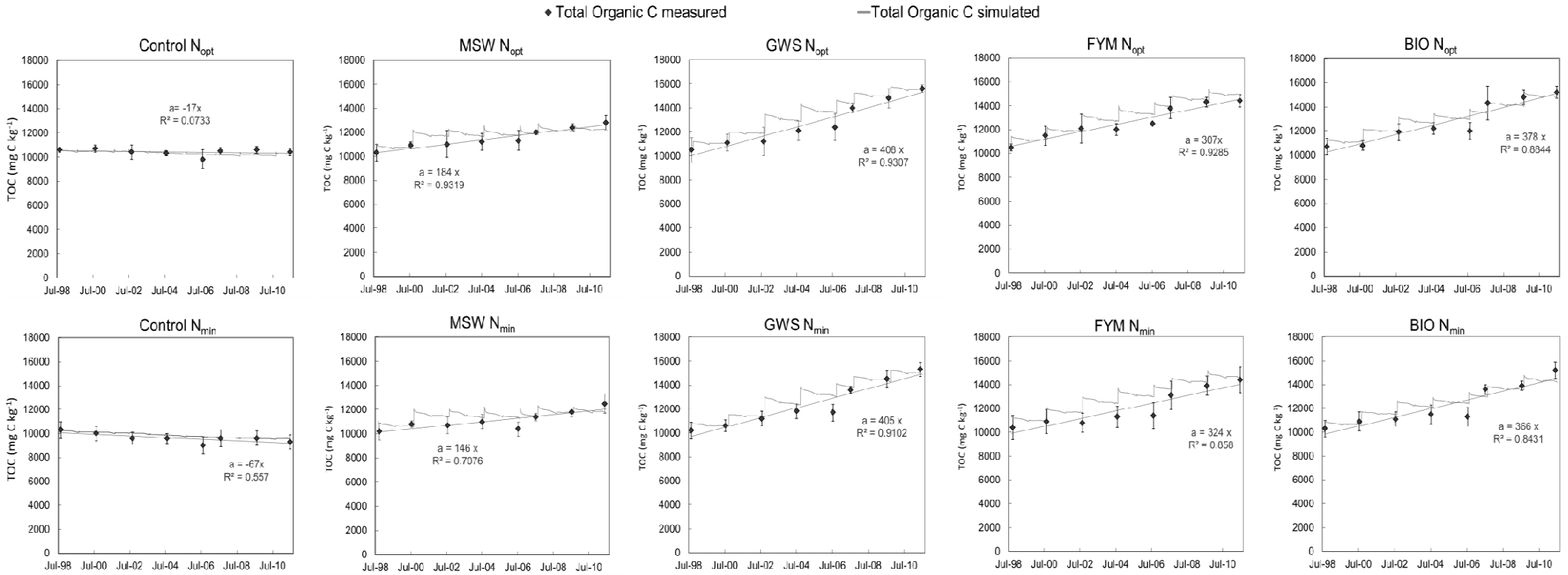


Figure 5. Evolution of Soil Organic Carbon Content (mg C kg-1 soil), simulated (line) and measured (dots) in the different treatments of the Qualiagro field experiment. The rate of C storage i.e. slope of measured trend is expressed in mg C kg-1 yr-1.

The C storage over the 13 years period could be sorted as follows: Control < MSW < FYM < BIO < GWS. The differences between N_{opt} and N_{min} remained very low, except for control treatments which could be explained by substantial differences in crop residues produced and incorporated into the soil. The EOM also differed by their EOC storage efficiency i.e. the % of total EOC applied stored at the end of the period considered: for N_{opt}, they increased according to the order MSW (39%) < FYM (50%) < BIO = GWS (65%), in accordance with increasing OM stability as expressed through I_{ROC}-derived *Cr*.

Simulations of TOC contents fitted to measurements well (Table 4 and Fig. 5), with low CV values (between 1% and 7%). Simulations performed well in reproducing TOC evolution for the different treatments (Fig. 5). The final simulated TOC were slightly underestimated in the GWS, BIO and MSW treatments, suggesting a too fast decomposition of the EOM, and the opposite for FYM but the final biases were always lower than the Standard Error (SE) of 2011 TOC measurements except for BIO $N_{min.}$ This was consistent with lab-scale study with NCSOIL for BIO, GWS for which C mineralisation was overestimated. However, the opposite was expected for MSW and FYM for which NCSOIL underestimated and overestimated lab C mineralisation measurements respectively.

For both N_{opt} and N_{min} treatments, the final simulated TOC content (Table 4) increased following the order: Control < MSW < BIO < FYM < GWS which slightly differed from the observed data: Control <MSW < FYM < BIO < GWS. However, measured TOC differences between FYM and BIO were in the same order of magnitude as the SE of final TOC.

Table 4. CERES-EGC simulation results of soil C contents, soil N stocks and leachings, crop yields and N exports.

	N_{min}[a]										N_{opt}									
	Control		**BIO**[b]		**GWS**		**FYM**		**MSW**		**Control**		**BIO**		**GWS**		**FYM**		**MSW**	
Soil TOC																				
Final soil TOC measured (mg C kg^{-1} soil)	9300*	*600*	15200	*700*	15300	*600*	14400	*1100*	12500	*800*	10400	*300*	15200	*500*	15600	*300*	14400	*500*	12800	*600*
Measured storage (mg C kg^{-1}soil)	-1000		4900		5100		4000		2300		-200		4500		5100		3900		2500	
Simulated storage (mg C kg^{-1}soil)	-741		3981		4735		4130		1581		-438		4058		4925		4377		1910	
Final TOC bias (% final value measured)	2.8%		-6.0%		-2.4%		0.9%		-5.8%		-2.3%		-2.9%		-1.1%		3.3%		-4.6%	
Global CV (% mean measurements)	4%		5%		5%		7%		5%		1%		5%		5%		4%		4%	
EOC storage efficiency (% EOC applied)			71%		65%		52%		36%				65%		65%		50%		39%	
Overall crop yields[c]																				
Measurements average (Mg ha^{-1})	5.61	*0.55*	6.77	*0.31*	7.42	*0.38*	7.22	*0.31*	7.29	*0.33*	7.98	*0.41*	8.42	*0.38*	8.47	*0.38*	8.72	*0.39*	8.56	*0.4*
Simulations average (Mg ha^{-1})	5.12	*1.51*	7.16	*2.2*	7.24	*2.14*	6.98	*2.22*	7.51	2.2	8.12	*0.97*	8.81	*1.21*	8.86	*1.18*	8.68	*1.17*	8.77	*1.25*
Global CV (% mean)	27.0%		14.2%		9.9%		14.2%		12.2%		9.3%		9.1%		9.4%		7.3%		9.1%	
Total N exports																				
Measurements (kg N ha^{-1})	77	*10*	119	*9*	137	*13*	129	*11*	132	*11*	148	*14*	162	*11*	175	*14*	151	*14*	173	*13*
Simulations (kg N ha^{-1})	78		126		135		125		139		137		170		172		167		170	
Mean bias (% of measurement)	1%		6%		-2%		-3%		5%		-7%		5%		-2%		11%		-2%	
Global CV (% mean measurements)	35%		25%		20%		23%		21%		17%		17%		17%		25%		16%	
N exports by crop grains																				
Measurements (kg N ha^{-1})	58	*7*	90	*5*	104	*8*	98	*7*	99	*7*	112	*7*	124	*6*	133	*8*	114	*6*	132	*7*
Simulations (kg N ha^{-1})	56		97		103		95		106		109		135		136		132		134	
N exports by crop residues[d]																				
Measurements (kg N ha^{-1})	19	*3*	29	*3*	33	*5*	31	*5*	33	*4*	36	*7*	38	*5*	42	*6*	37	*7*	41	*6*
Simulations (kg N ha^{-1})	22		29		32		30		32		28		35		35		35		36	
Soil mineral N stocks																				
Mean measurements at 30cm (kg N ha^{-1})	14		20		28		24		23		19		23		32		29		26	
Mean measurements at 90cm (kg N ha^{-1})	29		46		76		54		52		42		55		85		67		61	
Global CV 30cm (% mean measurements)	154%		186%		163%		165%		128%		128%		200%		177%		153%		194%	
Global CV 90cm (% mean measurements)	134%		162%		120%		130%		103%		103%		175%		148%		128%		189%	
N leachings: annual fluxes[e]																				
Measurements at 50cm depth (kg N ha^{-1})											35	*38*	38	*39*	54	*49*	37	*8*	50	*45*
Simulations at 50cm depth (kg N ha^{-1})	60	*3*	124	*7*	139	*8*	123	*7*	145	*8*	78	*4*	158	*9*	175	*9*	153	*8*	176	*10*

*Values and *Standard Error* for measurements or *Standard deviation* for simulations.

[a]Nmin and Nopt correspond to the minimal and optimal level of mineral fertilisation in the field experiment.

[b]BIO: bio-waste compost, GWS: green waste and sewage sludge co-compost, FYM: dairy farmyard manure, MSW: municipal solid waste compost.

[c]Measurements and simulations yearly averaged for crop yields, crop N exports and N leachings, averaged over the period 1998-2013 for soil mineral N stocks.

[d]Maize straws were left on field, and wheat straws only were exported from field.

[e]N leachings were measured and simulated over the period 2004-2009.

4.3.2. Crop yield and N uptake

Repeated EOM applications enhanced crop yields compared to control, for both N_{opt} and N_{min} in a higher extent for N_{min} (Table 4). Similarly, crop N uptake was greater in the EOM treatments than in the control especially for N_{min} treatments.

Crop yields were well predicted especially for N_{opt} treatments and CV varied from 7.3 to 9.3% for N_{opt} and from 9.9 to 27.0% for N_{min} (Table 4). The wheat yields were systematically underestimated in the N_{min} treatments in 2000, the only year where wheat did not receive mineral N (Fig. 1S). The control N_{min} showed the largest biases especially for maize (underestimations in 2001, 2003, 2008); these latter biases could be due to an overestimation of plant N stress by CERES-EGC. Good simulations of yields concerned both grains and residues (Fig. 6): simulations plotted against measurements for all crops and all years gave trend with slopes close to 1 (1.12 and 0.999 for grains and residues respectively) with high R^2 (0.79 and 0.82).

Crop N uptake was also quite well estimated. Total N exports biases ranged from -7 to +11% of the measured total N exports which was always lower than SE values except for FYM N_{opt} (Table 4). The CV ranged from 16 to 25% for N_{opt} and from 20 to 35% for N_{min} treatments, revealing slightly better estimations for N_{opt} treatments as for crop yields, emphasising the hypothesis of CERES-EGC overestimating N plant stress. N partitioning between grains and residues were also quite well simulated as shown by the small differences between simulations and measurements. The simulated N uptakes were significantly correlated to measured N uptakes for all crops and all years with slopes of 0.997, 1.011 and 1.077 for grains, residues and total N uptake, respectively and R^2 of 0.61; 0.87; 0.73 (Fig. 6).

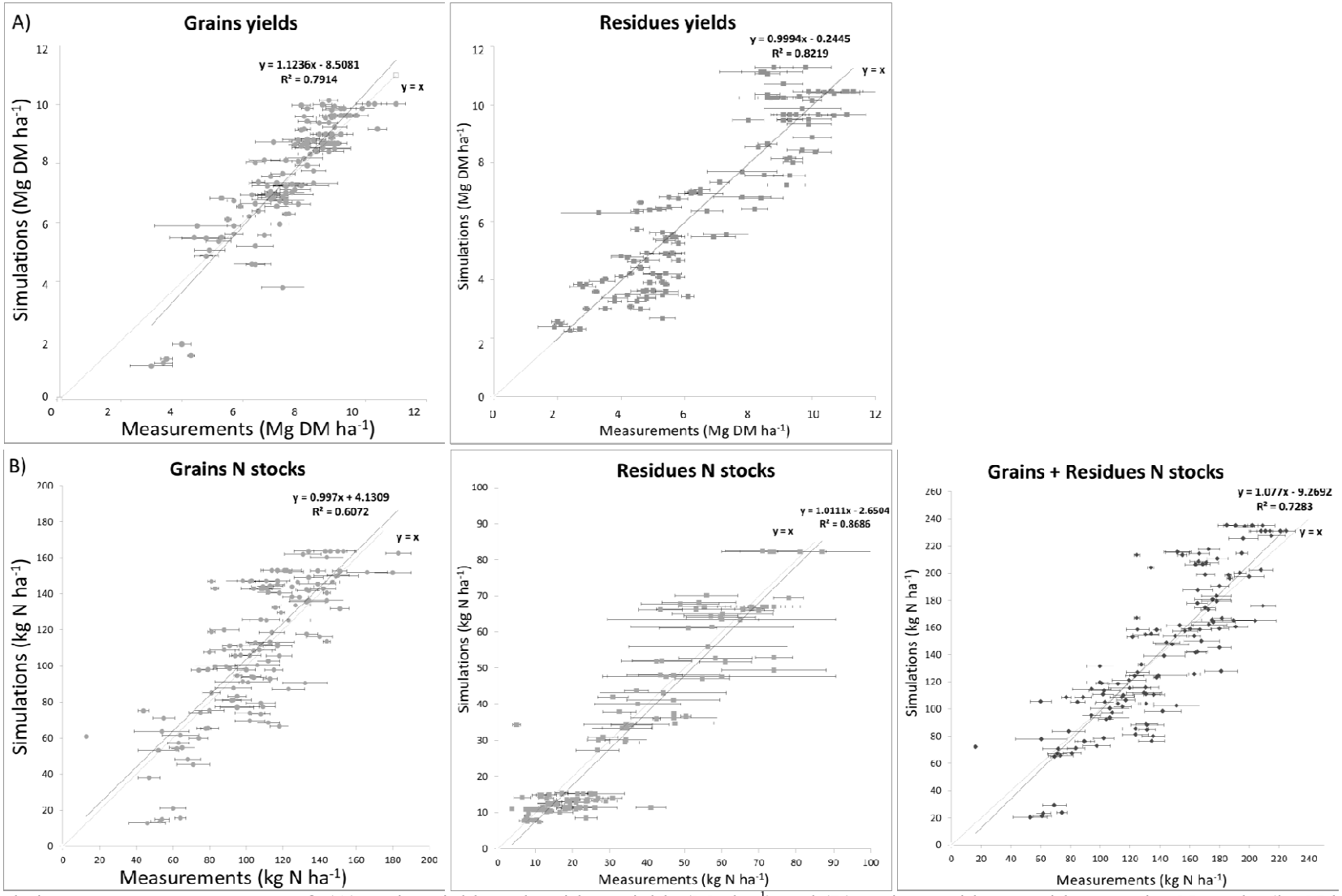


Figure 6. Simulations versus measurements of: (A) grains yields and residues yields (Mg ha^{-1}) and (B) grains, residues, residues+grains N stocks (kg N ha^{-1}). Residues stand for straws+leaves. Average and standard error of the 4 replicate plots are presented for the experimental results

4.3.3. Soil mineral N

The measurements of soil mineral N stocks are presented in Fig. 7. Over the whole period, stocks were on average of 29 and 42 kg N ha^{-1} in the 90 first cm for control N_{min} and N_{opt} respectively (Table 4). With EOM, these stocks were at least increased by 150% with MSW and by more than 200% with GWS. After EOM application, measured mineral N stocks increased until November related to EOM mineral N input and organic N mineralisation, then often decreased until February because of mineral N leaching.

As previously observed by Gabrielle et al. (2005), the model tended to over-estimate topsoil mineral N during the winter and spring periods following compost application, i.e. before and during early maize growing season (Fig. 7). The stocks were better predicted during the wheat growing season and crop N uptake. This could suggest an overestimation of the rates of EOM organic N mineralisation. While soil mineral N at 30-cm depth during the winter following the EOM application sharply decreased, this decrease was only little at 90-cm depth (Fig. 2S). This suggests, that mineralised N from the first 30-cm layer leached to lower horizons but remained stored in the 90-cm and therefore accessible by plant roots.

Both simulated and measured soil mineral N stocks were greater in EOM treatments than in control treatment, and higher from 2004 to 2013, compared to the 1998-2003 period, presumably because of cumulated SOC increase resulting in higher N release from OM mineralisation. However, SOC increased steadily whereas mineral N increased sharply especially in 2005, which could be explained by a drier year in 2005 leading to a drying up of the entire soil profile the soil and a slow water refilling for the deep horizons over the year, resulting in concentrating N in the upper layer.

4.3.4. N leaching

Annual N leaching measurements were on average 35 kg N ha^{-1} for control N_{opt} (Table 4). The N leaching increased in the EOM treatments, slightly for FYM (37 kg N ha^{-1}) and BIO (38 kg N ha^{-1}) and a little more for MSW (50 kg N ha^{-1}) and GWS (54 kg N ha^{-1}) but measurements included large errors of about 100% of the mean.

Simulations of N leaching at 50 cm were generally overestimated, even though biases amplitude was variable. For control N_{opt}, simulations (78 kg N ha^{-1}) more than doubled the measurements and such overestimation was even more important for EOM treatments reaching 3 to 5 times the measured fluxes. However N leaching simulations at 90 cm-depth remained largely smaller as shown by soil mineral N dynamics and total N leaching (at 195cm) seemed to be reasonable as yearly averaged values ranged from 6 kg N ha^{-1} for control N_{min} to 21 kg N ha^{-1} for GWS N_{opt} (Table 5).

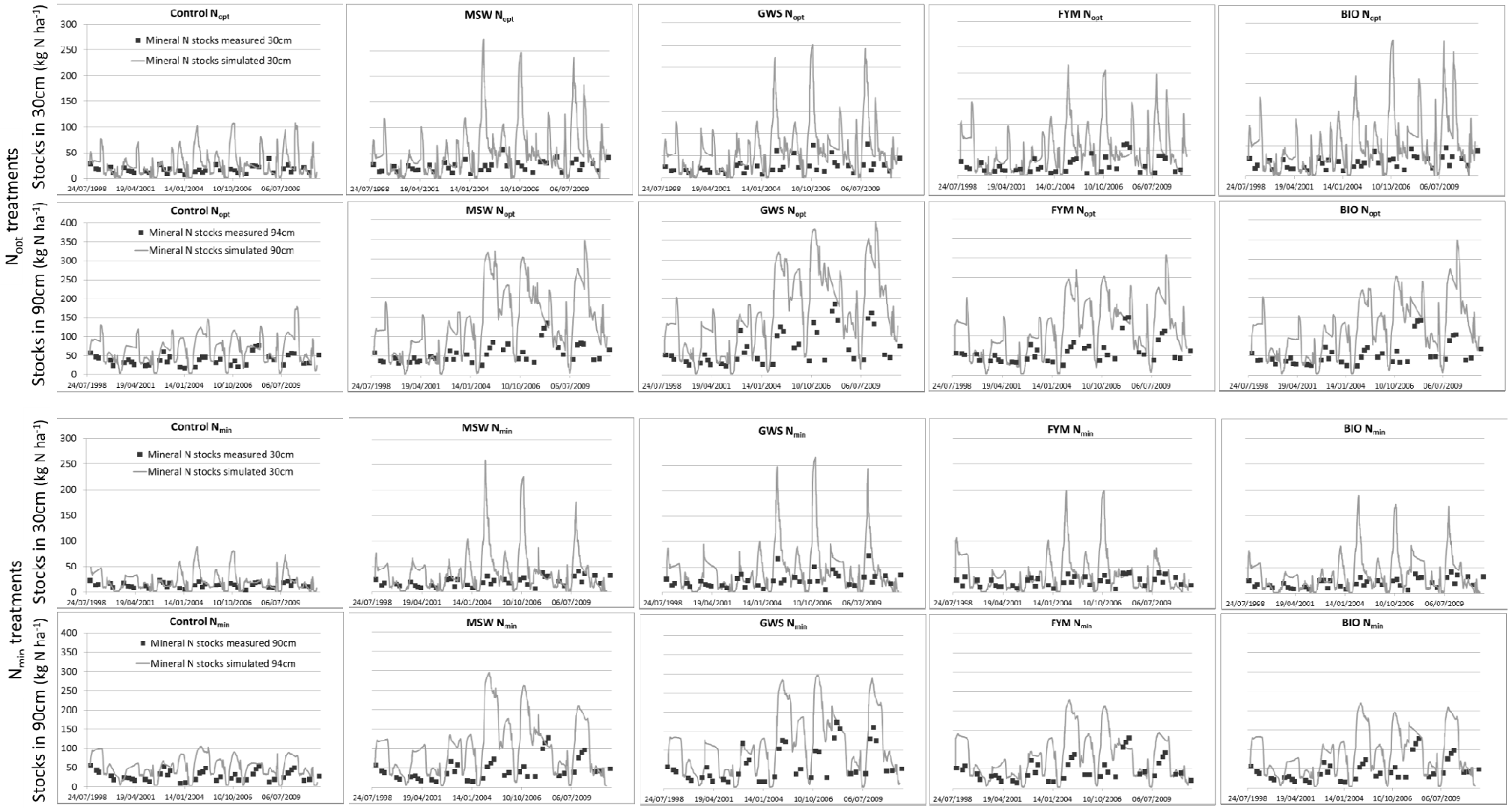


Figure 7. Soil mineral N stocks in 0-90 cm depth (kg N ha^{-1}) measured (dots) and simulated (lines). Stocks are calculated from the measured contents multiplied by soil mass in 3 soil layers (0-30, 30-60 and 60-90 cm) considering measured bulk density in the layers (see Table 1S).

Table 5. CERES-EGC simulation of N fluxes in the trial.

	N_{min}[a]					N_{opt}				
	Control	BIO[b]	GWS	FYM	MSW	Control	BIO	GWS	FYM	MSW
Organic N (kg N ha^{-1} yr^{-1})										
Exogenous Organic N (EON) applied	0[c]	170	181	157	109	0	170	181	157	109
Residues not exported	66	88	94	85	94	95	111	114	108	113
Net mineralisation	84	154	155	148	167	100	169	173	162	171
Organic N balance	-18	104	121	95	35	-5	112	123	104	52
Mineral N (kg N ha^{-1} yr^{-1})										
Mineral N applied	31	31	31	31	31	109	109	109	109	109
EOM mineral N applied	0	10	33	15	16	0	10	33	15	16
Total mineral N applied	31	41	64	46	47	109	119	142	124	125
Wet deposition	20	20	20	20	20	20	20	20	20	20
Net mineralisation	84	154	155	148	167	100	169	173	162	171
Total N mineral inputs	135	215	238	213	234	229	308	334	306	316
Crop N uptake	125	188	200	183	205	206	248	254	243	254
Above ground biomass N at harvest	78	126	135	125	139	137	170	172	167	170
Total N losses	9	19	28	23	19	19	35	44	38	36
Nitrate leaching	6	11	15	12	13	7	16	21	15	20
Ammonia emissions	-2	2	6	5	0	7	11	14	16	8
N_20	2	2	2	2	2	2	3	3	3	3
N_2	3	4	4	4	4	4	5	6	5	5
Total N mineral outputs	134	207	228	206	224	225	283	299	282	289
Mineral N balance	1	8	10	7	10	4	25	36	24	26
Efficiency calculation (%)										
Contribution of EOM to soil organic N[d]		68%	65%	66%	43%		65%	60%	63%	45%
% of mineral N in EOM mineral N inputs[e]		13%	32%	19%	16%		13%	31%	19%	19%
Intensity of EON mineralisation[f]		41%	39%	40%	76%		40%	40%	39%	65%
Apparent N use efficiency[g]		27%	27%	28%	49%	76%				
Potential of N from EOM to be lost[h]		6%	9%	8%	9%		9%	12%	11%	13%

[a]Nmin and Nopt correspond to the minimal and optimal level of mineral fertilisation in the field experiment.
[b]BIO: bio-waste compost, GWS: green waste and sewage sludge co-compost, FYM: dairy farmyard manure, MSW: municipal solid waste compost.
[c]fluxes and N balance were calculated between two harvest dates and yearly averaged for the 13 simulated years.
[d]calculated as soil (organic N balance in EOM treatment – organic N balance in control treatment)/ total N from EOM applied.
[e]calculated as EOM mineral N applied / (EOM mineral N applied + (N mineralised in EOM treatment – N mineralised in control treatment)).
[f](N mineralised in EOM treatment – N mineralised in control treatment)/EON applied.
[g]calculated for control N_{opt} as (Above ground biomass N in control N_{opt} – Above ground biomass N in control N_{min}) / (surplus of mineral N applied in control N_{opt}), and calculated for EOM treatments as (Above ground biomass N in EOM treatment – Above ground biomass N for control N_{min} treatment)/ total N from EOM applied.
[h]calculated as (N losses in EOM treatment – N losses in control treatment) / (total N from EOM applied).

4.3.5. N fluxes and balances

Average yearly N inputs and outputs as well as final simulated N balances (inputs – outputs) for both mineral and organic N are presented in Table 5.

Simulated organic N balance, which could be assimilated to organic N storage in soil, was negative for control treatments with on average a loss of 18 and 5 kg N ha^{-1} yr^{-1} for N_{min} and N_{opt}, respectively. On the other hand, organic N balance was positive for EOM treatments. Ranking of organic N balance for both N_{min} and N_{opt} treatments was the following: Control < MSW < BIO < FYM < GWS. This ranking was explained by both the quantity of exogenous organic N (EON) applied and the stability of their OM (I_{ROC}). Regarding the contribution of total N from EOM to the soil organic N balance, similar proportions were found for GWS, BIO and FYM (positive N storage equivalent to 60 to 68% of total N from EOM in the N_{opt} and N_{min} treatments), MSW contribution being lower (43-45%). The N_{opt} treatments resulted in larger organic N storage, because of more residues produced, thus more N returned to soil after harvest. Higher degradability of MSW may explain that N input from net N mineralisation in the MSW treatments were quite similar to GWS and BIO, despite differences of EON quantities applied. Indeed for MSW, organic N mineralised reached 76% and 65 for N_{min} and N_{opt}, respectively, where it ranged between 39 and 41% for the other EOM treatments.

Mineral N balances were positive in all treatments meaning an increase of soil mineral N stocks, in a lower extent for control: increase of 1 and 4 kg N ha^{-1} yr^{-1} for N_{min} and N_{opt}, respectively and in a greater extent for EOM especially for N_{opt} treatments: increase of 7 and 24 kg N ha^{-1} yr^{-1} for FYM and up to 10 and 36 kg N ha^{-1} yr^{-1} for GWS for N_{min} and N_{opt}, respectively in the whole soil profile (0-195cm). The mineral N balance was larger in the GWS treatments probably related to the larger mineral N inputs, i.e. the sum of mineral N applied and net N mineralised. The treatments ranked differently in terms of crop N uptake which was larger in all organic treatments compared to controls. The crop N uptakes were similar in GWS and MSW treatments and larger than in the BIO and FYM treatments. The nitrate leaching was also larger in the organic treatments compared to controls with larger leaching simulated in the GWS and MSW treatments.

The apparent N use efficiency was calculated as the differences of total N in above ground biomass in a treatment and control N_{min} divided by the surplus of N (organic+mineral) applied in each treatment compared to control N_{min}. The average apparent N use efficiency was 76% for N_{opt}, quite reasonable for mineral N, where it was rather high for the EOM and higher for MSW (49%) than for the other EOM (from 27 to 28%). This was related to the intensity of EON mineralisation, i.e. the surplus of net mineralisation divided by the quantity of EON applied. Simulated N losses were mainly due to nitrate leaching, ammonia emissions, and in a lower extent to denitrification. In the N_{min} treatments, EOM application only increased total N losses from 11 kg N ha^{-1} yr^{-1} in the BIO treatment to 20 kg N ha^{-1} yr^{-1} in the GWS treatment which represented 6 to 9% of the total N applied from EOM. This potential of N losses from EOM was calculated as the additional N losses compared to control divided by total N from EOM. It was larger for GWS and MSW than for BIO and FYM. This sorting should be first

related to the percentage of EOM mineral N in total mineral N inputs from EOM, for which BIO was the lowest and GWS the highest, and then to intensity of EON mineralisation which increased the N mineral input for MSW and resulted in more loss than for FYM.

5. Discussion

5.1. Relevancy of NCSOIL parameterisation methods

The EOM mineralisation was simulated and EOM dynamic parameters adjusted through comparison with endogenous soil OM mineralisation in a control treatment. Thus the goodness of fitting of the control treatment somehow guaranteed that control soil parameterisation did not cause too much bias in soil-EOM mixtures mineralisation simulations.

NCSOIL simulations based on I_{ROC}-derived resistant pool size sometimes resulted in slight biases in the laboratory incubations but biases were very low in field conditions. Thus we could validate in our field experiment the use of I_{ROC} to parameterise the resistant fraction of EOM. Nevertheless, in the present study, only quite stable EOM were used. The use of I_{ROC} remained to be validated for other more biodegradable EOM. On the other hand, efforts remained to be made to determine generic decomposition rates for a larger range of EOM types. We also investigated a simple parameterisation method for EOM C/N ratio pools (all equal to measured CN_{EOM}). NCSOIL simulations of lab N mineralisation showed only little biases but we can wonder whether they were due to C mineralisation biases or to inappropriateness of this C/N ratio parameterisation. This method seemed in overall quite reasonable. Again, the EOM were rather similar and homogenous in this study. This assumption on C/N ratio may not be relevant for less stable and more heterogeneous EOM.

5.2. Scale-transfer between lab NCSOIL parameterisation and CERES-EGC long-term field simulations

Slight underestimation of soil C mineralisation during short term laboratory incubation with NCSOIL was essentially due to the pool II resistant decomposition rate (k_{pII}) set to 0.002 day^{-1} which was three times lower than the 0.006 day^{-1} usually used (Molina et al., 1983; Nicolardot et al., 1994; Corbeels et al., 1999; Burger and Venterea, 2008). All these studies only worked with short term lab incubations data (3-6 months) and parameterisations were, to the best of our knowledge, not used for long term field data. We indeed tested the k_{pII} value of 0.006 day^{-1} and verified that whereas it improved the fitting of lab measurements, it resulted in a pool II size fall over the 13 y-time series of CERES-EGC field simulations. Gabrielle et al. (2004) and Menasseri (1994) already chose a lower k_{pII} value (i.e. 0.003 day^{-1}). As our main objective was to succeed in correctly simulating long-term C and N dynamics all together with crop yields in field conditions, we focused the first CERES-EGC parameterisation stage on control soil. For this purpose, we set the k_{pII} to this lower value of 0.002 day^{-}

[1] inferred from lignin decomposition rate (Corbeels et al., 1999) and incorporated lignin inputs from residues directly into the pool II. It permitted to maintain pool II size over the 13 years for control soil N_{opt} which was considered as an equilibrium as C stocks and yields remained constant. Lignin is often treated separately from the other residues components in SOC dynamics models. In Century v4.5® (Stolpe, 2012), residues are divided into 2 pools (metabolic and structural) depending on the lignin/N ratio, and the lignin fraction of the structural pool goes to the slow soil pool while the rest goes to the active soil pool (less stable) after decomposition. In the CANTIS model (Garnier et al., 2003), part of the lignin decomposed goes into a more stable pool of humified OM while residues are directed to a soluble OM pool. Moreover, decreasing the rate of decomposition of active OM pool was in accordance with parameterisation of active OM in other models that were always lower: 0.000055 d^{-1} for Roth-C "humus" pool (Jenkinson and Coleman, 2008), 0.0006 d^{-1} for humified OM in CANTIS (Garnier et al., 2003), or 0.000043 d^{-1} for active OM pool in DAISY (Bruun et al., 2003).

The pool II C/N ratio optimised thanks to laboratory measurements was 10.4 for a CN_{TOC} of 9.2. Along with a relatively high pool II size (28% of TOC), this suggested a potential high N supply from SOM compared to previous studies. Indeed, for clayey soils, Corbeels et al. (1999) found a CN_{PII} of 16.7 along with a size of 13% of TOC, and Antil et al. (2011) found a CN_{PII} of 13.8 along with a 10% size. This may partly explain the overestimation of mineral N stocks within the 30 cm and the 90 cm of soil and N leaching at 50 cm depth: inferring more OM and especially N in the pool III would result to less N mineralisation and less leaching. N leaching simulated at 195 cm depth ranging from 6 to 21 kg N ha^{-1} was slightly higher than the range of 2.0-17.0 kg N ha^{-1} previously found by Gabrielle et al. (2005) in the same experiment for the period 1998-2002. This shift was expectable as OM accumulated with the consecutive additional applications i.e. more N was likely to be mineralised and leached over time.

Such issues of using parameters fitted on simulations of OM dynamics during lab short-term incubations to further simulate data set from long-term experiment in field conditions question the relevancy of these parameters transposition. Indeed, lab incubations are usually made in optimal constant conditions for microbial activity with sieved soil which is very different from a structured soil in field in terms of mineralisation behaviour. Nevertheless, part of the sieving-derived bias was taken into account by means of the till parameter (Molina et al., 1983) creating a labile fraction of pool II, used in NCSOIL simulation in lab conditions and set to 0 in CERES-EGC field simulations.

A similar issue arose when using EOM parameters tested against lab incubations data and directly transposed to field conditions. To parameterise OM for field simulations, Peltre et al. (2012) found that the use of I_{ROC} determined through lab measurements, was possible. Here, the C storage induced by EOM applications was also rather well simulated with CERES-EGC using the I_{ROC} indicator to calibrate the resistant pool. EOM C/N ratios appeared also quite important to fit N mineralisation during laboratory incubations. Indeed, larger biases were observed for mineralised N kinetics than for organic C mineralisation kinetics.

Over 13 years of field experiment, CERES-EGC allowed good simulations of crop growth and crop N uptake and partitioning of biomass and N between grain and residues simultaneously. However, soil mineral N contents were not so well simulated, and mainly overestimated right after EOM application. Yet at the end of the Maize-Wheat cycle i.e. at wheat harvest, residual stocks of mineral N were satisfactory. This could suggest that the issue of unperfect soil mineral N simulation rather originates from unperfect simulation of OM mineralisation dynamics than from prediction of the overall quantity mineralised. Indeed mineralisation rate should be lower at the beginning of the cycle but maintained to conserve overall mineralised N quantity. Parameterisation of temperature or water content dependency of biological activities within soil could be inadequate. For instance, Henriksen and Breland (1999) found that microbial biomass growth i.e. N immobilisation was excessively limited by low temperatures. Our model may require different functions of temperature effects on mineralisation for microbial and soil pool dynamics, and this could prevent from too high mineralisation and N leaching in surface layers in winter. Others authors also had issues transposing OM parameters calibrated during lab optimal conditions for mineralisation to non-optimal conditions such as limiting N availability (Giacomini et al., 2007). Part of the discrepancies between lab and field conditions could also be related to the EOM preparation for laboratory incubation. Grounding the EOM enhanced mineralisation and also organisation of mineral N when CN_{EOM} is high. This could be the case for the MSW compost for which important organisation was observed in lab conditions although larger availability of N was simulated in field conditions.

5.3. Impacts of EOM applications

The 4 EOM studied here were all quite stable and C storage efficiencies ranged from 36 to 71%. Ros et al. (2006) found lower efficiencies for similar BIO or GWS composts in Austria with efficiencies of 23 to 40%; Li and Evanylo (2013) found efficiencies of about 20-25% for composts and near 0% for biosolids after repeated applications, whereas Hua et al. (2014) found 15 and 19 % for pig and cattle manure. Despite their similarities, the 4 EOM considered in the present study displayed differences in terms of impacts on soil and crop. EOM type should be chosen carefully depending on the farmer purpose and the farm context. EOM such as BIO and GWS should be chosen to increase SOM, but a stable EOM like BIO with low mineral N content would be preferable to avoid N pollutions. If the purpose is to increase yields and N uptake to substitute synthetic fertilisation, more degradable EOM like MSW would be more adapted.

In the experimental settings, the MSW N_{min} treatment turned out to be interesting concerning N cycle. Indeed, compared to the control N_{opt}, its annual synthetic fertilisation was lower by 78 kg N ha^{-1}, but taking into account net mineralisation, its mineral N input was quite similar and it resulted in equivalent N losses and N uptakes. It decreased yields by 0.69 Mg ha^{-1} only but mostly for early yields when SOM had not increased yet because of only few applications. This confirmed that EOM

amendment practices do have consistent benefits in terms of C storage (Peltre et al., 2012) and synthetic fertiliser saving (Liu et al., 2009) compared to conventional practices.

With EOM applications N losses were always greater than for control. However these EOM application practices did not reflect the reality of farmer's practices since rather large amounts of EOM were applied, between 250 and 428 kg N ha^{-1} on average per application, for the purpose of overstating their effects in time and amplitude. More reasonable EOM application rates are likely to result in less N losses, while maintaining crop yields, storing C in soil and substituting synthetic fertilisers. Similarly, the EOM were applied in September in the experiment, 7 months ahead from maize sowing, without any intercropping. This may explain the high sensitivity of N losses to the quantity of mineral N from EOM applied. Presently, a mandatory practice is to sow an intercalary crop, which would sharply decrease N losses through leaching, while even more enhancing C storage.

Finally, EOM are also known to potentially contain pollutants that are likely to be transferred to soil, water, atmosphere or plants and hazardous for both the environment and human health. Indeed, EOM amendments can induce phosphorus or trace element accumulation in soil and runoff or leaching with water (Edmeades, 2003; Rosen and Allan, 2007; Legind et al., 2012; Cambier et al., 2014), as well as organic pollutants (Houot et al., 2008; Gaylor et al., 2014; Muñoz et al., 2014). They can directly bring pathogens (Jones, 1999; Brinton et al., 2009) or pharmaceutical residues mainly when sewage sludge or animal effluents are applied (Christian et al., 2003; Kemper, 2008; Miège et al., 2009). Besides the present assessment on C and N, impacts of EOM use in agriculture therefore require to be widely evaluated.

6. Conclusion

In this study, we evaluated the ability of the crop model CERES-EGC to predict EOM applications impacts on crop production along with soil C and N dynamics over 13 years at the scale of the QualiAgro field experiment. We first elaborated a strategy for CERES-EGC parameterisation based on laboratory results of soil and soil-EOM short-term incubations, prior to transferring the parameters into CERES-EGC for the purpose of simulating long-term field crop conditions. Then we were able to characterise the different EOM through their specific impacts on C and N cycles.

Concerning soil kinetics parameters, we had to decrease the constant rate of decomposition of active organic pool in soil (pool II) to fit the 13-y simulation of C evolution in the control treatment. Concerning EOM and using (i) similar decomposition rates for EOM grouped by OM stability characteristics, (ii) the same C/N ratio for both labile and resistant pools derived from the measured total C/N ratio and (iii) the I_{ROC} indicator to set the resistant EOM pool size, resulted in performing good simulations of C storage over the span of the 13-y field-scale simulations. Crop yields and N uptake were also well predicted although soil mineral N stocks and nitrate leaching at 50cm depth were generally overestimated. It also ended up with reasonable total N leaching. GWS and BIO

allowed the greatest increases in SOC and soil organic N. The MSW and GWS treatments displayed the highest potential loss of N and MSW treatment the highest apparent N use efficiency by plants. Further research should consider a larger panel of EOM especially with less stable EOM to test the parameterisation strategy. Moreover, if further efforts are still needed to improve N cycle predictions, for limiting environmental impacts and controlling N fertilisation, our results are encouraging for the further use of CERES-EGC as a tool for managing EOM practices in an environmentally and agronomically responsible manner.

Acknowledgments

This publication benefited financial and technical support from the French Environment and Energy Management Agency (ADEME) in the framework of the Prostock-Gessol3 project and from the LabeX BASC of Campus Paris Saclay in both frameworks of the SOCSENSIT project and the IDEX PhD initiative Paris-Saclay 2012. The field experiment QualiAgro has been managed since 1998 within a cooperation between INRA-Veolia Research and Innovation. The authors are grateful to Jean-Noel Rampon, Guillaume Bodineau and Vincent Mercier, in charge of the daily management of the field experiment, all the sampling and analysis.

7. Supplementary data

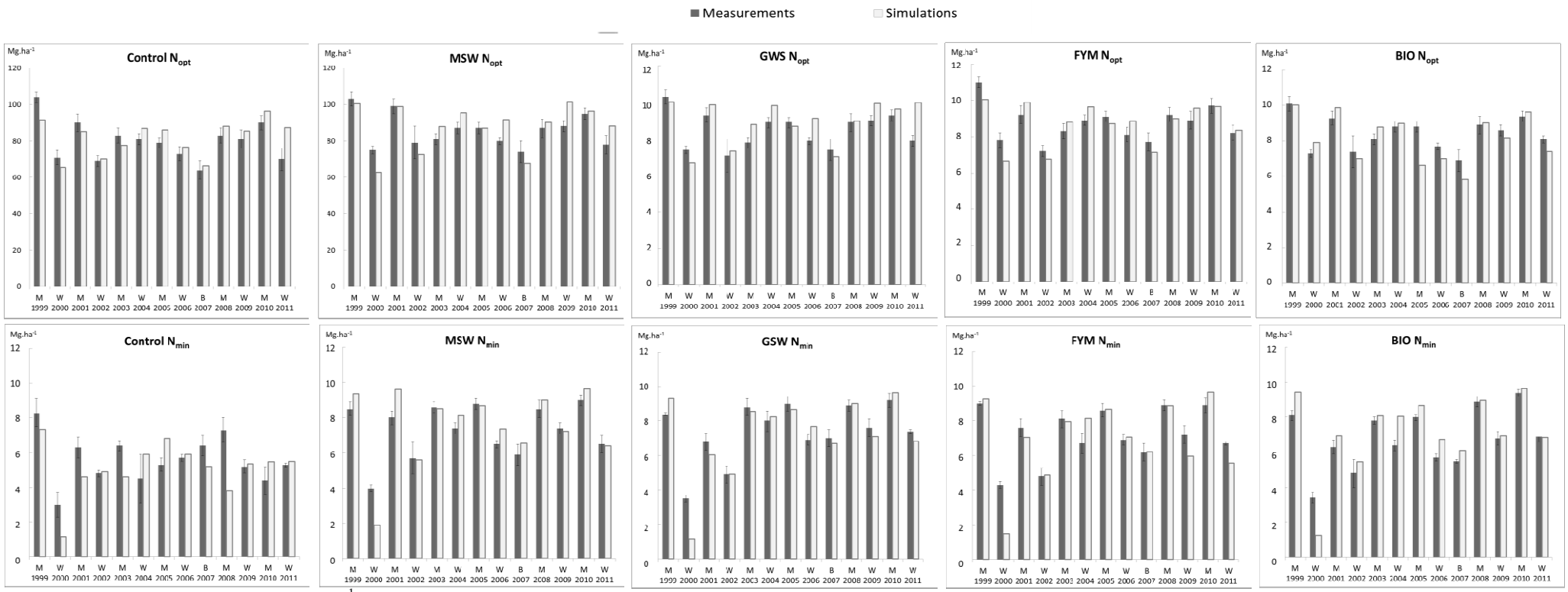


Figure 1S. Crop yield (Mg ha^{-1}) simulated (black) and measured (white).

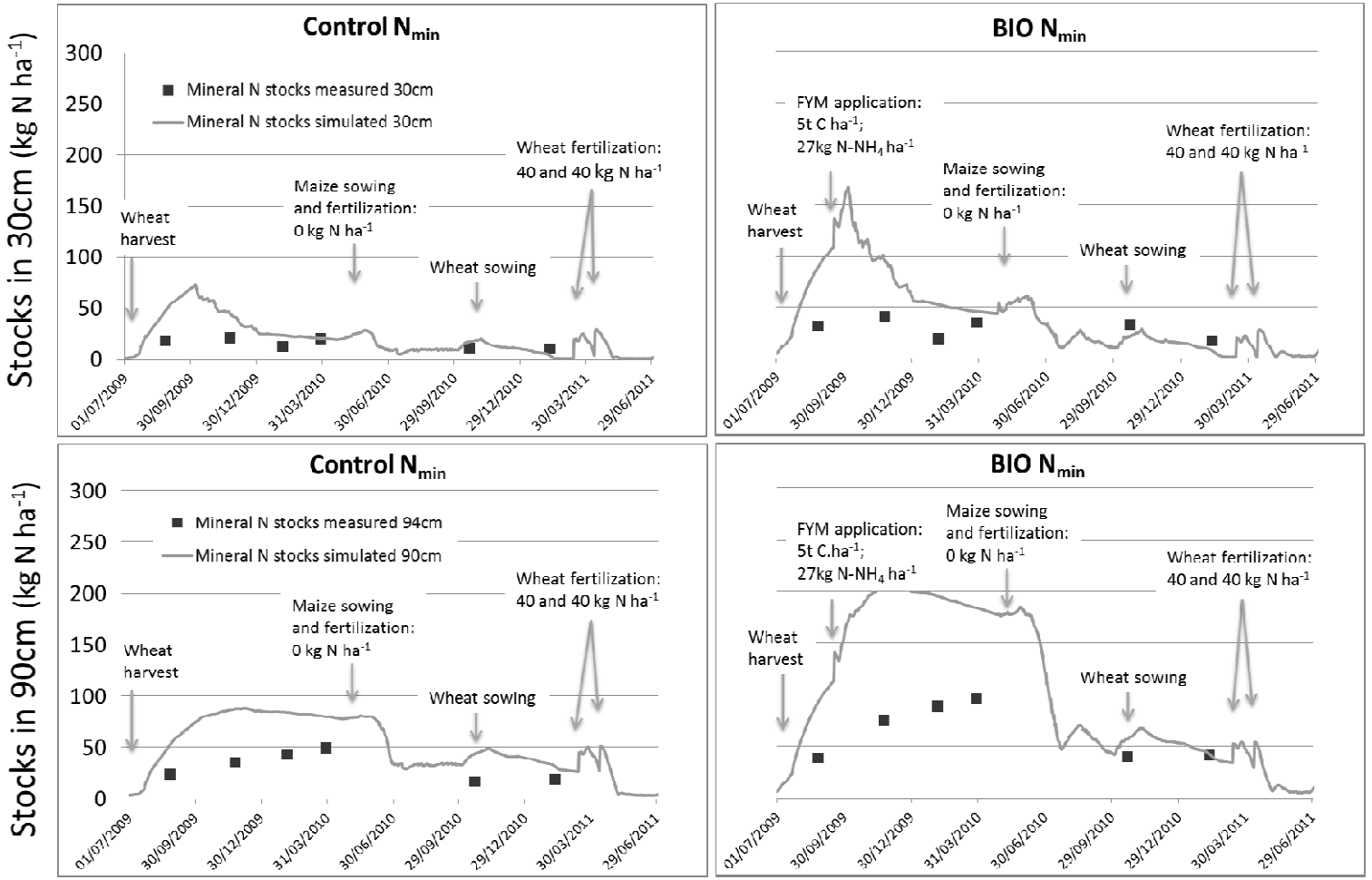


Figure 2S. Soil mineral N stocks (kg N ha^{-1}) measured (dots) and simulated (lines) for control and BIO treatments with the minimum level of fertilisation (N_{min}), focus on the 1 July 2009 – 1 July 2011 period including one maize-wheat cycle. Stocks calculated from contents multiplied by soil density and the considered depth.

Table 1S. Initial main physico-chemical characteristics of the soil (mean of 5 sampled profiles ± standard deviation).

	A (0-30cm)	E (31-51cm)	Btgd (52-94cm)	Bt/IC (95-118cm)	IC (119-149cm)	IIC (150-195cm)
Clay, g kg^{-1}	150 (±10)	219 (±18)	311(±18)	272 (±23)	238 (±22)	226 (±23)
Silt, g kg^{-1}	783 (±20)	724 (±24)	646 (±18)	702(±26)	722(±45)	679 (±38)
Sand, g kg^{-1}	67(±9)	58 (±11)	43 (±7)	26 (±6)	41 (±22)	95 (±23)
Bulk density, Mg m^{-3}	1.32 (±0.0)	1.49 (±0.0)	1.50 (±0.0)	1.50 (±0.0)	1.50 (±0.0)	1.38 (±0.0)
pH, H_2O	6.9 (±0.2)	7.3 (±0.2)	7.5 (±0.3)	7.6 (±0.3)	7.6 (±0.3)	8.4 (±0.1)

Table 2S. RMSE of soil total organic carbon content (TOC) predictions over 13 years and for final TOC for both control Nopt and Nmin treatments (mg C kg-1 soil). In 1998, TOC were equal to 10600 and 10300 mg C kg-1 for Nopt and Nmin respectively.

Pool II size in 1998 (%TOC)	Control N_{opt}		Control N_{min}		mixed results	
	13y of TOC	Final TOC	13y of TOC	Final TOC	13y of TOC	Final TOC
24%	252	107	424	419	349	306
26%	252	154	391	358	329	275
28%	270	213	363	302	319	261
30%	297	315	371	284	336	300
32%	312	364	310	183	311	288

Table 3S. Control soils parameters in the plough layer (0-30cm): total organic carbon content (TOC), mineral N and total C/N ratio were measured, till was set according to Molina et al. (1983). Pool II C/N, χ^2, CV and biases serve as indicators of optimisation performance.

Soil	TOC (mg C kg^{-1} soil)	Mineral N (mg N kg^{-1} soil)	Total C/N ratio	Pool II C/N ratio	Till	χ^2	CV (%C)	CV (% N)	Final bias relatively to % C mineralised mesured	Final bias relatively to % N mineralised mesured	Inflexion bias (28days) relatively to % C mineralised mesured
1998	10192.5	62.8	9.2	10.4	16%	7.19E-02	23	8	-10%	-9%	33%

8. References

AFNOR, 1995. Norme NF ISO 10694. Qualité du sol - Dosage du carbone organique et du carbone total après combustion sèche, Qualité des sols, vol. 1. AFNOR, Paris.

AFNOR, 1998. NF ISO 13878. Qualité du sol - Détermination de la teneur totale en azote par combustion sèche ("analyse élémentaire"), Qualité des sols, vol. 1. AFNOR, Paris.

AFNOR, 2002. NF EN 13654-2. Amendements du sol et supports de culture - Détermination de l'azote - Partie 2 : méthode de Dumas. AFNOR, Paris.

AFNOR, 2003. NF X 31-107. Qualité du sol – Détermination de la distribution granulométrique des particules du sol – Méthode à la pipette. AFNOR, Paris.

AFNOR, 2005. NF ISO 10390. Qualité du sol - Détermination du pH. AFNOR, Paris.

AFNOR, 2009a. Norme XP U 44-162. Amendements organiques et supports de culture - Caractérisation de la matière organique par fractionnement bio-chimique et estimation de sa stabilité biologique. AFNOR, Paris.

AFNOR, 2009b. Norme XP U 44-163. Amendements organiques et supports de culture - Caractérisation de la matière organique par la minéralisation potentielle du carbone et de l'azote. AFNOR, Paris.

AFNOR, 2014. NF EN ISO 10693. Qualité du sol - Détermination de la teneur en carbonate - Méthode volumétrique. AFNOR, Paris.

Annabi, M., Houot, S., Francou, C., Poitrenaud, M., Bissonnais, Y.L., 2007. Soil Aggregate Stability Improvement with Urban Composts of Different Maturities. Soil Sci. Soc. Am. J. 71, 413–423.

Antil, R.S., Bar-Tal, A., Fine, P., Hadas, A., 2011. Predicting Nitrogen and Carbon Mineralization of Composted Manure and Sewage Sludge in Soil. Compost Sci. Util. 19, 33–43.

Barak, P., Molina, J.A.E., Hadas, A., Clapp, C.E., 1990. Optimization of an ecological model with the Marquardt algorithm. Ecol. Model. 51, 251–263.

Beraud, J., Fine, P., Yermiyahu, U., Keinan, M., Rosenberg, R., Hadas, A., Bar-Tal, A., 2005. Modeling Carbon and Nitrogen Transformations for Adjustment of Compost Application with Nitrogen Uptake by Wheat. J. Environ. Qual. 34, 664.

Bisutti, I., Hilke, I., Raessler, M., 2004. Determination of total organic carbon – an overview of current methods. TrAC Trends Anal. Chem. 23, 716–726.

Brinton, W.F., Storms, P., Blewett, T.C., 2009. Occurrence and Levels of Fecal Indicators and Pathogenic Bacteria in Market-Ready Recycled Organic Matter Composts. J. Food Prot. 72, 332–339.

Bruun, S., Christensen, B.T., Hansen, E.M., Magid, J., Jensen, L.S., 2003. Calibration and validation of the soil organic matter dynamics of the Daisy model with data from the Askov long-term experiments. Soil Biol. Biochem. 35, 67–76.

Burger, M., Venterea, R.T., 2008. Nitrogen Immobilization and Mineralization Kinetics of Cattle, Hog, and Turkey Manure Applied to Soil. Soil Sci. Soc. Am. J. 72, 1570.

Cambier, P., Pot, V., Mercier, V., Michaud, A., Benoit, P., Revallier, A., Houot, S., 2014. Impact of long-term organic residue recycling in agriculture on soil solution composition and trace metal leaching in soils. Sci. Total Environ. 499, 560–573.

Chalhoub, M., Garnier, P., Coquet, Y., Mary, B., Lafolie, F., Houot, S., 2013. Increased nitrogen availability in soil after repeated compost applications: Use of the PASTIS model to separate short and long-term effects. Soil Biol. Biochem. 65, 144–157.

Christian, T., Schneider, R.J., Färber, H.A., Skutlarek, D., Meyer, M.T., Goldbach, H.E., 2003. Determination of Antibiotic Residues in Manure, Soil, and Surface Waters. Acta Hydrochim. Hydrobiol. 31, 36–44.

Ciais, P., Wattenbach, M., Vuichard, N., Smith, P., Piao, S.L., Don, A., Luyssaert, S., Janssens, I.A., Bondeau, A., Dechow, R., Leip, A., Smith, P., Beer, C., Van Der Werf, G.R., Gervois, S., Van Oost, K., Tomelleri, E., Freibauer, A., Schulze, E.D., Carboeurope Synthesis Team, 2010. The European carbon balance. Part 2: croplands. Glob. Change Biol. 16, 1409–1428.

Corbeels, M., Hofman, G., Cleemput, O.V., 1999. Simulation of net N immobilisation and mineralisation in substrate-amended soils by the NCSOIL computer model. Biol. Fertil. Soils 28, 422–430.

Edmeades, D.C., 2003. The long-term effects of manures and fertilisers on soil productivity and quality: a review. Nutr. Cycl. Agroecosystems 66, 165–180.

Falloon, P., Smith, P., Szabó, J., Pásztor, L., 2002. Comparison of approaches for estimating carbon sequestration at the regional scale. Soil Use Manag. 18, 164–174.

Gabrielle, B., Da-Silveira, J., Houot, S., Francou, C., 2004. Simulating Urban Waste Compost Effects on Carbon and Nitrogen Dynamics Using a Biochemical Index. J. Environ. Qual. 33, 2333.

Gabrielle, B., Da-Silveira, J., Houot, S., Michelin, J., 2005. Field-scale modelling of carbon and nitrogen dynamics in soils amended with urban waste composts. Agric. Ecosyst. Environ. 110, 289–299.

Gabrielle, B., Laville, P., Duval, O., Nicoullaud, B., Germon, J.C., Henault, C., 2006. Process-based modeling of nitrous oxide emissions from wheat-cropped soils at the subregional scale. Glob. Biogeochem. Cycles 20, 1–13.

Gabrielle, B., Menasseri, S., Houot, S., 1995. Analysis and Field Evaluation of the Ceres Models Water Balance Component. Soil Sci. Soc. Am. J. 59, 1403–1412.

Gabrielle, B., Roche, R., Angas, P., Cantero-Martinez, C., Cosentino, L., Mantineo, M., Langensiepen, M., Henault, C., Laville, P., Nicoullaud, B., Gosse, G., 2002. A priori parameterisation of the CERES soil-crop models and tests against several European data sets. Agronomie 22, 119–132.

Garnier, P., Néel, C., Aita, C., Recous, S., Lafolie, F., Mary, B., 2003. Modelling carbon and nitrogen dynamics in a bare soil with and without straw incorporation. Eur. J. Soil Sci. 54, 555–568.

Gaylor, M.O., Mears, G.L., Harvey, E., La Guardia, M.J., Hale, R.C., 2014. Polybrominated Diphenyl Ether Accumulation in an Agricultural Soil Ecosystem Receiving Wastewater Sludge Amendments. Environ. Sci. Technol. 48, 7034–7043.

Giacomini, S.J., Recous, S., Mary, B., Aita, C., 2007. Simulating the effects of N availability, straw particle size and location in soil on C and N mineralization. Plant Soil 301, 289–301.

Guo, L., Falloon, P., Coleman, K., Zhou, B., Li, Y., Lin, E., Zhang, F., 2007. Application of the RothC model to the results of long-term experiments on typical upland soils in northern China. Soil Use Manag. 23, 63–70.

Hadas, A., Kautsky, L., Goek, M., Erman Kara, E., 2004. Rates of decomposition of plant residues and available nitrogen in soil, related to residue composition through simulation of carbon and nitrogen turnover. Soil Biol. Biochem. 36, 255–266.

Hadas, A., Portnoy, R., 1994. Nitrogen and Carbon Mineralization Rates of Composted Manures Incubated in Soil. J. Environ. Qual. 23, 1184.

Haskett, J.D., Molina, J.A.E., Clapp, C.E., 1986. User's manual for NCSOIL, version 1.0. Dep. of Soil Science, University of Minnesota, Saint Paul, MN.

Henriksen, T.M., Breland, T.A., 1999. Decomposition of crop residues in the field: evaluation of a simulation model developed from microcosm studies. Soil Biol. Biochem. 31, 1423–1434.

Houot, S., Brochier, V., Deschamps, M., Lhoutellier, C., 2008. Input of organic pollutants in soil through compost application: possible transfer to plant. Int. Congr. CH-Soloth. 27th-29th Febr. 2008.

Houot, S., Pons, M.-N., Pradel, M., Caillaud, M.-A., Savini, I., Tibi, A., 2014. Valorisation des matières fertilisantes d'origine résiduaire sur les sols à usage agricole ou forestier. Impacts agronomiques, environnementaux, socio-économiques. Expertise scientifique collective. INRA-CNRS-Irstea. Synthèse 113pp (http://institut.inra.fr/Missions/Eclairer-les-decisions/Expertises/Toutes-les-actualites/Expertise-Mafor-effluents-boues-et-dechets-organiques#).

Hua, K., Wang, D., Guo, X., Guo, Z., 2014. Carbon Sequestration Efficiency of Organic Amendments in a Long-Term Experiment on a Vertisol in Huang-Huai-Hai Plain, China. PLoS ONE 9, e108594.

Janssen, B.H., 1996. Nitrogen mineralization in relation to C:N ratio and decomposability of organic materials. Plant Soil 181, 39–45.

Jenkinson, D.S., Coleman, K., 2008. The turnover of organic carbon in subsoils. Part 2. Modelling carbon turnover. Eur. J. Soil Sci. 59, 400–413.

Jones, C.A., Kiniry, J.R., Dyke, P.T., 1986. CERES-Maize: a simulation model of maize growth and development. Tex. AM Univ. Press Coll. Stn. 194.

Jones, D.L., 1999. Potential health risks associated with the persistence of Escherichia coli O157:H7 in agricultural environments. Soil Use Manag. 15, 76–83.

Kemper, N., 2008. Veterinary antibiotics in the aquatic and terrestrial environment. Ecol. Indic. 8, 1–13.

Kundu, S., Bhattacharyya, R., Prakash, V., Ghosh, B.N., Gupta, H.S., 2007. Carbon sequestration and relationship between carbon addition and storage under rainfed soybean–wheat rotation in a sandy loam soil of the Indian Himalayas. Soil Tillage Res. 92, 87–95.

Lal, R., Griffin, M., Apt, J., Lave, L., Morgan, M., 2004a. Managing soil carbon. Science 304 : 393.

Lal, R., Follett, F., Stewart, B.A., Kimble, J.M., 2004b. Soil carbon sequestration to mitigate climate change and advance food security. Soil Sci. 304, 1623–1627.

Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuries, L., Chaussod, R., Guillotin, M.L., Lineres, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S., 2009. Indicator of potential residual carbon in soils after exogenous organic matter application. Eur. J. Soil Sci. 60, 297–310.

Legind, C.N., Rein, A., Serre, J., Brochier, V., Haudin, C.-S., Cambier, P., Houot, S., Trapp, S., 2012. Simultaneous Simulations of Uptake in Plants and Leaching to Groundwater of Cadmium and Lead for Arable Land Amended with Compost or Farmyard Manure. PLoS ONE 7, e47002.

Li, J., Evanylo, G.K., 2013. The Effects of Long-term Application of Organic Amendments on Soil Organic Carbon Accumulation. Soil Sci. Soc. Am. J. 77, 964–973.

Liu, M., Hu, F., Chen, X., Huang, Q., Jiao, J., Zhang, B., Li, H., 2009. Organic amendments with reduced chemical fertilizer promote soil microbial development and nutrient availability in a subtropical paddy field: The influence of quantity, type and application time of organic amendments. Appl. Soil Ecol. 42, 166–175.

Mamo, M., Rosen, C.J., Halbach, T.R., 1999. Nitrogen Availability and Leaching from Soil Amended with Municipal Solid Waste Compost. J. Environ. Qual. 28, 1074.

Marmo, L., Feix, I., Bourmeau, E., Amlinger, F., Bannick, C.G., De Neve, S., Favoino, E., Gendebien, A., Gibert, J., Givelet, M., Leifert, I., Morris, R., Rodriguez Cruz, A., Ruck, F., Siebert, S., Tittarelli, F., 2004. Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection, vol. 3, taskegroup 4 Exogenous Organic Matter.

Menasseri, S., 1994. Experimental evidence of dynamic compartments in soil organic matter. Modelling and field application. (PhD thesis). Natl. Agron. Paris-Grignon, Paris.

Miège, C., Choubert, J.M., Ribeiro, L., Eusèbe, M., Coquery, M., 2009. Fate of pharmaceuticals and personal care products in wastewater treatment plants – Conception of a database and first results. Environ. Pollut., Special Issue Section: Ozone and Mediterranean Ecology: Plants, People, Problems 157, 1721–1726.

Molina, J. a. E., Clapp, C.E., Shaffer, M.J., Chichester, F.W., Larson, W.E., 1983. NCSOIL, A Model of Nitrogen and Carbon Transformations in Soil: Description, Calibration, and Behavior. Soil Sci. Soc. Am. J. 47, 85–91.

Muñoz, M., Gomez-Rico, M.F., Font, R., 2014. PCDD/F and dioxin-like PCB concentrations during municipal solid waste biomethanation and subsequent composting. Chemosphere 98, 73–77.

Nicolardot, B., Molina, J.A.E., Allard, M.R., 1994. C and N fluxes between pools of soil organic matter: Model calibration with long-term incubation data. Soil Biol. Biochem. 26, 235–243.

Parnaudeau, V., Génermont, S., Henault, C., Farrugia, A., Robert, P., Nicolardot, B., 2009. Measured and Simulated Nitrogen Fluxes after Field Application of Food-Processing and Municipal Organic Wastes. J. Environ. Qual. 38, 268–280.

Peltre, C., Christensen, B.T., Dragon, S., Icard, C., Katterer, T., Houot, S., 2012. RothC simulation of carbon accumulation in soil after repeated application of widely different organic amendments. Soil Biol. Biochem. 52, 49–60.

Rosen, C.J., Allan, D.L., 2007. Exploring the benefits of organic nutrient sources for crop production and soil quality. Horttechnology 17, 422–430.

Ros, M., Klammer, S., Knapp, B., Aichberger, K., Insam, H., 2006. Long-term effects of compost amendment of soil on functional and structural diversity and microbial activity. Soil Use Manag. 22, 209–218.

Smith, J.L., Bell, J.M., Jr, H.B., Bailey, V.L., 2007. The initial rate of C substrate utilization and longer-term soil C storage. Biol. Fertil. Soils 44, 315–320.

Smith, P., Smith, J.U., Powlson, D.S., McGill, W.B., Arah, J.R.M., Chertov, O.G., Coleman, K., Franko, U., Frolking, S., Jenkinson, D.S., Jensen, L.S., Kelly, R.H., Klein-Gunnewiek, H., Komarov, A.S., Li, C., Molina, J.A.E., Mueller, T., Parton, W.J., Thornley, J.H.M., Whitmore, A.P., 1997. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 153–225.

Stolpe, N.B., 2012. Century simulated organic carbon content in soils of south central Chile with diverse ratios of 1500 kPa water to measured clay. Acta Agric. Scand. Sect. B — Soil Plant Sci. 63, 11–18.

Ce premier article nous a permis de montrer qu'il était possible de correctement simuler les effets des amendements organiques sur les cycles carbone (C) et azote (N) dans les différents compartiments sol-plante essentiellement, testé dans le cadre de l'expérimentation longue durée QualiAgro, avec le modèle CERES-EGC/NCSOIL. La calibration a été effectuée également à l'aide de données laboratoire d'incubations du sol et des Produits Résiduaires Organiques (PRO) permettant de mesurer les dynamiques de minéralisation de C et N, et ainsi de paramétrer la matière organique (MO) du sol et des PRO. La transposition du paramétrage déterminé grâce aux données labo pour les simulations les cycles C et N en champ est délicate et a été discutée. Pour la MO sol, le calage d'un paramètre sur les données de champ de QualiAgro concernant l'évolution du C du sol a été nécessaire, tandis que pour les PRO de simples indicateurs soit facilement mesurables soit existants pour un large panel de PRO, ont été utilisé pour leur paramétrisation, et les simulations obtenues de dynamique de la MO mesurés en laboratoire et au champ sont bonnes.

Par rapport aux objectifs de la thèse qui sont entre autres d'explorer le potentiel d'utilisations variées des PRO du territoire sur le stockage de C, les économies en engrais azotés de synthèse et les risques associés de pertes azotées, cette première étape n'a étudié les impacts et testés la paramétrisation que d'un nombre restreint de PRO, 4, et assez similaires par rapport à la diversité de PRO existante ne serait-ce que sur le territoire. Cela n'a également été fait que dans un seul système de culture et sur un sol particulier. Une deuxième étape consisterait donc en la caractérisation de la MO des différents PRO et sols du territoire.

Chapter 3. Parameterisation of the NCSOIL model to simulate C and N short-term mineralisation of exogenous organic matters in different soils.

P.E. Noirot-Cosson[1,2], Dhaouadi K., Etievant V., Vaudour E.[2,1], Houot S.[1,2]

[1] INRA, UMR 1402 ECOSYS, F-78850 Thiverval-Grignon

[2] AgroParisTech, UMR 1402 ECOSYS, F-78850 Thiverval-Grignon

*Email: Paul-Emile.Noirot-Cosson@grignon.inra.fr

1. Abstract

Recycling of Exogenous Organic Matter (EOM) by agriculture can be an efficient way to enhance Soil Organic Matter (SOM) and supply crops with readily available nutrients. It can also cause environmental damages such as nitrate leaching. Characterising EOM for predicting their C and N dynamics of mineralisation when applied on cropped soils is essential for improving their use in agriculture. Measurements of C and N mineralisation through soil laboratory incubations of 18 EOM were used along with EOM biochemical fractionations for parameterising the NCSOIL model and simulating these C and N mineralisation kinetics. The soil type did not interfere significantly with EOM mineralisation and parameterisation allowing to extrapolate parameters adjusted into one soil to other soil types. Four groups of EOM were distinguished based on their C and N dynamics: (1) stable composts, (2) more reactive composts and stable manures, (3) manures and (4) very reactive EOM as sludges, litters that should be used as fertilisers. The use of easily accessible indicators such as the I_{ROC} indicator (Lashermes et al., 2009) or measured organic C/N ratio (CN_{EOM}) were appropriate for parameterisation of groups 1 and 4. A regression was found (R^2 of 0.967) to better improve the pool sizes for the remaining EOM, relating an optimised resistant pool size and Van Soest fractions and CN_{EOM}. Further research remained needed to improve the parameterisation.

Key-words: exogenous organic matter, compost, NCSOIL, C and N mineralisation, lab incubations, biochemical fractionation, parameterisation

2. Introduction

Recycling exogenous organic matter (EOM) by agriculture has recently become of major interest in peri-urban areas which are likely to be source of urban waste derived EOM. As a matter of fact, EOM can both supply nutrients for plants and enhance soil organic matter contents (SOM) (Stratton et al., 1995; Marmo et al., 2004) especially in the context of increase in fertiliser prices and cropland soil organic carbon (SOC) depletion (Ciais et al., 2010). However, the progressive release of mineral nitrogen (N) after EOM application on soil makes more difficult the assessment of N availability for crops and raises environmental risks of excess mineral N leaching (Chalhoub et al., 2013). For a proper EOM use, the mineralisation behaviour of EOM in terms of C and N dynamics remained to be better predicted after application on soil (Parnaudeau and Génermont, 2014).

Several studies have been conducted to predict EOM mineralisation behaviour through biochemical characterisation of EOM (Gabrielle et al., 1995; Thuriès et al., 2002; Morvan et al., 2006; Morvan and Nicolardot, 2009) and tried to relate the biochemical composition to simple models (Thuriès et al., 2002). Both biochemical indicators and models improved the understanding of EOM behaviour in laboratory controlled conditions. The NCSOIL model simulates SOM dynamics through C and N flows between series of pools, describing SOM and EOM additions (Molina et al., 1983; Nicolardot et al., 1994). This model has successfully simulated C and N soil dynamics in various soil under laboratory conditions (Corbeels et al., 1999). Moreover, the NCSOIL model has also been coupled to the crop model CERES-EGC therefore making possible to simulate EOM mineralisation under field conditions (Gabrielle et al., 2005).

Few authors worked on mineralisation dynamics of EOM using such model (Burger and Venterea, 2008; Antil et al., 2011). However, they studied only a small number of EOM types, and further study remains to be made on a larger panel of EOM. In the NCSOIL model, EOM are characterised with conceptual pools with numerous parameters which can allow good simulations but are poorly related to simple indicators or measurable fractions such as biochemical fractions (Hadas and Molina, 1993), which makes such model parameterisation still challenging. For this purpose, indicators characterising EOM could be used such as the I_{ROC} which determines a proportion of OM which will remain in soil for a long-term (Lashermes et al., 2009). Peltre et al., (2012) showed it could be used to predict soil organic C under repeated applications of EOM with the RothC model. Other indicator such as the C/N ratio has been used to simulate N mineralisation (Chadwick et al., 2000; Morvan et al., 2006). The biochemical fractions determined through the Van Soest fractionation method (AFNOR, 2009a) have been used to simulate organic N mineralisation from EOM (Parnaudeau et al., 2004; Chalhoub et al., 2013). Indeed, the integration of simple indicators in models could be an interesting option to answer to the parameterisation challenge for simulation models. In a previous work, the NCSOIL and CERES-EGC models have been parameterised and used to simulate C and N dynamics in the long-

term Qualiagro experiment where EOM have been regularly applied since 15 years (Noirot-Cosson et al., 2016). The NCSOIL model was parameterised for EOM based on the I_{ROC} and C/N ratio indicators.

This was the first step of a more general objective dedicated to the optimisation of EOM use at a regional scale. The territory was the "Plain of Versailles", a peri-urban region located 30 km west from Paris, France. In the present work, the C and N mineralisation of all EOM that could be found on the territory were studied in the four dominant soils, The objectives were: (1) to evaluate the variation of EOM C and N mineralisation in different soils, (2) to parameterise NCSOIL for the large panel of EOM based on their characteristics (3) to characterise EOM for their potential N availability for crops or efficiency at increasing soil organic matter in relation with their characteristics.

3. Materials and methods

3.1. Soils and EOM

3.1.1. Soils

Four soil types that represented the main crop soils from the "Plaine de Versailles" were used (Table 1). According to the World Reference Base for Soil Resources they corresponded to: (1) "Fontenay", a calcaric cambisol, (2) "Feucherolles", an arenic cambisol, (3) "Grignon", a haplic luvisol; (4) "Qualiagro", a haplic luvisol (retic). The plough horizons were sampled, air-dried and 2mm-sieved for further characterization. All analysis were done at the central laboratory of analysis of INRA (LAS, Arras, France): particle size distribution (g/kg) in coarse sand (200-2000µm), fine sand (50-200µm), coarse silt (20-50µm), fine silt (2-20µm), clay (< 2µm) (AFNOR, 2003), the soil pH (AFNOR, 2005) and carbonate contents (AFNOR, 2014), total organic C (TOC) and N by dry combustion using elementary analyser after removal of carbonates by acidification (AFNOR, 1995), and the C to N ratio calculated.

Table 1. Characteristics of the four soil types. Granulometry: clay (< 2µm), Fine Silt (2-20µm), Coarse Silt (20-50µm), Fine Sand (50-200µm), Coarse Sand (200-2000µm). TOC stands for Total Organic Carbon.

Location	Soil type	Granulometry (g/kg)					Calcareous content (g/kg)	pH	TOC (g/kg)	C/N
		Clay	Fine Silt	Coarse Silt	Fine Sand	Coarse Sand				
Fontenay	calcaric cambisol	350	219	238	146	47	202.0	8.3	18.1	11.2
Feucherolles	arenic cambisol	170	144	259	369	58	1.8	7.4	15.2	11.0
Grignon	haplic luvisol	346	222	295	101	36	33.9	8.1	22.7	10.4
QualiAgro	haplic luvisol (retic)	149	270	513	60	7	0.0	6.9	9.2	9.5

3.1.2. Exogenous Organic Matters

The 18 EOM available or potentially available in the region were collected and analysed (Table 2). They included EOM from urban activities among which are many green waste (GW) based composts such as GWb, GWa, Bio resulting from composting of biowaste mixed with green waste, "Fertylis" mixing GW compost and bone meals, GWS issued from co-composting sewage sludge and GW, but also composted leaves (Lc), a Mulch (Mul), and municipal solid waste compost (MSW) and two sewage sludges, one dried (SSd) and one limed (SSl). They also included EOM deriving from animal effluents: four horse manures, either mixed with high amounts of straw (HMh) or with low amounts of straw (HMl) or with wooden shavings (HMw) or composted (HMc), a cattle manure (CM), poultry litter either dried (PL) or composted (PLc) and an imported dried pig slurry (Humival).

Mineral N was extracted with 200 mL of 0.5 M KCl from 50 g fresh samples and analysed by colorimetry on a continuous flow analyser (Skalar, The Netherlands). The EOM organic C and organic N (C_{EOM} and N_{EOM}) were determined by elemental analysis of air-dried 200 µm ground soil samples, and corresponding organic C/N ratio (CN_{EOM}) were then calculated for all EOM.

Total organic matter was measured based on mass loss after calcination at 550°C (AFNOR, 2011). Total organic matter was fractionated into soluble (*SOL*), hemicellulose- (*HEM*), cellulose- (*CEL*) and lignin-like (*LIG*) fractions according to their solubility in detergents using crude-fibre analysis derived from the Van Soest and Wine method (Van Soest and Wine, 1967) as described in the French standard XP U 44-162 (AFNOR, 2009a). The soluble organic fractions include both organic matter extractable in boiling water (100°C) and hot neutral detergent (NDF fraction of the Van Soest extraction procedure). The I_{ROC} indicator, which reflects the proportion of EOM that may be retained in the soil for decades (Lashermes et al., 2009), was calculated based on the Van Soest fractions and the proportion of EOM-TOC mineralised after 3 days of laboratory incubation (C_{3d}) as described in the French standard XP U 44-162 (AFNOR, 2009a):

$$I_{ROC} = 44.5 + 0.5\ SOL - 0.2\ CEL + 0.7\ LIG - 2.3\ C_{3d} \qquad [1]$$

where *SOL*, *CEL* and *LIG* are expressed as % of *TOM* and C_{3d} as % of TOC. Biochemical and chemical EOM characteristics are presented in Table 2.

3.1.3. Incubation experiments

The potential mineralisation of total soil organic C and N was assessed in the 4 representative soils during 91-day incubations under controlled conditions (temperature of 28°C and gravimetric water content 23%). Fresh 4mm-seived soil samples (equivalent to 25 g of dry soil) were incubated in air-tight 500 mL jars, with four replicates, for either the organic C or N mineralisation measurements in separate parallel experiments. The C-CO_2 was trapped in 10 mL of NaOH (0.5 M), which was periodically sampled and replaced. The C-CO_2 trapped in NaOH was analysed by colorimetry on a continuous flow analyser (Skalar, The Netherlands). The opening renewed the jar atmosphere and

maintained aerobic conditions. On the same dates, the mineral N ($N\text{-}NH_4^+$ and $N\text{-}NO_3^-$) was extracted with 100 mL of 0.5 M KCl and then analysed by colorimetry on a continuous flow analyser (Skalar, The Netherlands). All jars incubated for N mineralisation were opened simultaneously to those used for C mineralisation to ensure aerobic conditions.

Table 2. Summary description of the EOM measurements.

EOM	C_{EOM}	N_{EOM}	Nmin	CN_{EOM}	SOL	HEM	CEL	LIG	C_{3d}	I_{ROC}
	g/kg of Dry Matter				(%)				(% C_{org})	
Biowaste Compost (Bio)	68.6	8.0	0.45	8.6	49.2	8.8	12.9	29.1	1.3	84.0
Green Waste compost BYS (from Bio Yvelines Service) (GWb)	187.8	17.6	0.55	10.6	51.5	4.9	11.5	31.2	1.9	85.9
Horse manure composted (HMc)	178.7	14.0	0.62	12.8	59.8	6.1	12.6	21.5	2.2	81.9
Leaves composted (Lc)	174.7	13.3	0.25	13.2	61.9	1.0	7.6	29.6	1.6	90.9
Green Waste and Sludge compost 2009 (GWS)	265.6	20.9	2.46	12.1	45.4	7.2	21.9	25.6	6.7	74.8
Green waste and bone meals (“Fertylis”)	250.9	14.9	0.01	16.7	78.0	3.5	6.2	12.5	2.5	85.3
Cattle manure (CM)	363.3	19.4	0.51	18.7	59.8	12.2	16.6	11.4	2.8	72.6
Municipal Solid Waste compost (MSW)	308.0	17.2	0.09	16.1	47.6	6.6	23.3	22.5	18.1	63.8
Green Wastes Sep (from SEPUR) (GWa)	314.0	13.6	0.02	23.1	54.2	3.9	19.6	22.2	2.3	78.0
Horse manure with wood shavings (HMw)	385.2	13.1	0.07	29.3	58.9	6.5	26.3	8.3	8.6	54.7
Horse manure high straw content (HMh)	359.5	15.8	0.11	22.8	54.2	13.9	22.4	9.5	2.5	67.9
Horse Manure, low straw content (HMl)	251.3	7.9	0.06	31.9	43.9	47.2	1.9	7.0	5.4	58.7
Mulch (Mul)	454.7	5.2	0.19	87.9	61.3	7.6	22.7	8.5	2.3	71.2
Dried Sewage Sludge (SSd)	380.9	69.1	2.38	5.6	80.8	13.6	1.6	4.0	26.6	26.2
Limed Sewage Sludges (SSl)	263.0	44.5	0.14	5.9	92.7	5.5	0.8	1.0	27.5	28.1
Dried Poultry Litter (PL)	315.3	35.8	2.16	8.8	62.2	21.5	14.4	1.9	34.8	0.0
Dried Pig Slurry (“Humival”)	402.8	52.1	0.60	7.8	82.8	7.3	5.6	4.2	15.5	52.1
Poultry litter composted (PLc)	175.3	18.5	1.14	9.5	57.8	12.0	18.9	11.3	10.3	53.9

C_{EOM}, N_{EOM}, N_{min} and CN_{EOM} stand for organic C content, organic N content, mineral N content and EOM organic C/N ratio. SOL, HEM, CEL and LIG stand for Soluble, hemicellulose-, cellulose- and lignin-like EOM fractions.

The kinetics of C and N mineralisation were also measured for all applied organic amendments during 91-day incubations of soil-organic amendment mixtures under laboratory controlled conditions as described above (temperature of 28°C and gravimetric water content 23%) following a standard method (AFNOR, 2009b). The "Qualiagro"soil was used for the incubations and prepared as previously described. Each incubation was performed in four replicates. The equivalent of 25 g of dry soil was mixed to 0.5 to 1 g of EOM and sealed in hermetic jars of (0.5 L) before either CO_2 or N mineral measurements. Mineral N was added at the beginning of the incubation (25 mg $N\text{-}NO_3$ kg^{-1} soil) to prevent a mineral N stress affecting EOM organic matter degradation. As previously, the $C\text{-}CO_2$ was trapped in 10 mL of NaOH (1 mol L^{-1}) periodically sampled and replaced. The $C\text{-}CO_2$ trapped in NaOH was analysed by colorimetry on a continuous flow analyser (Skalar, The Netherlands). The jars were left opened 2 min after sampling to renew the air and avoid anaerobic conditions. On the same days, four replicate jars were destructively sampled for inorganic N measurements made by colorimetry after extraction with 200 mL of KCl (1 mol.L^{-1}) with a soil to KCl ratio of 1:5.

Additionally, 4 EOM were incubated in the 4 soils during up to 175 days following the same procedure. This concerned SSd, HMl, GWb and GWS.

Incubations data with the four types of soils were used to study the soil effect on EOM mineralisation, whereas the results of EOM incubated in the same type of soil were used to compare EOM behaviour and model parameterisation.

3.2. Modelling

3.2.1. The NCSOIL model

The NCSOIL model (Molina et al., 1983; Barak et al., 1990; Hadas and Molina, 1993) was used to simulate C and N mineralisation of soil organic matter (SOM) during the lab experiment. Assuming a homogeneous soil at both constant temperature and water content, NCSOIL simulates EOM decomposition after incorporation into soil, nitrification, immobilisation and mineralisation of nitrogen, along with SOM decomposition and formation, by means of a system of controlling rates and decisional switches amongst a series of pools or compartments (Haskett et al., 1986). The OM flow series implies 4 pools involved in N and C turnover and represented in Fig. 1: (i) pool 0, termed the "zymogenous biomass", developed from the degraded EOM, (ii) pool I, representing the endogenous microbial biomass; (iii) pool II, a fraction of humified OM termed "mineralisable SOM"; (iv) pool III, corresponding to highly-humified OM with a very low decomposition rate, and supplied by a fraction (EFSINK) of decomposed pool II. EOM and soil pools 0, I and II are divided into labile and resistant "fractions" with different degradation rates (k_l and k_r). Each pool i, is defined by its C content (or "pool size"), Cp_i, its C to N ratio, $R_i=Cp_i/Np_i$ (where Np_i is its N content), its decomposition rate, k_i, and a microbial use efficiency factor (EFFAC), i.e. the fraction of decomposed C that is incorporated

into the microbial biomass pool I or the zymogenous biomass pool 0 for EOM (Hadas et al., 2002) and a humification factor (EFSINK(i)) i.e. the fraction of decomposed C that is incorporated into the following mineralisable soil pool (II or III). It decomposes following first-order kinetics:

$$-(dC_i / dt) = k_i \times C_i \qquad [2]$$

The N flows follow the C flows respecting the C/N ratio of the pool being decomposed, inorganic N being added or withdrawn by the following pool to maintain its C/N ratio. Organic residues and/or EOM can also be modelled and are characterised as the soil pools but with different C/N ratios for labile and resistant components. Parameters are summarised in Table 3.

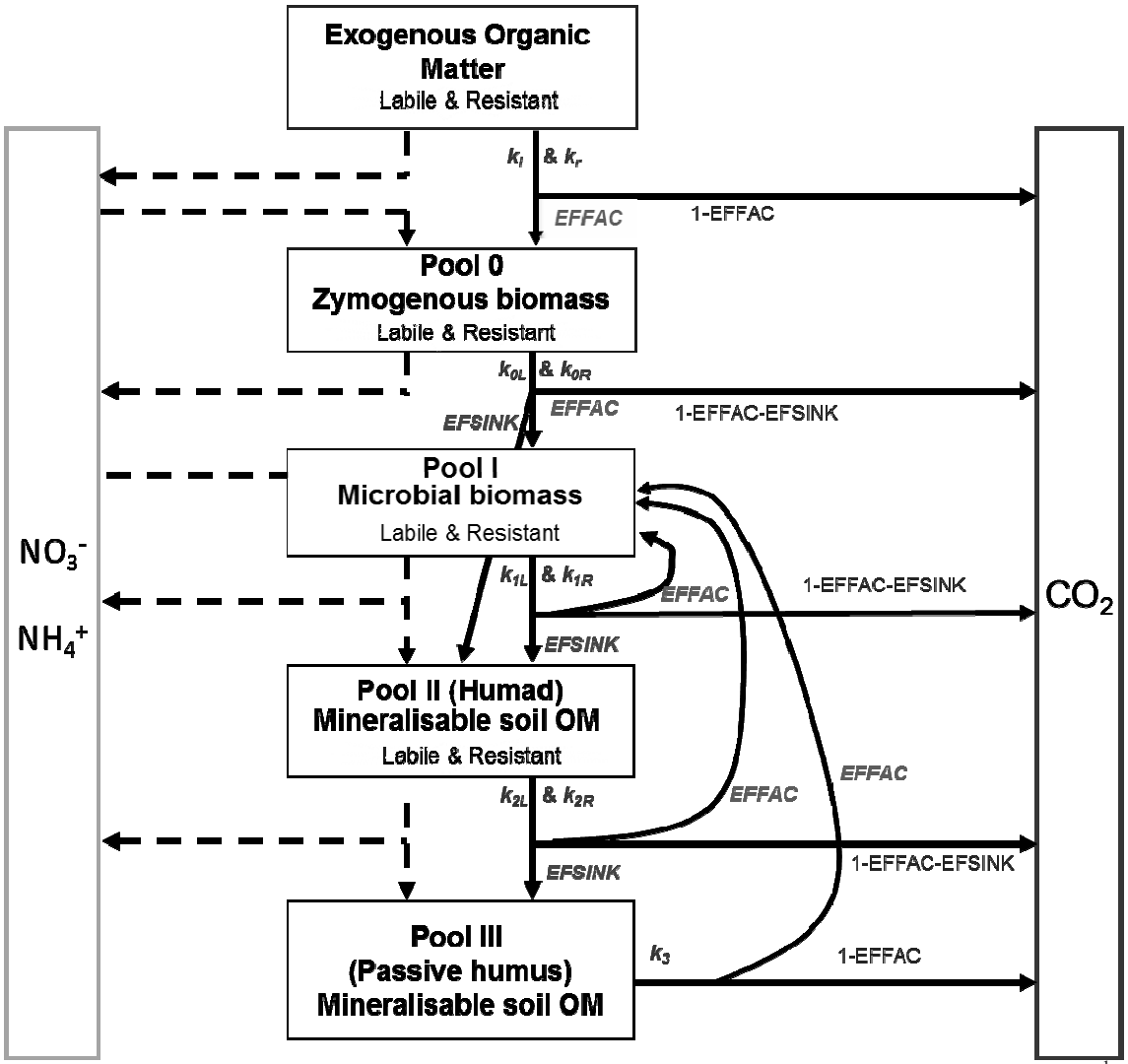


Figure 1. Structure of the NCSOIL model, k_{ij} stands for decomposition rate of pool ij in days^{-1}, i varying from 0 to 3 and j being either labile or resistant, EFFAC for the proportion of decomposed OM recycled in biomass pool, EFSINK for the proportion of the decomposed OM recycled in the following mineralisable soil pool.

3.2.2. Model parameterisation

3.2.2.1. "QualiAgro" soil parameterisation

Only the "QualiAgro" soil was parameterised for NCSOIL simulations. The model was parameterised using a set of parameters from Molina (1983), Nicolardot et al., (1994), Corbeels et al. (1999) and Hadas et al., (2004) (Table 3). The initial pool II size and pool II C/N ratio were parameterized using the optimisation module added to NCSOIL (Barak et al (1990). This module adjusts unknown parameters, by fitting simulations against measurements of CO_2 evolution and inorganic N accumulation at all the sampling dates during the incubation of control soils, using the Marquardt algorithm which minimises a merit function:

$$\chi^2 = \sum_{i=1}^{N} \left[(y_i - y(x_i, a)/\sigma_i \right]^2 \qquad [3]$$

Where i=1,2,…,N the number of observations; yi the ith measured dependent variable; $y(x_i, a)$ is the predicted value of a variable at x_i based on the parameter values a, and the standard deviation of y_i. The kinetics of C and N mineralisation of the 4 soils were used to optimise the soil parameters.

3.2.2.2. EOM parameterisation

EOM decomposition rates

Fig. 2 summarised the successive steps used for EOM parameterisation in the NCSOIL model. The C and N mineralisation kinetics of the 18 EOM incubated with the soil "QualiAgro" were used.

First decomposition rates of labile and resistant pools of EOM (k_{EOM}: k_l and k_r) were obtained by fitting non-linear regressions (NLR) as described in Gabrielle et al., (2004). The kinetics of C mineralisation were used (Step 0 in Fig. 2).

The NLR used a two-compartment, exponential C-decay model fitted against the CO_2 release data of the incubated soil-EOM mixtures, after subtracting the contribution of endogenous soil OM, assuming, as Mamo et al. (1999), that the mineralisation of endogenous SOM did not interfere with the application of EOM. This model can be written as:

$$C_{release} = C_l\left[1 - \exp(-k_l t)\right] + C_r k_r t \qquad [4]$$

Where $C_{release}$ is in mg C kg^{-1} dry soil, t is time (day), C_l and C_r are the sizes of the labile and resistant fractions, respectively (mg C kg^{-1} dry soil), k_l and k_r the decomposition rates of the labile and resistant fractions, respectively (day^{-1}).

As C_r can be calculated from C_l as (Ci-C_l), where Ci is the total EOM carbon content in mg C kg^{-1} dry soil, the C_l, k_l and k_r parameters are the only unknown parameters. C_r and C_l were then expressed in % of C_i.

The NLR procedure fits all parameters simultaneously. When the first part of the curve was hardly fitted, it tended to be offset by the fitting of the second part resulting in strong biases in slopes i.e. k_r estimates. However, the latter slope was more sensitive to fitting bias as it is generally 100 times lower than the slope of the first part. It was thus decided to calculate a new k_r by directly using the slope of the second phase between day 49 and the last incubation day. Both k_l and k_r were multiplied by a 1.5 factor as NLR do not take into account microbial recycling in NCSOIL model (Gabrielle et al., 2004).
The decomposition rates (k_l and k_r) were then grouped using Ascending Hierarchical Classification (AHC) with the Ward's method (Ward, 1963), based on 3 discriminative characteristics: k_r, I_{ROC}, CN_{EOM} on one hand and k_l, I_{ROC}, CN_{EOM} on the other hand.
The clustering was made avoiding isolated EOM. Within each cluster, the k_l and k_r were averaged and assigned to each EOM in the cluster for NCSOIL simulations.

EOM pool sizes

We tested two methods to parameterise these fraction sizes for NCSOIL: method-A consisted in using the I_{ROC} indicator to define the size of the resistant pool and method-B in optimising the fraction sizes with the optimisation module (Step 2 in Fig. 2).

EOM pool C/N ratios

We also tested two methods of parameterisation: method-1 consisted in setting both pools C/N ratios as the CN_{EOM} measured, whereas in the method-2, we optimised CN_l and recalculated CN_r.
We tested all methods combinations (A;B)x(1;2), resulting in 4 parameterisation tests for each EOM.

Table 3. NCSOIL soil OM and EOM parameters, sources and setting methods

NCSOIL pools	Pool 0 and I	Pool II	Pool III	EOM
Initial Size (mg C.kg^{-1} soil)	C_{P0}=0 C_{PI}=2.6% of TOC[2]	Optimised	TOC-C_{PII}-C_{PI}	C_{EOM}
Labile fraction (% total C of pool)	56%[2]	16[4]	-	$C_l = 100 - C_r$
Resistant fraction (% total C of pool)	44%[2]	84[4]	-	$C_r = I_{ROC}$ or optimised (see in text 2.4.3)
C decay rate constants (k in day^{-1})			0.000055[2]	
Labile	0.332[1;2]	0.16[1]		k_l = Predicted (see §3.2)
Resistant	0.0404[1;2]	0.002[5]		k_r = Predicted (see §3.2)
Fraction of decomposed C recycled into following pool (see fig. 1) (EFFAC)	0.6[1,2]	0.4[1]	0.25[1]	0.6
Fraction of decomposed C incorporated into the following pool (see fig. 1) (EFSINK)	0.2[1;2]	0.2[1;2]	-	-
C/N ratio	6.0[1;2]	Optimised	$CN_{PIII} = C_{PIII} / [(TOC/CN_{TOC})-(C_{PI}/CN_{PI})-(C_{PII}/CN_{PII})]$	CN_{EOM}
Labile	-	-	-	CN_l = Set as CN_{EOM} or optimised
Resistant	-	-	-	$CN_r = C_r / (C_{EOM} / CN_{EOM} - C_l / CN_l)$

PI stands for pool I, PII for Pool II, $_r$ for EOM resistant pool and $_l$ for labile, TOC for Total Organic Carbon Content, EOM for total Exogenous Organic Carbon, C for pool size (mg C.kg^{-1} soil), CN for organic C/N ratio, k for C decay rate constant ; [1]Corbeels et al. (1999), [2]Nicolardot et al. (1994a), [3]Hadas et al. (2004), [4]Molina et al. (1980); [5]Noirot-Cosson et al., 2016

3.2.2.3. Parameterisation evaluation

To evaluate the goodness of the simulation fitting to the measurements, we first visually compared the simulated with the experimental kinetics. To evaluate the appropriateness of the use of I_{ROC} and CN_{EOM}, we also evaluated the difference between I_{ROC} and optimised C_r on one hand and between CN_{EOM} and optimised CN_l or recalculated CN_r on the other hand.

3.2.3. statistical evaluation of soil effect on EOM mineralisation

The influence of soil characteristics on parameters of EOM mineralisation kinetics was assessed through statistical comparison of NLR results of C-CO_2 kinetics of mineralisation of 4 EOM incubated into the 4 soils (Step 1 in Fig. 2). The parameters were subjected to a Kruskal-Wallis test at $p<0.05$ to determine significant effects and differences of soils or EOM treatments. Statistical analyses were performed using XLSTAT 2014 software.

3.2.4. Factor analysis: relation between EOM parameters and characteristics

Factorial analyses were performed using XLSTAT 2014 software to relate EOM characteristics, NCSOIL parameters and EOM groups of similar mineralisation behaviours (Step 3 in Fig. 2). First, a PCA and a correlation matrix using the Pearson correlation coefficient were made to relate the k_{EOM} parameters to measured Van Soest fractions (LIG, CEL, HEM, SOL), I_{ROC}, optimised C_r, $CN_{EOM.}$ Secondly, a PCA was performed relating EOM and EOM groups based on mineralisation behaviours to Van Soest fractions and CN_{EOM}.

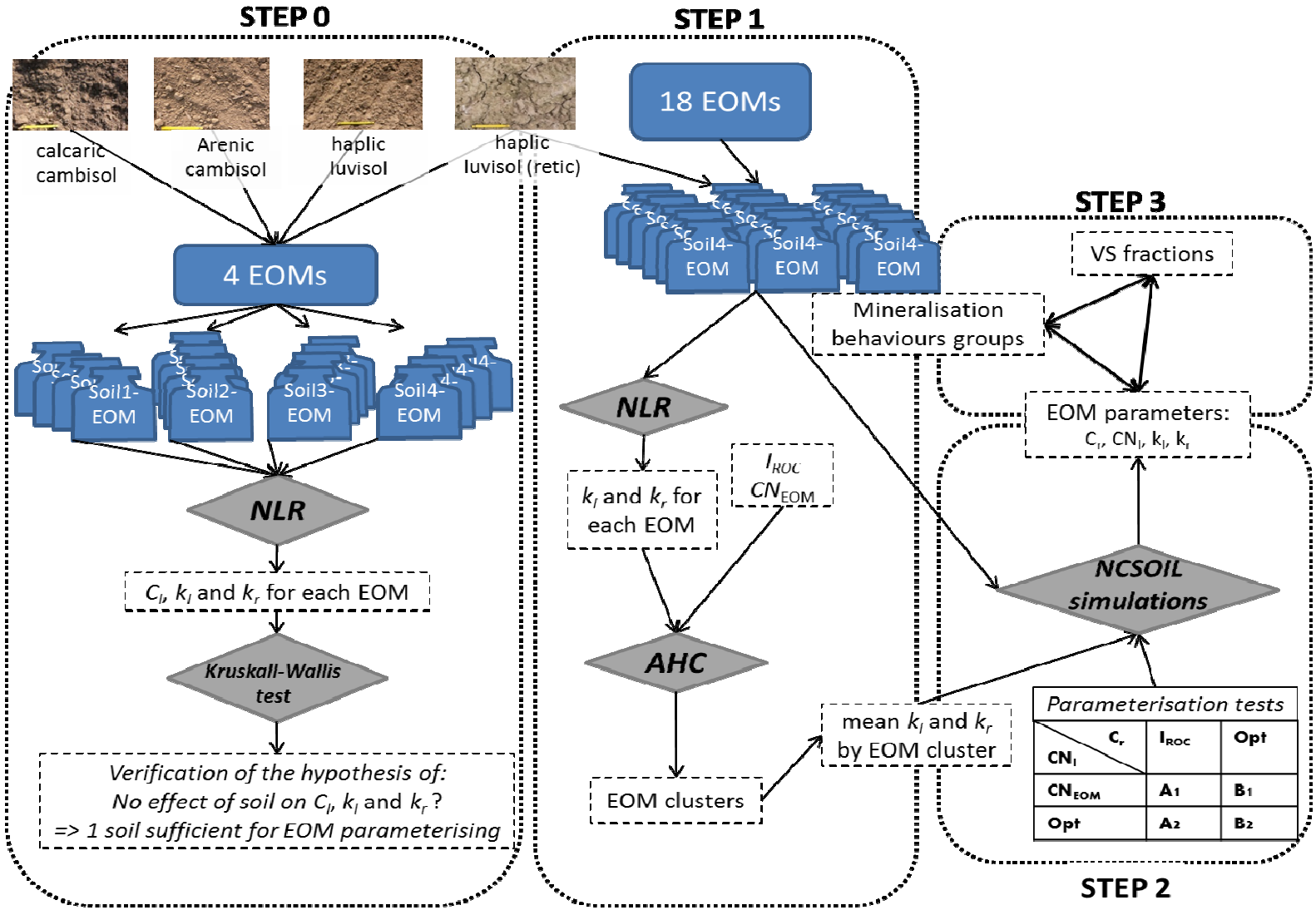


Figure 2. Overall approach. C_l, k_l, CN_l and C_r, k_r, CN_r stand for pool size, decomposition rate and organic C/N ratio of EOM labile and resistant pools respectively. NLR, AHC stand for Non-Linear Relationship or Regression and Ascending Hierarchical Classification. Opt stands for optimised and VS for Van Soest.

4. Results

4.1. Variation of C mineralisation kinetics of EOM with soil type

The effect of soil type on the dynamics of EOM mineralisation was tested on 4 EOM (GWb, GWS, HMh, SSd). The kinetics of mineralisation of these 4 EOM were fitted with NLR model and the effect of soil was statistically tested on the parameters of the NLR equations (Table 4). The soil effect was never significant, (high p-values). On the contrary, EOM effect was generally significant with p-values always lower than 0.05 except for k_r directly from NLR (p-value 0.38).

These results showed the absence of soil effect on net EOM mineralisation compared to the differences between EOM and therefore confirmed the relevancy of studying EOM characteristics and parameterisation based on incubations made with only one type of soil.

Table 4. Kruskal-Wallis tests of soil effects of EOM parameters

P-Value test of effect ~x on factor *y	Data with 4 soils	
	~EOM	~SOILxEOM interaction
*k_l	0.006597	0.9778
*C_l	0.004848	0.7924
*k_r	0.01904	0.6785

4.2. Classification of EOM for k_{EOM} parameterisation

k_r could not be determined for 5 EOM because mineralisation slopes were negative. They were thus excluded for classifications. Dendrogram trees for k_r and k_l are presented in Fig. 3a and b with associated cluster characteristics in Fig. 4.

For k_l, height differences suggested to divide the whole EOM group into 4 EOM clusters. However, in order to avoid cluster with only one EOM, and because clusters presented quite similar k_r we decided to group three sub-clusters together in the k_l-S cluster, leading to a distinction of cluster k_l-U and cluster k_l-S for k_l attribution. The cluster k_l-U was characterised by low CN_{EOM} (7.0), low I_{ROC} (26.8) and high k_l (0.376). Cluster k_l-S on the contrary was characterised by higher CN_{EOM} (22.4), higher I_{ROC} (73.1) and lower k_l (0.114). Cluster k_l-S gathered all the composts and the manures, when cluster U the sludges, non composted litter and slurry (humival).

For k_r, height differences suggested a division into 5 clusters with k_r-Sr, k_r-Sl, k_r-Sludges, HMl and Mul. Again, in order to avoid cluster with only one EOM and as the latter HMl and Mul had both high CN_{EOM} and high k_r, it was thus decided to keep them together in the cluster k_r-U with high CN_{EOM}, leading to 4 distinct clusters for k_r. Clusters k_r-Sr and k_r-Sl included all the composts and manures (except HMl). These two clusters k_r-Sr and k_r-Sl were quite similar in terms of CN_{EOM} (17.6 and 17.1) but differed by their I_{ROC} (78.3 and 58.1%) and their k_r (0.971E-3 and 1.59 E-3).

The cluster k_r-Un was characterised by higher k_r (2.09E-3), very low I_{ROC} (27%) and very low CN_{EOM} (5.8) while k_r-Uc with the highest k_r (2.59E-3), quite high I_{ROC} (46%) and very high CN_{EOM} (59.9).

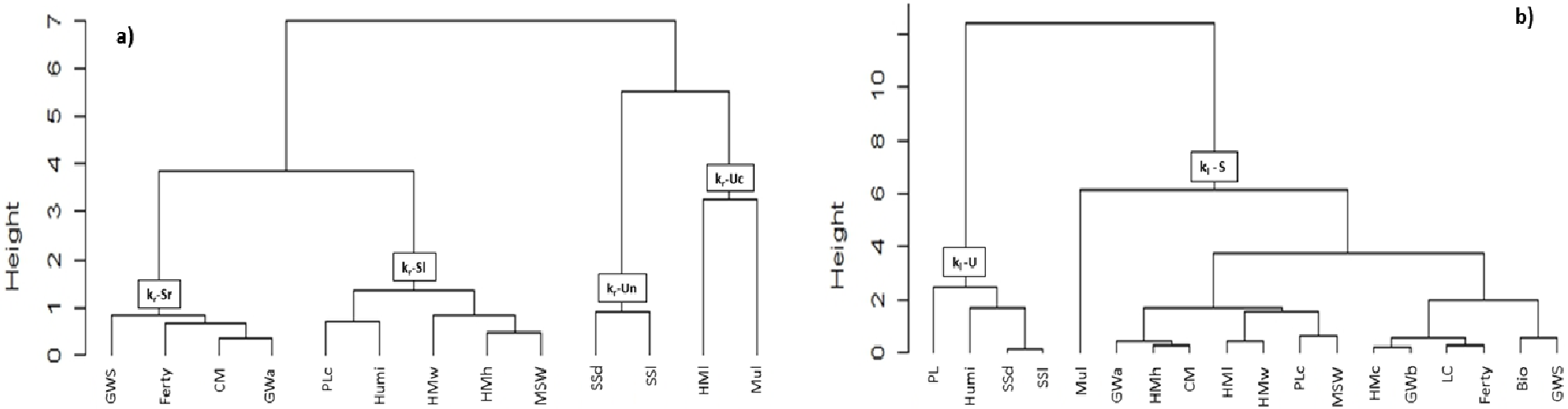


Figure 3. Dendrogram trees for k_l clustering (a), and k_r clustering (b).
In clusters names, S stands for stable, U for Unstable, l for labile, r for resistant. EOM labels: Dried Sewage Sludge (SSd), Green Waste compost BYS (from Bio Yvelines Service) (GWb), Green Waste and Sludge compost 2009 (GWS), Horse Manure, low straw content (HMl), Municipal Solid Waste compost (MSW), Horse manure high straw content (HMh), Horse manure with wood shavings (HMw), Horse manure composted (HMc), Cattle manure (CM), Dried Poultry Litter (PL), Poultry litter composted (PLc), Leaves composted (Lc), Green Wastes Sep (from SEPUR) (GWa), Mulch (Mul), Biowastes Compost (BIO), Limed Sewage Sludges (SSl), Dried Pig Slurry ("Humival"), Green waste and bone meals ("Fertylis")

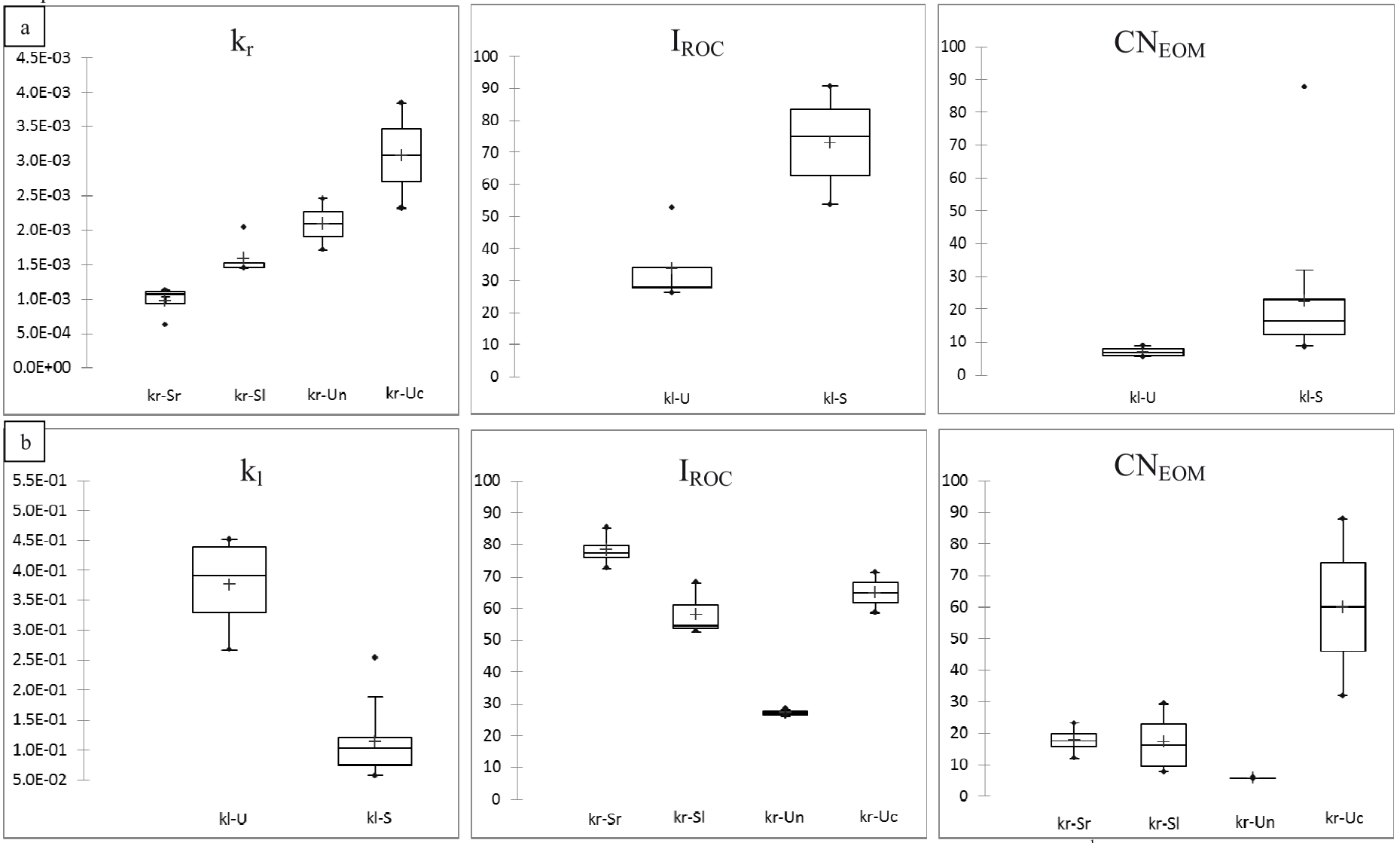


Figure 4. Characteristics of the EOM clusters for k_r parameterisation (a) and k_l parameterisation (b) in terms of k_{EOM} ($days^{-1}$), I_{ROC} (% of C_{EOM}) and CN_{EOM}. In cluster names, S stands for stable, U for Unstable, l for labile, r for resistant, n for rich in nitrogen and c for rich in carbon.

In order to allow/facilitate further k_{EOM} parameterisation, EOM were plotted on an CN_{EOM};I_{ROC} graph where the characteristics of EOM within each k_r and k_l cluster as defined above were drawn (Fig. 5). The k_l-Sl cluster included the k_r-sludge cluster and humival while k_l-Sr cluster included the other k_r clusters, except Humival. In our dataset, EOM for which k_r could not be calculated were near the k_r-Sr based on their I_{ROC} and CN_{EOM}, therefore, they were attributed the k_r value of the k_r-Sr cluster.

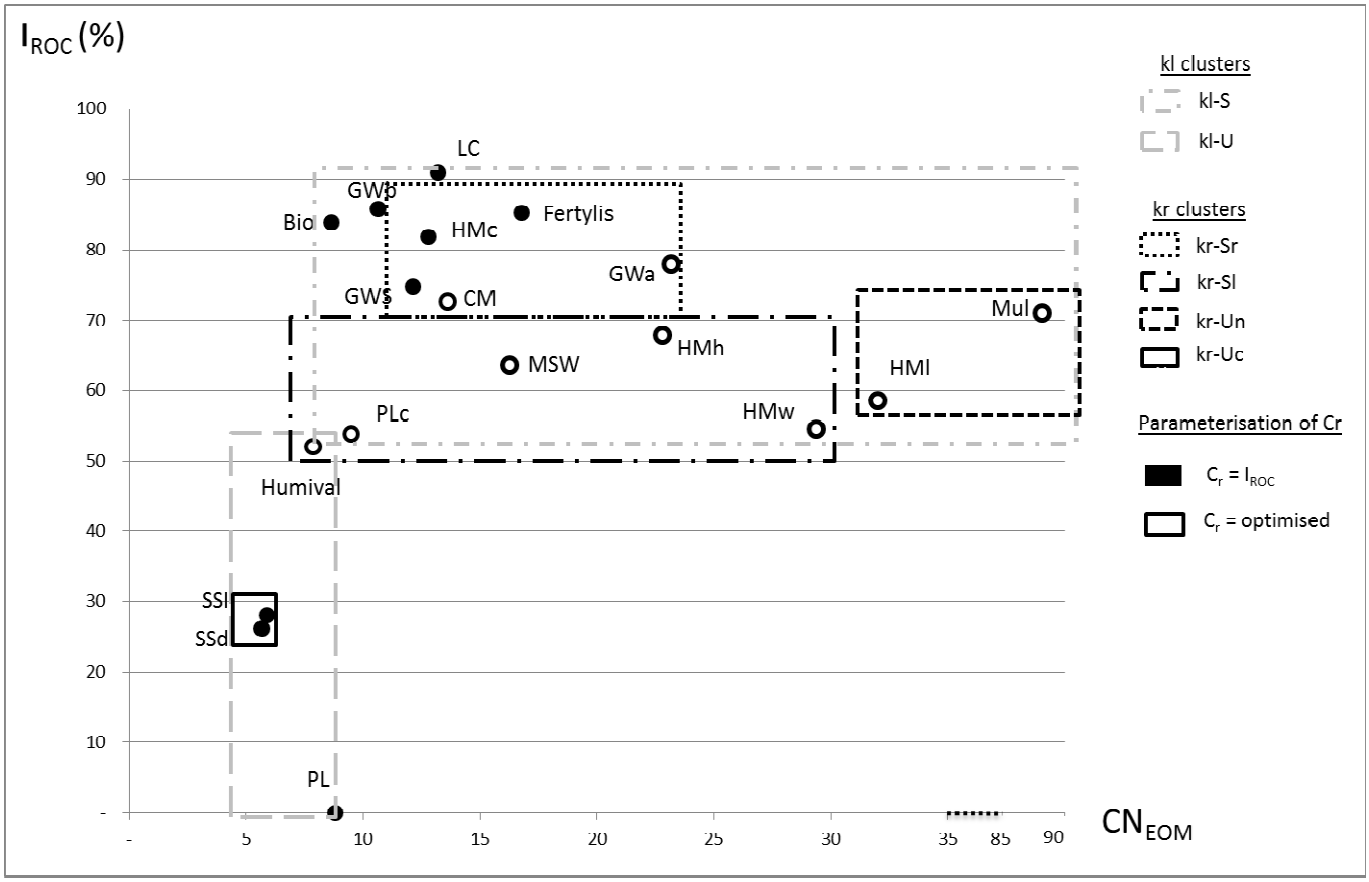


Figure 5. EOM parameterisation clusters.
In cluster names, S stands for stable, U for Unstable, l for labile, r for resistant, n for rich in nitrogen and c for rich in carbon. EOM labels defined table 2.

4.3. Simulation of EOM mineralisation

4.3.1. EOM mineralisation behaviours

The simulated kinetics of C and N mineralisation for all the EOM incubated in the QualiAgro soil and for the 4 tested methods of parameterisation are presented in Fig. 1S and results are summarised in Table 5. Four groups of mineralisation behaviour were distinguished (Fig. 1S). In Fig. 6 are presented mineralisation kinetics for one EOM from each group: HMc, MSW, HMh and SSl. Mineralisation values are given for 91 days in the following paragraph.

In the first group (example of HMc), the C and N mineralisation were very low with 10% of Exogenous Organic Carbon (EOC) and -10 to 10% of Exogenous Organic Nitrogen (EON) mineralised. This group of 6 EOM only included composted EOM.

The second group (example of MSW) was characterised by higher C mineralisation (around 30% of EOC) and little (-4% of EON for CM) to high N immobilisation at day 7 (-51% of EON for GWa). It could be considered as a transition group between the first and the third and included 2 composts and 1 cattle manure.

The third group was characterised by high C mineralisation (between 51 and 81%), and strong N immobilisation (-39 to -151%). This third group included the mulch and all manures but one. The 3 horse manures were included and only the cattle manure remained excluded from this group.

The fourth group was characterised by high C mineralisation (37 to 97%) and high N mineralisation (from 12 to 48%). It included 5 EOM: 2 sludges, 2 poultry litters and the slurry Humival.

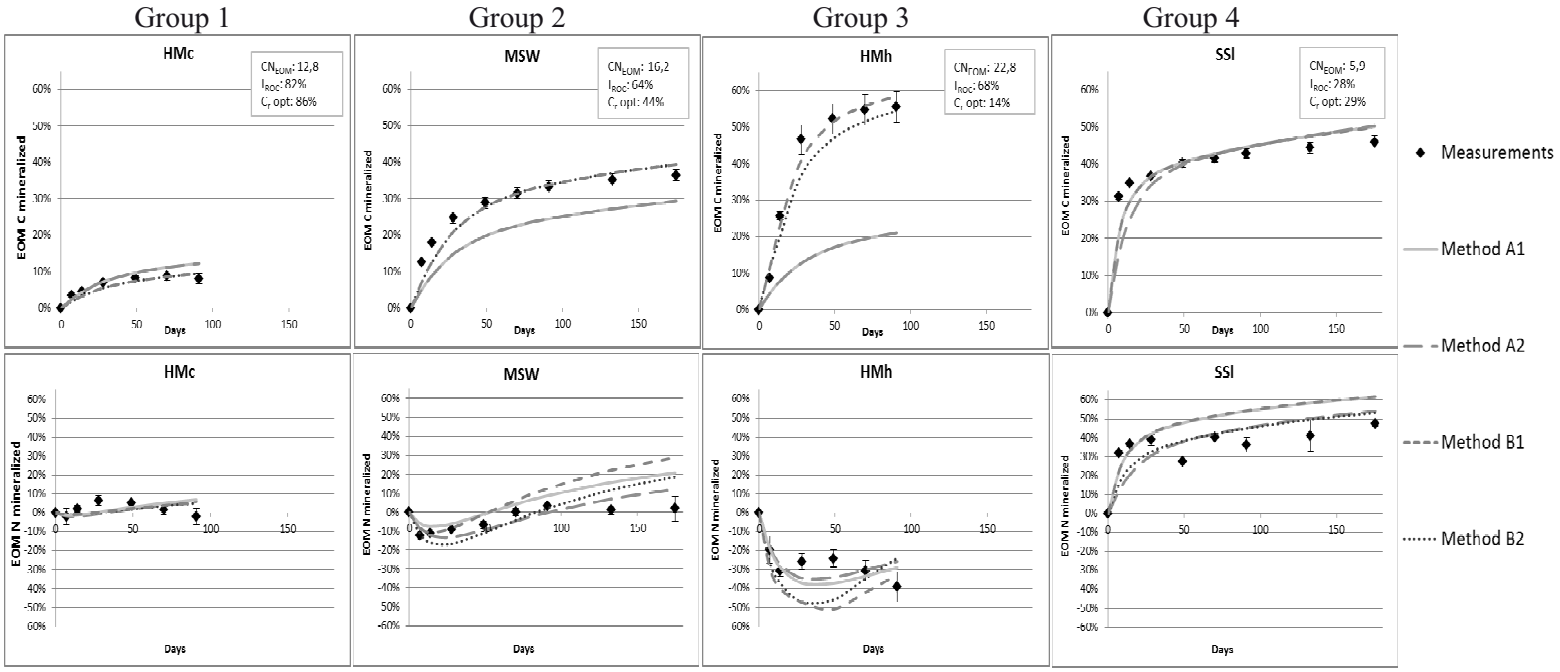


Figure 6. C and N EOM mineralisation curves; examples of group 1 (HMc, Horse Manure composted), group 2 (MSW, for Municipal Solid Waste), group 3 (HMh, Horse Manure with high amount of straw) and group 4 (SSl, for limed Sewage Sludge).
Method A and B refer to C_r parameterisation with: the use of I_{ROC} and the optimisation of C_r respectively, and method 1 and 2 refer to CN_l and CN_r parameterisation with the use of CN_{EOM} for both and the optimisation of CN_l respectively

4.3.2. Appropriateness of parameterisation methods

For EOM from the 1st group, C mineralisation visually seemed to be quite well simulated (Fig. 6 and 1S) whatever the resistant pool size parameterisation method. The resistant pool size estimated with I_{ROC} (method A) or optimised (method B1) little differed (less than 12% of absolute difference) (Table 5). The N mineralisation dynamics were quite well represented too. Optimising CN_l only changed CN_r by less than 1.1 compared to method 1 using CN_{EOM} for both pools.

For EOM from the second and third groups, C mineralisation was always underestimated with method A. The optimised resistant pool sizes were always lower than I_{ROC} by at least 19% (absolute) for CM up to 61% for Mulch. To better fit N mineralisation measurements, the optimisation of CN_l resulted in

higher CN_l for the 2nd group, slightly for CM (from 13.6 to 14.1) with larger differences for MSW (from 16.2 to 21.6) and GWa (from 23.1 to 39.7). For group 3, the optimisation did not converge for HMw and was not made for HMl since 100% of the carbon was in the labile pool. The CN_l decreased a little for HMh (from 22.8 to 20.5) and Mulch (from 87.9 to 82.2) without well-fitting the N dynamic for the Mulch. However, the simulated N immobilisation did not last enough and simulations showed re-mineralisation at the end of the experiment. Therefore, the CN optimisation seemed inefficient at improving the simulations.

In the fourth group, C mineralisation was well simulated for the sludges and Humival whatever the parameterisation methods, with a slight underestimation for Humival with method A, well corrected with method B. However, method A resulted in a strong underestimation for PLc also corrected with method B and for PL both methods led to underestimation of mineralisation. The C_r optimised was similar than the I_{ROC} for SSl (29% for 28%), decreased for SSd and Humival (from 26 to 17% and from 52 to 40%, respectively), and for PLc (from 54 to 13%). The model hardly simulated PL mineralisation dynamics because this EOM only contained a labile fraction with very high k_l. The CN_{EOM} optimisation allowed better N mineralisation simulations for sludges and slightly changed CN_l from 5.6 and 5.9 to less than 6.8 for SSd and SSl, respectively with the two methods A and B. For Humival, method B2 also changed a little CN_l (from 7.8 to 6.9) but surprisingly better simulations were obtained with method A2 with I_{ROC} and a CN_l changed to 6.0. For PLc, better simulations were obtained with B2 which implied consequent change in CN_l (from 9.5 to 12.3). The C/N ratios were not changed for PL since 100% of the matter was only as labile pool with both method A or B.

Finally, for the first group with composts, using I_{ROC} for C_r size and CN_{EOM} for both pool C/N ratios seemed to be adequate for parameterisation. For the second and third EOM groups composed with less stable compost and manures, optimisation was necessary to determine the resistant pool C_r. Using total CN_{EOM} could be acceptable for group 2, but for group 3 increasing CN_l was needed to better represent N dynamics. For group 4, I_{ROC} was adequate for sludges but underestimated C_r for PLc and slightly for Humival. Additionally, despite better simulations when optimising CN_l, changes in C/N ratios were quite small.

Table 5. EOM parameterisation NCSOIL tests results

Groups of EOM based on mineralisation behaviour	Group 1						Group 2			Group 3				Group 4				
EOM	Bio	GWb	HMc	LC	GWS	Fertylis	CM	MSW	Gwa	HMw	HMh	HMl	Mulch	SSd	SSl	PL	Humival	PLc
Resistant pool size																		
I_{ROC}	84%	86%	82%	91%	75%	85%	73%	64%	78%	55%	68%	59%	71%	26%	28%	0%	52%	54%
C_r optimised	92%	87%	86%	84%	87%	79%	54%	44%	45%	21%	14%	0%	10%	17%	29%	0%	40%	13%
Labile C/N ratio																		
CN_{EOM}	8.6	10.6	12.8	13.2	12.1	16.7	13.6	16.2	23.1	29.3	22.8	31.9	87.9	5.6	5.9	8.8	7.8	9.5
CN_l optimised (A2)	11.9	20.9	13.7	30.7	17.4	18.3	13.1	21.6	2674.4	30.2	67.4	232.7	40.3	6.3	6.8	8.8	6	10.5
CN_l optimised (B2)	9.6	21.7	13.1	21.7	16.2	17.9	14.1	20.3	39.7	NA	20.5	31.9	82.2	6.8	6.8	8.8	6.9	12.3
Resistant C/N ratio																		
CN_{EOM}	8.6	10.6	12.8	13.2	12.1	16.7	13.6	16.2	23.1	29.3	22.8	31.9	87.9	5.6	5.9	8.8	7.8	9.5
CN_r calculated (A2)	8.2	9.8	12.6	12.5	11	16.5	13.8	14.2	18.1	28.6	17.3	19.9	168.3	4.3	4.3	8.8	10.9	8.8
CN_r calculated (B2)	8.5	9.9	12.7	12.3	11.7	16.5	13.2	12.9	15.3	NA	74.7	31.9	104.2	3.2	4.3	8.8	10	3.8

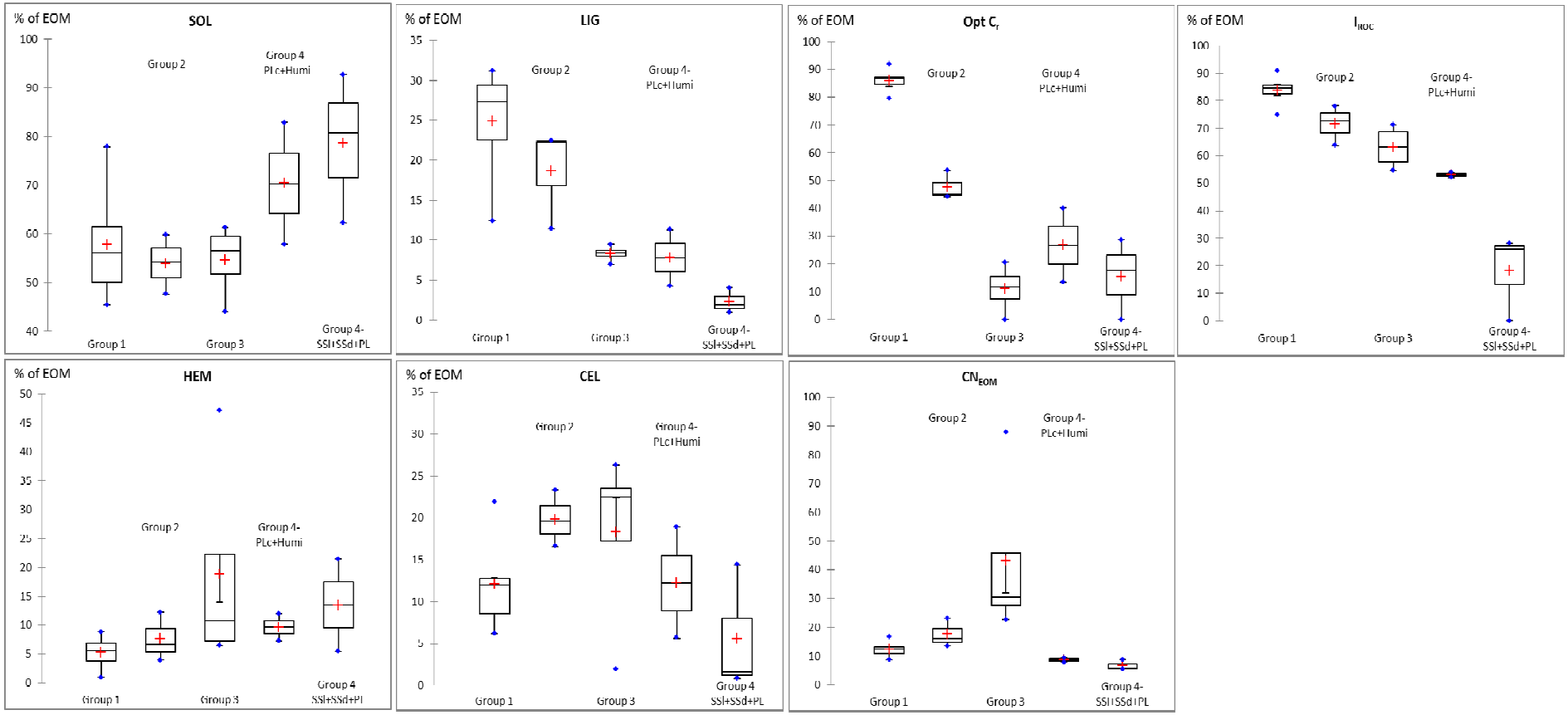


Figure 7. Characterisation of EOM groups with VS fractions and EOM parameters: Optimised C_r, I_{ROC} and CN_{EOM}. Group 2, 3 and 4-PLc+Humi are those for which the parameterisation method A1 is not suitable.

4.3.3. EOM characterisation requiring another parameterisation approach

Fig. 5 represents EOM according to their I_{ROC} and CN_{EOM}. Nine EOM needed a consequent C_r increase compared to their I_{ROC} (EOM from group 2 and 3 but also Humival and PLc from group 4). These 9 EOM corresponded to EOM with CN_{EOM} ratio higher than 20, and EOM with I_{ROC} ranging from 50 to 73. On the other hand, EOM requiring CN optimisation were also in this group but were not enough to define any trends of characteristics. EOM with either very high or very low I_{ROC} did not need optimisation of parameters to simulate correctly C and N mineralisation. These 9 types of EOM could also be distinguished from their analytical characteristics such as their Van Soest Fractions as presented in Fig. 7. Comparatively, they had high CEL (17.5 against 12.1 and 5.6 for 1st EOM group and group including the sludges and the poultry litter, respectively) and intermediate LIG (11.7 against 24.9 and 2.3).

For these 9 EOM, regressions between optimised C_r and analytical characteristics including Van Soest fractions were attempted. CM was always an outlier and was removed to build the regression. The best regression was found with HEM, CEL, LIG and CN_{EOM} with a R^2 of 0.967, R^2-adj of 0.923 and a P-value for the Fischer test of 0.015.

$C_r = 49{,}514 - 1{,}8228 \times HEM - 2{,}0345 \times CEL + 1{,}2705 \times LIG + 1{,}00891 \times CN_{EOM}$ [5]

With C_r standing for resistant pool size (% of C_{EOM}), HEM, CEL, LIG the Van Soest fractions (%) and CN_{EOM}, the total C/N ratio of EOM.

4.4. Use of Van Soest fractionation for parameters and group characterisation

Relationships between EOM simulation parameters (I_{ROC}, optimised C_r, CN_{EOM} and averaged cluster k_{EOM}), and VS fractions were observable on PCA results in Fig. 8 and in the correlations matrix (Table 6). The k_l was closely and positively related to SOL, and slightly and negatively related to CEL and LIG. The k_r was slightly related to the HEM fraction (positively) and slightly to LIG (negatively). The optimised C_r and I_{ROC} were positively related to LIG. CN_{EOM} was only weakly related to VS fractions but rather positively to CEL and HEM.

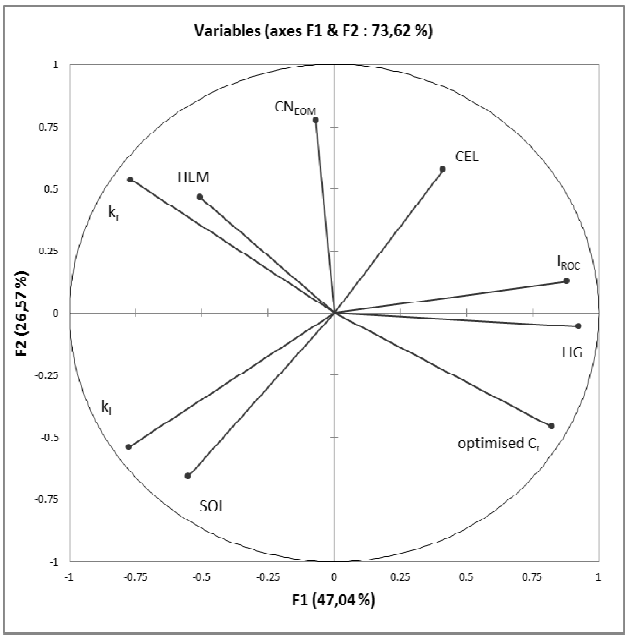


Figure 8. PCA with VS fractions (SOL, HEM, CEL and LIG) and NCSOIL-EOM parameters (k_r, k_l, optimised C_r, I_{ROC}, CN_{EOM}.

Table 6. EOM parameters and VS fractions correlations matrix

Variables	I_{ROC}	Opt C_r	CN_{EOM}	HEM	CEL	LIG	SOL	k_I	k_r
I_{ROC}	**1**	0.714	0.184	-0.356	0.220	0.772	-0.435	-0.821	-0.535
Opt C_r		**1**	-0.347	-0.546	-0.061	0.820	-0.158	-0.377	-0.809
CN_{EOM}			**1**	0.102	0.379	-0.145	-0.198	-0.340	0.578
HEM				**1**	-0.261	-0.407	-0.295	0.077	0.632
CEL					**1**	0.240	-0.583	-0.538	-0.162
LIG						**1**	-0.577	-0.635	-0.664
SOL							**1**	0.736	0.113
k_I								**1**	0.315
k_r									**1**

EOM groups based on mineralisation behaviours could also be distinguished based on Van Soest fractions, CN_{EOM} as showed in the PCA (Fig. 9). Indeed, EOM from group 4 were mostly characterised by high SOL fraction, except PLc. EOM from group 1 were mostly characterised by high LIG fraction, Fertylis being also characterised by large SOL fraction, probably due to animal bone meals addition to the green waste compost. EOM from group 2 were characterised by both high proportion of LIG and CEL, and group 3 by CEL and HEM.

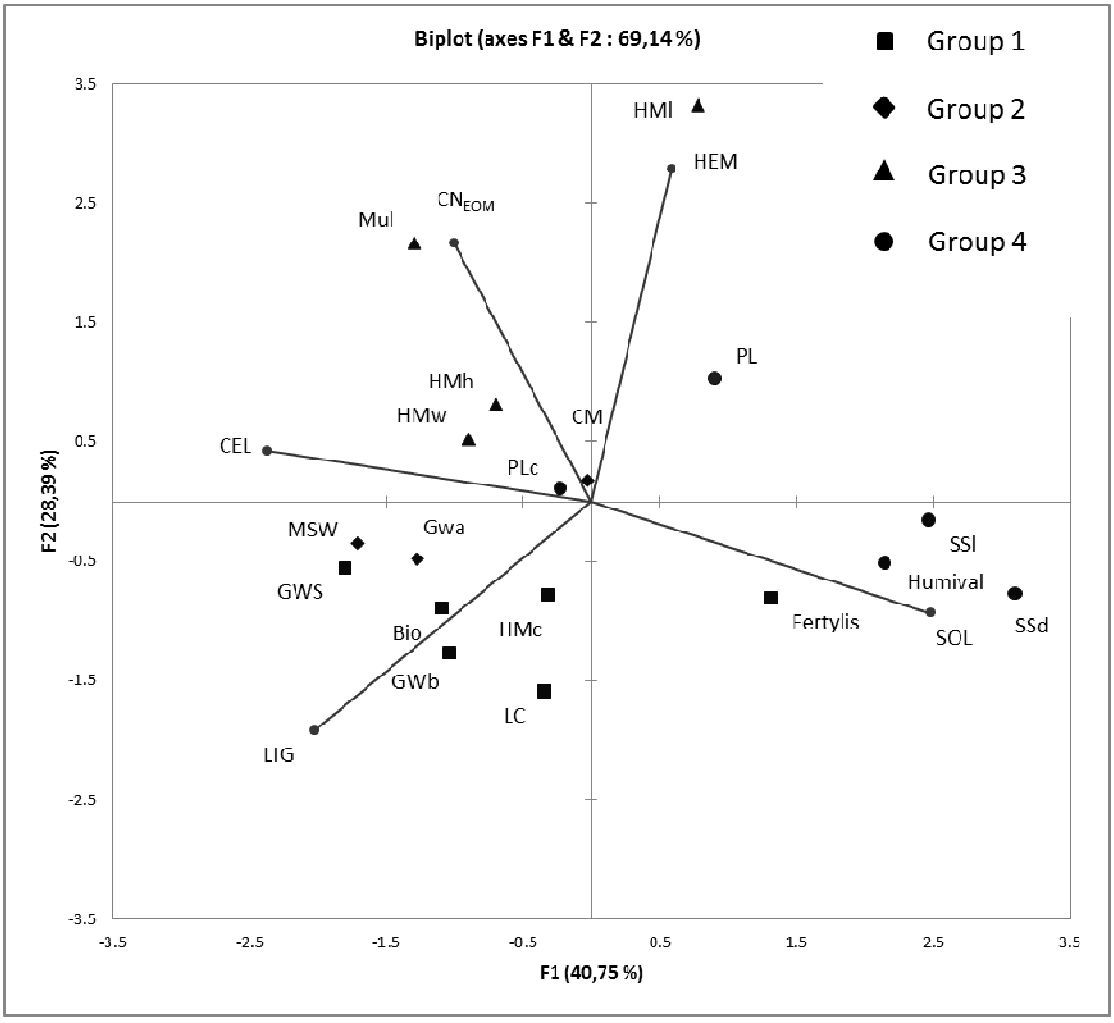


Figure 9. PCA representation of all EOM by group with CN_{EOM} and VS fractions

5. Discussion

5.1. Parameterisation methods

In this study, EOM parameterisation in the NCSOIL model was based on experimental kinetics of C and N mineralisation determined during time-consuming incubations. Parameterising k_{EOM} and especially k_r based on such rather short-term incubations was very sensitive to the data and small errors or variations in the experimental results could easily lead to inadequate parameters. We first decided to set the k_r based on calculated slopes on the end-period of the incubations rather than on fitted NLR parameters considering the entire experimental kinetics. However, we still faced some issues probably due to experimental uncertainties leading to negative slopes useless as parameters. Thuriès et al. (2002) also found that estimation of k_{EOM} were always less accurate than estimation of pool fractions size. Thus, the dataset of EOM and their characteristics was clustered and averaged k_{EOM} were calculated for each cluster, then attributed to each EOM of the clusters. This made also possible to attribute dynamic parameters for EOM without information of C and N mineralisation but based on their characteristics, avoiding time-consuming incubations.

Using the same NCSOIL model, Antil et al. (2011) determined k_{EOM} for 4 EOM similar as in our study and in the same order of magnitude, between 0.031 and 0.461 day^{-1} for k_l and between 5.2E-4 and 1.24E-3 day^{-1} for k_r which was a little lower than in our study. Beraud et al., (2005) found for two composted EOM similar k_r (1.19E-4 and 1.42E-4 day^{-1}) but lower k_l (around 0.024 day^{-1}).

The I_{ROC} indicator represented the stable fraction of EOM, potentially incorporated into SOM and was calculated as the proportion of non-degraded EOC when decomposition rate reached $0.02y^{-1}$ or 5E-5 d^{-1} (Lashermes et al., 2009). In this study, this indicator was used to determine the proportion of C_{EOM} in the resistant pool, which decomposition rates varied from 0.971 to 3.085E-3 d^{-1}. It seemed to be adequate for many stable EOM such as composts (Noirot-Cosson et al., 2016) or more surprisingly very unstable sludge and litter. However, I_{ROC} was not suitable to parameterise the C_r fraction for 8 EOM and a new regression was determined for these EOM. The total organic C/N ratio was kept for parameterisation of resistant and labile EOM fractions even when N mineralisation was not well simulated, mostly because of lack of experimental data to calculate different C/N ratios for both fractions.

Whatever the parameters, to determine a generic way for parameterisation, more EOM should be studied. Predicting EOM N fate based on easily available data or through classification or typology was already tried (Lashermes et al., 2010; Morvan et al., 2006) but doing it to parameterise a model was quite new.

Finally, the following parameters should be used for further simulations with NCSOIL: the optimised carbon pool sizes, the total organic C/N ratio (CN_{EOM}) for both EOM pools and averaged k_{EOM} for each cluster (Table 1S).

5.2. General scheme for relating model parameters to measured values

The stable fraction size C_r was positively and the k_r inversely related with the LIG fraction while only the k_l fraction was positively related with the SOL fraction. The LIG might be the most recalcitrant fraction in the EOM resistant pool, while SOL probably the most reactive OM in the EOM labile pool. Large SOL or LIG fraction traducing reactive or stable EOM, respectively is often admitted (Thuriès et al., 2002). However, the SOL fraction was not related with the size of the labile fraction of EOM, probably because this fraction is heterogeneous and changes during EOM treatment by composting (Peltre et al., 2010). The HEM fraction was positively related with the k_r, thus corresponding to more biodegradable resistant pool. Inversely, the CEL fraction was negatively related with the k_l meaning that high CEL fraction corresponded to EOM with less biodegradable labile pool. The HEM and CEL fractions more described the diversity of intermediate EOM from groups 2 and 3. Where they did not differ from their CEL fractions, large HEM fraction in group 3 was associated very with low C_r. Larger HEM fraction was related with high CN_{EOM}, an indicator used to parameterise both pools indifferently and explaining large N immobilisation observed with EOM for group 3. With a two pools model to describe EOM C and N dynamics, relating HEM and CEL fractions to a pool or another appeared not straightforward and more EOM should be studied for such purpose. Indeed, Thuriès et al. (2002) related simple model parameters to such fractions always based on multiple regressions concerning several biochemical fractions.

5.3. EOM types available in the "Plain of Versailles" region

Four groups of EOM were made based on C and N mineralisation kinetics. With more EOM types, Morvan et al. (2006) was able to distinguish 6 classes based on C and N mineralisation and to characterise them comparatively based on biochemical fractions. Our classes were in accordance to their results with a need to divide our groups 2 and 4. Indeed, our four groups showed similar C and N overall dynamics however all groups but group 1 showed intra-group heterogeneities even with our smaller set of 18 EOM.

These 4 groups could also be distinguished when represented with their optimised C_r and CN_{EOM} (Fig. 10). Such grouping could not be made based on k_{EOM} or size of resistant or labile pools since C and N mineralisation were in interaction with several parameters that could vary in different ways. Indeed, EOM from the homogeneous group 1 were characterised by identical k_{EOM} and similar pool sizes (Table 1S), while heterogeneities in C and N dynamics within the other groups were traduced by variable parameters within groups.

All EOM available or potentially available in the "Plain of Versailles" region have been studied, including (Fig. 10) highly decomposable EOM rich in N leading to high availability of N (group 4), highly degradable EOM with high CN_{EOM} ratio leading to strong N organisation (group 3), to very

stable compost (group 1), and with some intermediate EOM (group 2). Based on these groups characterisation, EOM may be inserted differently within technical itineraries. For instance, group 4 should be rather used as fertilisers because of large amount of available N (and probably phosphorus and potassium), i.e. applied close to crop N uptakes. Group 1 with very stable EOM should be applied as organic amendments, to increase SOM stocks. EOM in Group 3 despite their low ability for enhancing SOM, might cause N immobilisation prejudicial for crop production if applied in period corresponding to high crop N uptake. They could be interesting for preventing N leaching and should also be used as amendment.

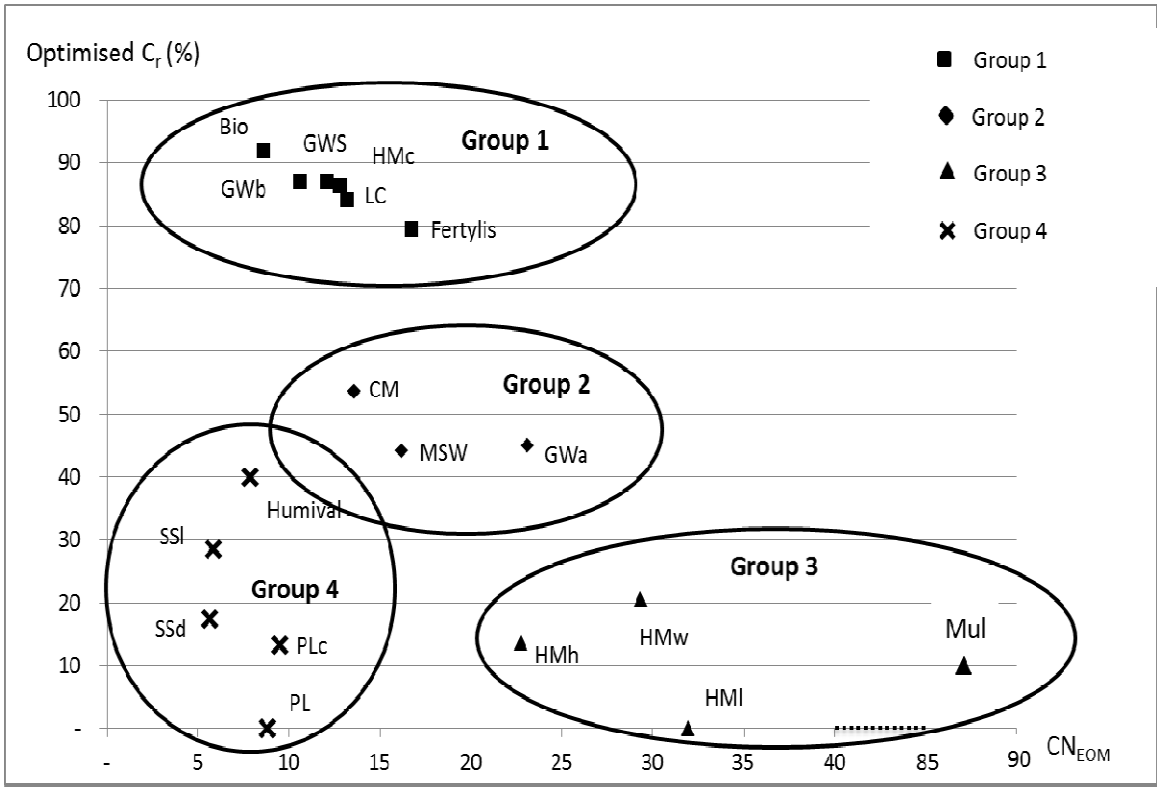


Figure 10. EOM groups distinguished by their C_r optimised and their CN_{EOM}.

6. Conclusion

This study aimed at characterising the interest for cropped soils of all available EOM in the "Versailles plain". Both the potential availability of N for crops (fertilising EOM) and their potential use as organic amendment to increase soil organic matter (amending EOM) were studied and related to their chemical and biochemical characteristics. Based on potential C and N mineralisation kinetics, four groups of EOM were defined: (group 1) stable compost, (2) more reactive compost and stable manures, (3) manures, all useable for organic amendments and (4) reactive EOM as sludges, litters, usable as organic fertilisers.

They were characterised by slow C and N mineralisation (group 1), high C mineralisation along with N immobilisation (group 3), intermediate C and N dynamics (group 2), and high C and N mineralisation (group 4).

The NCSOIL model was parameterised to simulate the kinetics of C and N mineralisation. In the NCSOIL model, EOM are modelled into two pools (resistant and labile) characterised by their relative sizes, their decomposition rates and their C/N ratios. The use of easily accessible indicators such as the I_{ROC} indicator to characterise the resistant pool size and measured CN_{EOM} to define the C/N ratio of both pools seemed appropriate for the groups 1 and 4. However, for some EOM characterised by intermediate I_{ROC}, high C/N ratios and also showing similar trends of biochemical compositions such as manures, both pool sizes and C/N ratios should be defined differently. A regression linking resistant pool size retrieved from optimisation and biochemical characteristics on 8 EOM was found significant and would thus need to be tested on an even higher number of EOM. Further investigations would also allow a better understanding of the N dynamics and CN parameterisation.

The soil type did not modify the mineralisation behaviour of EOM, thus same parameters could be used in NCSOIL whatever the soil in the studied area.

Despite issues about parameterisation and need of further investigation particularly for evaluation of C/N ratios of EOM pools, the NCSOIL model simulated quite well C and N mineralisation, allowing further study on C and N dynamics under field conditions after EOM application, using the crop model CERES-EGC in which the NCSOIL model has been integrated. All available EOM in the "Versailles plain" were parameterised in NCSOIL. This parameterisation will be used to simulate scenarios of EOM insertion in crop successions to evaluate the consequences in terms of carbon storage in soil and potential substitution of mineral N fertilisation.

Acknowledgments

This publication benefited financial and technical support from the French Environment and Energy Management Agency (ADEME) in the framework of the Prostock-Gessol3 project and from the LabeX BASC of Campus Paris Saclay in the framework of the SOCSENSIT project. The field experiment QualiAgro has started since 1998 within a cooperation between INRA-Véolia Environment Research and Innovation.

7. Supplementary data



Figure 1S. C and N EOM mineralisation curves for the 18 EOM. Method A and B refer to C_r parameterisation with: the use of I_{ROC} and the optimisation of C_r respectively, and method 1 and 2 refer to CN_l and CN_r parameterisation with the use of CN_{EOM} for both and the optimisation of CN_l respectively

Table 1S. Exogenous Organic Matter parameters retained for further simulations

EOM	CN_{EOM}	Opt C_r (% C_{EOM})	k_l (day^{-1})	k_r (day^{-1})
Biowaste Compost (Bio)	8.6	92	0.114	0.000971
Green Waste compost BYS (from Bio Yvelines Service) (GWb)	10.6	87	0.114	0.000971
Horse manure composted (HMc)	12.8	86	0.114	0.000971
Leaves composted (Lc)	13.2	84	0.114	0.000971
Green Waste and Sludge compost 2009 (GWS)	12.1	87	0.114	0.000971
Green waste and bone meals (“Fertylis”)	16.7	79	0.114	0.000971
Cattle manure (CM)	18.7	19	0.114	0.000971
Municipal Solid Waste compost (MSW)	16.1	44	0.114	0.00159
Green Wastes Sep (from SEPUR) (GWa)	23.1	45	0.114	0.000971
Horse manure with wood shavings (HMw)	29.3	21	0.114	0.00159
Horse manure high straw content (HMh)	22.8	14	0.114	0.00159
Horse Manure, low straw content (HMl)	31.9	0	0.114	0.00259
Mulch (Mul)	87.9	10	0.114	0.00259
Dried Sewage Sludge (SSd)	5.6	17	0.376	0.00209
Limed Sewage Sludges (SSl)	5.9	29	0.376	0.00209
Dried Poultry Litter (PL)	8.8	0	0.376	0.00209
Dried Pig Slurry (“Humival”)	7.8	40	0.376	0.00159
Poultry litter composted (PLc)	9.5	13	0.114	0.00159

CN_{EOM} stands for EOM organic C/N ratio, C_{EOM} stands for total organic carbon in EOM.

8. References

AFNOR, 1995. Norme NF ISO 10694. Qualité du sol - Dosage du carbone organique et du carbone total après combustion sèche, Qualité des sols, vol. 1. AFNOR, Paris.

AFNOR, 2003. NF X 31-107. Qualité du sol – Détermination de la distribution granulométrique des particules du sol – Méthode à la pipette. AFNOR, Paris.

AFNOR, 2005. NF ISO 10390. Qualité du sol - Détermination du pH. AFNOR, Paris.

AFNOR, 2009a. Norme XP U 44-162. Amendements organiques et supports de culture - Caractérisation de la matière organique par fractionnement bio-chimique et estimation de sa stabilité biologique. AFNOR, Paris.

AFNOR, 2009b. Norme XP U 44-163. Amendements organiques et supports de culture - Caractérisation de la matière organique par la minéralisation potentielle du carbone et de l'azote. AFNOR, Paris.

AFNOR, 2011. NF EN ISO 13039. Amendements du sol et supports de culture - Détermination de la matière organique et des cendres. AFNOR, Paris.

AFNOR, 2014. NF EN ISO 10693. Qualité du sol - Détermination de la teneur en carbonate - Méthode volumétrique. AFNOR, Paris.

Antil, R.S., Bar-Tal, A., Fine, P., Hadas, A., 2011. Predicting Nitrogen and Carbon Mineralization of Composted Manure and Sewage Sludge in Soil. Compost Sci. Util. 19, 33–43.

Barak, P., Molina, J.A.E., Hadas, A., Clapp, C.E., 1990. Optimization of an ecological model with the Marquardt algorithm. Ecol. Model. 51, 251–263.

Beraud, J., Fine, P., Yermiyahu, U., Keinan, M., Rosenberg, R., Hadas, A., Bar-Tal, A., 2005. Modeling Carbon and Nitrogen Transformations for Adjustment of Compost Application with Nitrogen Uptake by Wheat. J. Environ. Qual. 34, 664.

Burger, M., Venterea, R.T., 2008. Nitrogen Immobilization and Mineralization Kinetics of Cattle, Hog, and Turkey Manure Applied to Soil. Soil Sci. Soc. Am. J. 72, 1570.

Chadwick, D.R., John, F., Pain, B.F., Chambers, B.J., Williams, J., 2000. Plant uptake of nitrogen from the organic nitrogen fraction of animal manures: a laboratory experiment. J. Agric. Sci. 134, 159–168.

Chalhoub, M., Garnier, P., Coquet, Y., Mary, B., Lafolie, F., Houot, S., 2013. Increased nitrogen availability in soil after repeated compost applications: Use of the PASTIS model to separate short and long-term effects. Soil Biol. Biochem. 65, 144–157.

Ciais, P., Wattenbach, M., Vuichard, N., Smith, P., Piao, S.L., Don, A., Luyssaert, S., Janssens, I.A., Bondeau, A., Dechow, R., Leip, A., Smith, P., Beer, C., Van Der Werf, G.R., Gervois, S., Van Oost, K., Tomelleri, E., Freibauer, A., Schulze, E.D., Carboeurope Synthesis Team, 2010. The European carbon balance. Part 2: croplands. Glob. Change Biol. 16, 1409–1428.

Corbeels, M., Hofman, G., Cleemput, O.V., 1999. Simulation of net N immobilisation and mineralisation in substrate-amended soils by the NCSOIL computer model. Biol. Fertil. Soils 28, 422–430.

Gabrielle, B., Da-Silveira, J., Houot, S., Francou, C., 2004. Simulating Urban Waste Compost Effects on Carbon and Nitrogen Dynamics Using a Biochemical Index. J. Environ. Qual. 33, 2333.

Gabrielle, B., Da-Silveira, J., Houot, S., Michelin, J., 2005. Field-scale modelling of carbon and nitrogen dynamics in soils amended with urban waste composts. Agric. Ecosyst. Environ. 110, 289–299.

Gabrielle, B., Menasseri, S., Houot, S., 1995. Analysis and Field Evaluation of the Ceres Models Water Balance Component. Soil Sci. Soc. Am. J. 59, 1403–1412.

Hadas, A., Doane, T.A., Kramer, A.W., Van Kessel, C., Horwath, W.R., 2002. Modelling the turnover of 15N-labelled fertilizer and cover crop in soil and its recovery by maize. Eur. J. Soil Sci. 53, 541–552.

Hadas, A., Kautsky, L., Goek, M., Erman Kara, E., 2004. Rates of decomposition of plant residues and available nitrogen in soil, related to residue composition through simulation of carbon and nitrogen turnover. Soil Biol. Biochem. 36, 255–266.

Hadas, A., Molina, J. a. E., 1993. Simulation of nitrogen assimilation by heterotrophic soil microbial biomass. Physiol. Plant. 89, 664–668.

Haskett, J.D., Molina, J.A.E., Clapp, C.E., 1986. User's manual for NCSOIL, version 1.0. Dep. of Soil Science, University of Minnesota, Saint Paul, MN.

Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuries, L., Chaussod, R., Guillotin, M.L., Lineres, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S., 2009. Indicator of potential residual carbon in soils after exogenous organic matter application. Eur. J. Soil Sci. 60, 297–310.

Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuriès, L., Chaussod, R., Guillotin, M.L., Linères, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S., 2010. Typology of exogenous organic matters based on chemical and biochemical composition to predict potential nitrogen mineralization. Bioresour. Technol. 101, 157–164.

Marmo, L., Feix, I., Bourmeau, E., Amlinger, F., Bannick, C.G., De Neve, S., Favoino, E., Gendebien, A., Gibert, J., Givelet, M., Leifert, I., Morris, R., Rodriguez Cruz, A., Ruck, F., Siebert, S., Tittarelli, F., 2004. Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection, vol. 3, taskegroup 4 Exogenous Organic Matter.

Molina, J. a. E., Clapp, C.E., Shaffer, M.J., Chichester, F.W., Larson, W.E., 1983. NCSOIL, A Model of Nitrogen and Carbon Transformations in Soil: Description, Calibration, and Behavior. Soil Sci. Soc. Am. J. 47, 85–91.

Morvan, T., Nicolardot, B., 2009. Role of organic fractions on C decomposition and N mineralization of animal wastes in soil. Biol. Fertil. Soils 45, 477–486.

Morvan, T., Nicolardot, B., Péan, L., 2006. Biochemical composition and kinetics of C and N mineralization of animal wastes: a typological approach. Biol. Fertil. Soils 42, 513–522.

Nicolardot, B., Molina, J.A.E., Allard, M.R., 1994. C and N fluxes between pools of soil organic matter: Model calibration with long-term incubation data. Soil Biol. Biochem. 26, 235–243.

Noirot-Cosson, P.E., Vaudour, E., Gilliot, J.M., Gabrielle, B., Houot, S., 2016. Modelling the long-term effect of urban waste compost applications on carbon and nitrogen dynamics in temperate cropland. Soil Biol. Biochem. 94, 138–153.

Parnaudeau, V., Génermont, S., 2014. Valeur fertilisante azotée et pertes azotées influençant la valeur fertilisante. pp 369-396. In: Valorisation Des Matières Fertilisantes D'origine Résiduaire Sur Les Sols à Usage Agricole Ou Forestier. Impacts Agronomiques, Environnementaux, Socio-économiques. Houot et Al., 2014. Expertise Scientifique Collective. INRA-CNRS-Irstea. Synthèse 113pp (http://institut.inra.fr/Missions/Eclairer-Les-decisions/Expertises/Toutes-Les-actualites/Expertise-Mafor-Effluents-Boues-et-Dechets-Organiques#).

Parnaudeau, V., Nicolardot, B., Pagès, J., 2004. Relevance of Organic Matter Fractions as Predictors of Wastewater Sludge Mineralization in Soil. J. Environ. Qual. 33, 1885–1894.

Peltre, C., Christensen, B.T., Dragon, S., Icard, C., Katterer, T., Houot, S., 2012. RothC simulation of carbon accumulation in soil after repeated application of widely different organic amendments. Soil Biol. Biochem. 52, 49–60.

Peltre, C., Dignac, M.-F., Derenne, S., Houot, S., 2010. Change of the chemical composition and biodegradability of the Van Soest soluble fraction during composting: A study using a novel extraction method. Waste Manag. 30, 2448–2460.

Stratton, M.L., A. V. Barker, Rechcigl, J.E., 1995. Compost. Soil Amend. Environ. Qual. 249–309.

Thuriès, L., Pansu, M., Larré-Larrouy, M.-C., Feller, C., 2002. Biochemical composition and mineralization kinetics of organic inputs in a sandy soil. Soil Biol. Biochem. 34, 239–250.

Van Soest, P.J., Wine, R.H., 1967. Use of detergents in the analysis of fibrous feeds. IV. Determination of plant call-wall constituents. J. Assoc. Off. Anal. Chem. 50, 50–55.

Ward, J.H., 1963. Hierarchical Grouping to Optimize an Objective Function. J. Am. Stat. Assoc. 58, 236–244.

Ce deuxième article nous a permis de caractériser les PRO disponibles sur le territoire notamment grâce à des mesures de minéralisation de carbone et d'azote réalisées lors d'incubations de sol et de PRO. Nous avons montré que quel que soit le sol avec lequel les PRO ont été incubés, le paramétrage PRO obtenu n'est que très peu impacté, car l'effet sol sur ces paramètres est largement négligeable devant l'effet PRO. Ainsi, les incubations de tous les PRO faites avec un seul sol ont permis un paramétrage NCSOIL des PRO qui est utilisable pour des simulations dans d'autres contextes de sol.

Nous avons aussi testé l'utilisation d'indicateurs pour paramétrer ces PRO, plus facile à obtenir que leur optimisation basée sur des mesures de minéralisations fastidieuses. Par exemple, les constantes de décompositions déterminées rapidement grâce à des classifications réalisées dans l'article 2 peuvent être utilisées pour des simulations NCSOIL. D'autres indicateurs ont été utilisés, comme l'I_{ROC} pour déterminer des tailles de pools et le C/N organique total pour le C/N de chacun des pools. Ceux-ci ont permis de bonnes simulations NCSOIL des minéralisations C et N mesurées pour certains PRO comme les composts stables ou les PRO fertilisants, alors que l'optimisation des paramètres a été nécessaire pour d'autres comme pour les fumiers par exemple. Dans les cas où les indicateurs n'étaient pas adéquats, d'autres ont été proposés basés sur les paramètres optimisés.

Après avoir calibré le modèle CERES-EGC dans le chapitre 2, puis paramétré tous les PRO du territoire de la Plaine de Versailles dans ce chapitre 3, les impacts de scenarios d'apports de PRO au champ peuvent enfin être explorés par le biais de simulations CERES-EGC.

Chapter 4. Regional-scale scenarios of organic amendment use on cropped soils: impacts on soil organic carbon stocks and substitution of mineral fertilisation simulated with the CERES-EGC crop model

P.E. NOIROT-COSSON[1]*, E. VAUDOUR[1], C. AUBRY[2], J.M. GILLIOT[1], B. GABRIELLE[1], S. HOUOT[1]

[1]UMR ECOSYS, INRA, AgroParisTech, Université Paris-Saclay, 78850, Thiverval-Grignon, France

*Email: Emmanuelle.Vaudour@agroparistech.fr

[2] UMR SAD-APT, INRA, AgroParisTech, Université Paris-Saclay, F-75231 Paris cedex 05, France

1. Abstract

The insertion of Exogenous Organic Matter (EOM) in the fertilisation of crops may contribute to soil C sequestration, enhance Soil Organic Matter (SOM) benefits and partially substitute mineral fertilisers. The CERES-EGC model along with spatially-explicit data on soils and cropping systems were used to predict the impacts of application scenarios of a large panel of available or potentially available EOM on Carbon Storage (CS) and Nitrate Leaching (NL) at a regional scale (221 km²) in the Versailles plain. Constructions of EOM application scenarios were limited by their phosphorus and nitrogen inputs in order to prevent from pollution. The potential of synthetic nitrogen to be substituted by EOM applications and SOM increase was assessed after implementing a Nitrogen Savings (NS) calculation in CERES-EGC. The relative weight of CS, NS and NL regional drivers were assessed by means of a factorial analysis procedure. The largest CS were found in the calcaric cambisol and the retic haplic luvisol while the arenic cambisol resulted in the highest NL. The amending EOM could be applied in large quantities and allowed CS of up to 1.1 t C ha^{-1} yr^{-1} for a green compost while only 0.3 t C ha^{-1} yr^{-1} for a horse manure. A bio-waste compost reached CS efficiency of 63% of the carbon applied, while horse manures did not exceed 15%. Applications of dried sewage sludge and of green composts allowed NS of up to 74 kg N ha^{-1} yr^{-1}. The NS efficiencies were higher with only one application in the crop succession. They were the highest on maize with fertilising EOM such as dried sewage sludge and poultry litter, reaching 99% of N applied with EOM and on rapeseed for manures or municipal solid waste compost, up to 90%. NL also depended on EOM amendment, in an opposite way to NS: the higher the NS, the lower the NL. The highest NL, of up to 47 kg N ha^{-1} yr^{-1}, were caused by large application of the municipal solid waste compost and an horse manure, while with fertilising EOM such the dried sewage sludge, NL were lower than 11 kg N ha^{-1} yr^{-1}.
This study enabled to assess the potential impacts of various scenarios of EOM uses on crop plots of the studied region, and will be further used to optimise the potential use of EOM at a regional scale.

Key-words: exogenous organic matter, soil carbon storage, nitrogen saving and leaching, spatially-explicit simulations, regional scale

2. Introduction

Soil is known to be the biggest reservoir of terrestrial carbon (C) (Batjes, 1996; Lal et al., 2004) and is expected to contribute to the mitigation of climate change through its sequestering of C (Paustian et al., 1997; Smith et al., 2008; Ciais et al., 2010). The soil organic carbon (SOC) pool is also a key driver of soil quality affecting its biological, chemical and physical functions (Jandl et al., 2014). A decrease in SOC content (TOC) has been recorded in regions with intensive farming systems and has become a real threat for agricultural soil sustainability (Ciais et al., 2010), leading to soil fertility reduction and enhanced soil erosion or compaction risks (Lal et al., 2004). SOC evolution is related to management practices (Paustian et al., 1997; Sleutel et al., 2007 ; Smith et al., 2008) and particularly to manure or crop residues inputs (Sleutel et al., 2006a; Goidts and van Wesemael, 2007). Peri-urban areas cropped with cash annual crops and with reduced livestock such as in the large Paris area (Petit and Aubry, 2014) are likely to face an even more drastic decrease of SOC, as the incorporation of C into soil once favoured by the livestock effluent has been disrupted for very long, as early as the 1950s. However, near cities like Paris, peri-urban agriculture can benefit from organic wastes derived from urban activities. Exogenous Organic Matter (EOM) i.e. organic residues from cities, industries or agriculture, can be recycled and applied on crop fields to increase soil C and Nitrogen (N) availability (Edmeades, 2003; Marmo et al., 2004; Houot et al., 2014) and thus enable carbon storage (CS) and mineral fertiliser substitutions N, P and K usually issued from non-renewable sources (K and P) or synthetised from an industrial cost-energy process. Nevertheless, EOM use can also lead to environmental costs such as Phosphorus (P) and N pollutions depending on the amendment practice (Rosen and Allan, 2007). For instance P losses through runoff and N through leaching can reach underground and surface water leading to ground water pollution and eutrophication of surface waters (Ansari et al., 2011). N can also be lost in the atmosphere in different forms such as N_2O, NO_x, NH_3, being either Greenhouse Gases (GHG) participating to global warming, or hazardous for human health or disturbing ecosystems (Mosier, 2001).

Benefits and costs of EOM applications may vary first depending on EOM characteristics and second on other drivers and their interactions such as soil type, crop production system and fertilisation technique, climatic condition. EOM characteristics have been already studied for a large panel of EOM types (Morvan et al., 2006; Morvan and Nicolardot, 2009; Thuriès et al., 2002; Lashermes et al., 2010), however these studies only considered lab characterisation and did not take the other above-mentioned drivers of EOM behaviour into account and thus could not predict EOM impacts when used on crop fields. Mechanistic crop models such as DAISY (Jensen et al., 1997), CERES-EGC (Gabrielle and Kengni, 1996; Gabrielle et al., 2004, 2006), STICS (Brisson et al., 2009), integrate all these drivers together and may predict EOM use impacts on crop production, soil C and N cycle for a wide range of scenarios of EOM use (Gabrielle et al., 2004). Indeed, these soil-water-atmosphere-plant crop

models already showed good ability to predict EOM impacts on C and N cycles at the field scale for several long-term field monitoring experiments (Franko et al., 1997; Kelly et al., 1997; Bruun et al., 2003; Gabrielle et al., 2005; Bhattacharyya et al., 2007; Chalhoub et al., 2013).

At the regional scale, EOM degradation drivers may vary with the context of soil and cropping system and various EOM types may be available; hence the need to explore and assess the impacts of the various EOM uses for the various combinations of soil and crop production systems within the region: in this, spatially integrated studies are essential for policy makers. The relevance of regional/landscape scale for addressing agri-environmental issues which involve energy and matter fluxes across space and time is increasingly recognised (Paustian et al., 1997; Lettens et al., 2005; Viaud et al., 2010). Using dynamic models that can account for various EOM types and amendment scenarios along with spatially-explicit data to estimate potential regional changes of SOC was one of the recommendations by the IPCC guidelines (IPCC, 2006).

Few authors carried out long term spatially explicit simulations of SOC dynamics over croplands coupling a deterministic SOC model such as the DNDC model (Sleutel et al., 2006b; Xu et al., 2012), i-Century (Tornquist et al., 2009), Roth-C (Mondini et al., 2012; Yokozawa et al., 2010) with Geographic Information System (GIS). Among the spatially-explicit studies simulating C dynamics using SOC models, few considered EOM amendment practice. Moreover, these studies were generally focused on C sequestration at national or sub-national scales and considered a limited set of management scenarios along with other management practices. Those accounting for EOM amendment practices did not consider a variety of existing EOM types and amendments. Most spatial-temporal studies on EOM amendment considered SOC evolution and not the environmental impacts of EOM uses in terms of P and Nitrate Leaching (NL), or even potential synthetic Nitrogen Savings (NS) that contributes to mitigate indirect GHG emissions (Pellerin et al., 2013). In particular, while N fixation through industrial processes constitutes 2% of world energy consumption (Sutton et al., 2013), the impact of NS on GHG emissions offsetting is never considered in studies on EOM practice which are solely focused on the CS-related impacts. This study was original in studying EOM amendment taking these environmental impacts into account, which, as far as we know, was never done previously, especially in a spatially-explicit manner.

Spatially explicit simulations require the gathering of a set of geographical layers including soil types, initial SOC contents and crop successions on each considered spatial unit of simulation. This study aimed at assessing realistic scenarios of EOM use in terms of CS, NS and NL, combining the use of the crop model CERES-EGC, a set of GIS geographical layers, and averaged climatic data over a 20 y-period at the scale of a 221 km² peri-urban region. The scenarios of EOM amendment practice that were considered accounted for potential mineral P and N substitutions and regulation constraints for N applications. The objective was to assess the overall agronomical and environmental impacts of the considered scenarios.

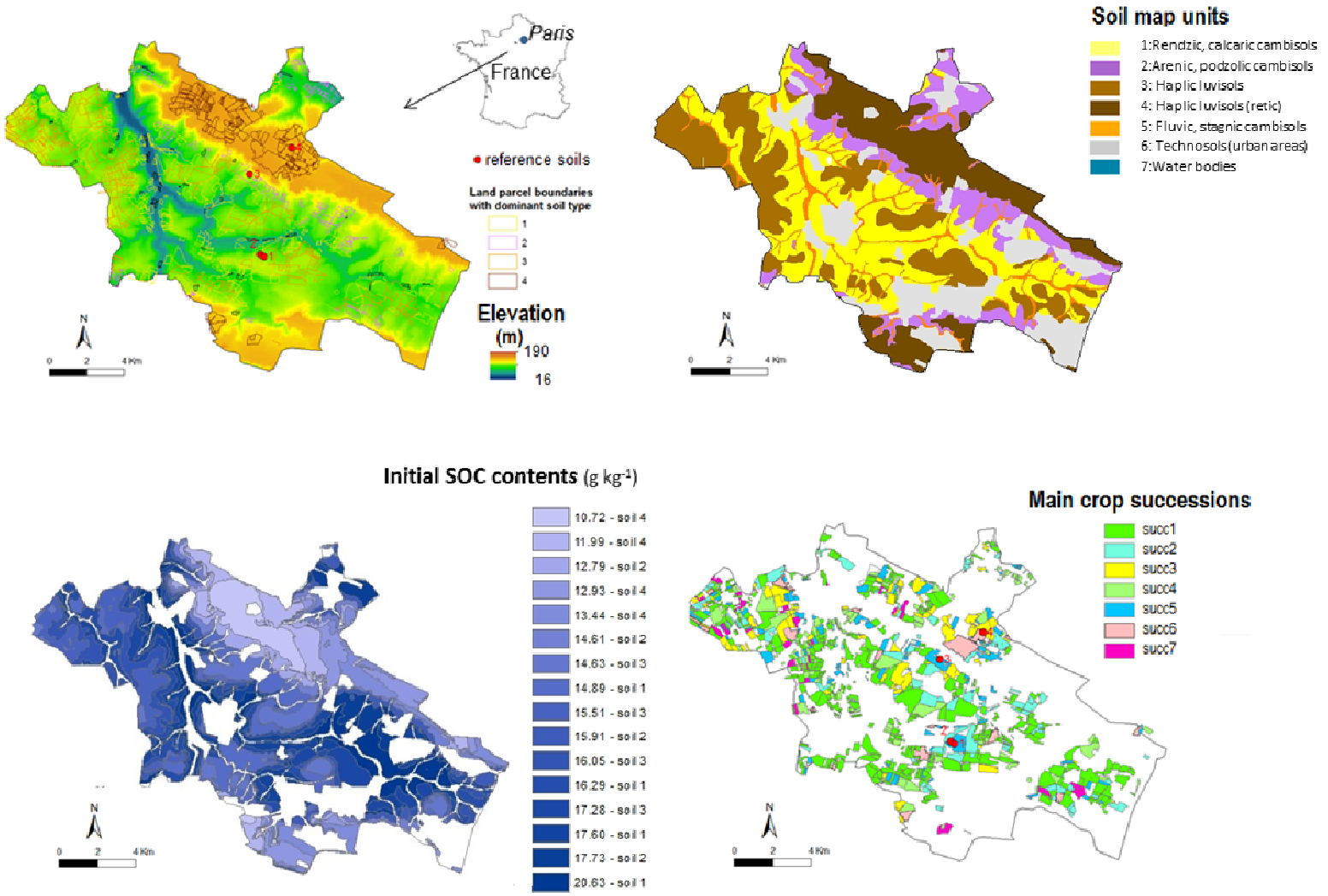


Figure 1. Geographical setting of the study zone: main crop successions and their land parcel boundaries, soil types, elevation and initial SOC contents. Soil types: 1, calcaric cambisols; 2, arenic cambisols; 3; haplic luvisols; 4, haplic luvisols (retic). Crop successions: succ1, winter rape+3 cereals; succ2, maize+3 cereals; succ3, winter rape-cereal-maize-cereal; succ4, winter rape+2 cereals; succ5, maize+2 cereals; succ6, maize+1 cereal+fodder/forage/legume+2 cereals; succ7, winter rape + cereals+fodder/forage/legume+2 cereals

3. Materials and methods

3.1. Study Area

The study zone was the "Versailles plain and Alluets plateau" area (48°46'-48°56' N; 1°50'-2°07' E), extending over 22 145 ha at about 30 km west of Paris, characterised by a semi-oceanic climate (Fig. 1). With a cropped surface of 9907 ha, this agricultural peri-urban region has low topographic contrast: cropping systems are distributed across the NW-SE axis characterised by intermediate plateaus covered with Quaternary loessic silts at about 120 m of elevation, surrounded at both their northern and southern edges by higher plateaus reaching 160-180 m (Vaudour et al., 2013, 2014). Quaternary loess deposits and loessic colluvium leave a mark on all landforms, particularly plateaus where they have been transformed into haplic luvisols (Crahet, 1992; Vaudour et al., 2013, 2014) according to the WRB classification (World Reference Base for Soil Resources, 2014). Calcaric, rendzic cambisols develop on plateau flanks, while colluvic cambisols originate from loessic colluvium and/or sands and may overly marl or chalk on lower slopes and at valley bottoms. The main crop successions are composed of winter Rapeseed (R), winter Wheat (W), spring Barley (B), Maize (M) or sometimes dry Pea (P) or fodder crops, and mostly managed with conventional tillage practices: late fall ploughing in November-December is followed by chisel in March then seedbed preparation for spring cereals (Vaudour et al., 2015). Spring barley and pea are usually sown in March, followed by maize in April, rapeseed in August and winter wheat in October.

3.2. Set of geographical layers included in the modelling

A set of several data layers was gathered and structured into ArcMap10.3® (Fig. 1).

Information on soil properties and spatial distribution of soil types was extracted from an available conventional soil map elaborated at the 1:50000 scale (Crahet, 1992) comprising 43 soil map units characterised from 2321 cores, which was simplified into 5 main soil map units. One of these soil map units was located near rivers where EOM application is restricted and pastures predominate, therefore the area concerned by this soil map unit was excluded. In addition to this map, detailed soil properties of 4 soil profiles representing each of the 4 soil map units were collected from soil pits (Fig. 1) or partially collected from previous studies (Laville et al., 2014): (1) a calcaric cambisol, (2) an arenic cambisol, (3) a haplic luvisol and (4) a haplic luvisol (retic) according to the WRB (2014). Morphological descriptions, horizon depths were recorded and bulk densities, soil C/N ratios and pH were determined (Table 1). Soil water content at wilting point (LL), field capacity (CC) and saturation (SAT) were determined by laboratory measurements of soil retention curves which were taken at at pF -1513, -33.9 and 0 kPa for LL, CC and SAT, respectively. Pedotransfer functions (Gabrielle et al., 2002) were used to calculate hydrodynamic parameters such as saturated hydraulic conductivity (Ks), except for haplic luvisol for which Ks were adjusted (Chalhoub, 2010).

Table 1. Characteristics and CERES-EGC parameters of the 4 main soil types
Ks stands for conductivity at saturation. Hwp, Hfc, Hsat are soil volumic humidity at the wilting point, field capacity and saturation, respectively. SL stands for silt loam, S for silt, CL for clay loam, LS for Loamy sand, SCL for sandy clay loam.

Depth (cm)	Texture	Hwp ($m^3\ m^{-3}$)	Hfc ($m^3\ m^{-3}$)	Hsat ($m^3\ m^{-3}$)	Ks ($cm\ d^{-1}$)	Bulk density ($g\ cm^{-3}$)	Clay ($g\ kg^{-1}$)	Sand ($g\ kg^{-1}$)	Calcareous content ($g\ kg^{-1}$)	C/N	pH
Rendzic Calcaric cambisol (2531 ha)											
0-15	SL	0.133	0.3	0.45	25	1.33	109	152	452	11.1	8.3
15-30	SL	0.133	0.3	0.45	25	1.33	109	152	452	11.1	8.3
30-60	S	0.153	0.32	0.49	25						
60-90	SL	0.19	0.28	0.48	25						
90-120	CL	0.195	0.25	0.45	25						
Arenic cambisol (662 ha)											
0-15	LS	0.17	0.266	0.384	90	1.3	43	823	0	10.7	7.3
15-30	LS	0.177	0.262	0.425	90	1.3	43	823	0	10.7	7.3
30-55	LS	0.129	0.195	0.329	90						
55-70	LS	0.165	0.266	0.34	90						
70-90	SCL	0.172	0.253	0.322	90						
90-130	SCL	0.172	0.253	0.322	90						
130-160	LS	0.199	0.273	0.335	90						
Haplic luvisol (3047 ha)											
0-15	SL	0.165	0.294	0.45	50	1.39	236	69	7	10.7	7.8
15-30	SL	0.165	0.294	0.45	50	1.39	236	69	7	10.7	7.8
30-45	SL	0.165	0.294	0.45	50						
45-60	SCL	0.177	0.31	0.4	50						
60-90	SCL	0.177	0.31	0.4	50						
90-120	SCL	0.177	0.31	0.4	50						
120-150	SCL	0.177	0.31	0.4	50						
Haplic luvisol (retic) (1259 ha)											
0-15	SL	0.167	0.375	0.428	87	1.32	167	103	0	9.2	6.9
15-30	SL	0.167	0.375	0.428	87	1.32	167	103	0	9.2	6.9
30-50	SL	0.15	0.35	0.398	10						
50-94	SCL	0.182	0.37	0.4	10						
94-115	SCL	0.185	0.38	0.403	10						
115-145	SL	0.223	0.36	0.417	10						
145-195	SL	0.211	0.37	0.417	10						

The map of initial SOC contents (TOCini) for the ploughed layer was elaborated by Hamiache et al. (2012) using a geostatistical model of cokriging with external drift (elevation) relying on 141 point sampled sites spread across the study area. In order to facilitate calculations in spatio-temporal simulations, the TOCini values were transformed to the median of the quartile classes to which they belonged, separately for each soil type (Fig. 2).

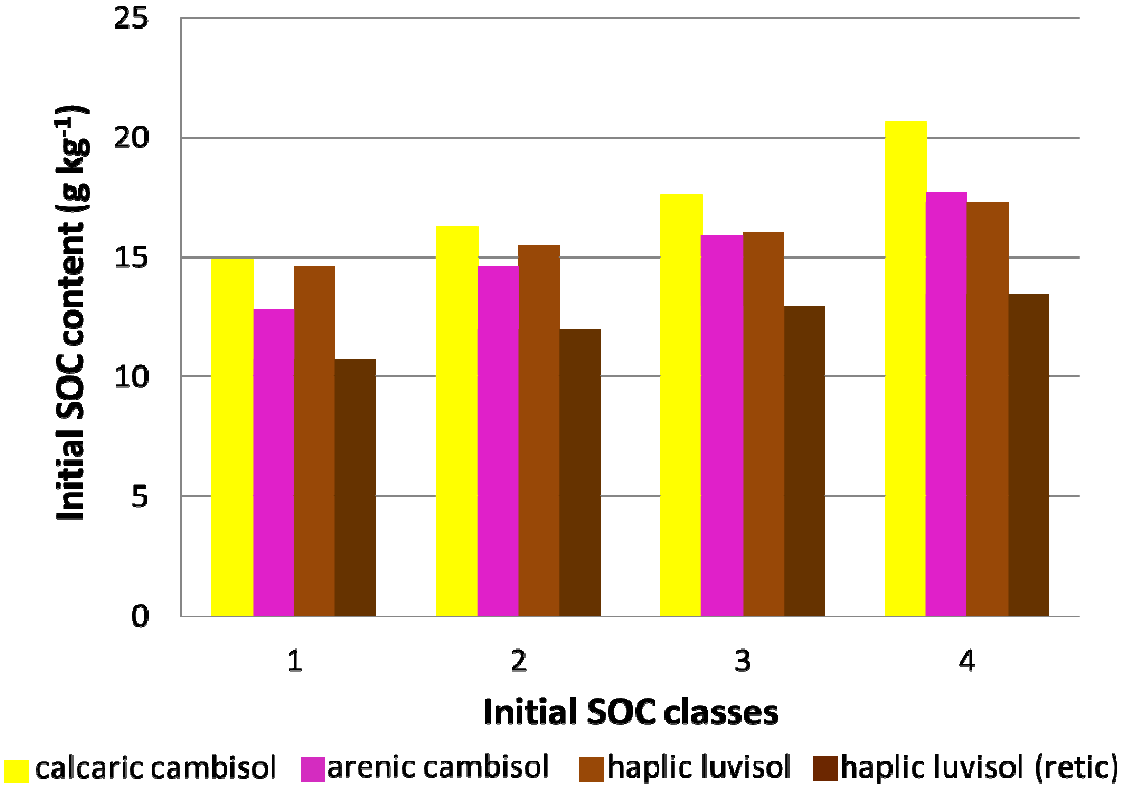


Figure 2. Initial SOC content (TOCini) classes (1 to 4: 1st to 4th quartiles) for each soil type

Information on cropping systems comprised the map of crop successions, and for each crop succession, sowing dates and densities, yield objectives, quantities and fractionation of N fertilisation. For each crop composing a crop sequence, characteristics were derived from a previous survey carried out over the study area (Dhaouadi, 2014) and from expert knowledge by agricultural advisers and local stakeholders (Table 2). As crop successions were mostly 4y or 3y-crop sequences, the spatial layer of crop successions was obtained from the intersection of four land parcel register layers between 2007 and 2011. These layers, named "*Registre Parcellaire Graphique*" (RPG) in France, locate all declared agricultural plots in the framework of the common agricultural policy of the European Union. Only the 1722 plots having stable boundaries over this period were retained for the simulations, considering the dominant crop within each plot. Then, in order to limit the amount of combinations of cases in the spatial simulations, the 7 main crop successions in terms of spatial coverage (77.1% of the utilised agricultural area) were selected: succ1, Rapeseed+3 cereals (mainly Rapeseed-Wheat-Wheat-spring Barley); succ2, Maize+3 cereals (mainly Maize-WWB); succ3, Rapeseed+cereal+Maize+cereal (RWMW); succ4, Rapeseed+2 cereals (on average RWWRWB); succ5, Maize+2 cereals (mainly MWW); succ6, Maize+cereal+fodder/forage/legume+2 cereals (MW-spring Pea-WW); succ7, Rapeseed+3 cereals+fodder/forage/legume (RWPWW).

Table 2. Description of cultural practices for scenarios

Crop	R		M		Spring B		Spring P		W after W		W after R		W after M		W after M in MWWB		W after P	
Sowing (density (grain m^{-2}) / date)	40	25th-Aug	9.2	17th-Apr	300	1st-Mar	85	1st-Mar	295	21st-Oct	295	21st-Oct	295	21st-Oct	295	21st-Oct	295	21st-Oct
Yield objective (Mg ha^{-1})	4.5		9.5		6		50		8		8.5		8		8.5		8.5	
Fertilisation Total quantity (kg N ha^{-1})	150		160		100		0		150		160		170		170		130	
F1 N (quantity / date)	75	14th-Feb	100	16th-Apr	30	01st-Apr			50	20th-Feb	50	20th-Feb	50	20th-Feb	55	20th-Feb	40	20th-Feb
F2	75	14th-Mar	60	16th-May	70	20th-Apr			60	25th-Mar	70	25th-Mar	80	25th-Mar	85	25th-Mar	60	25th-Mar
F3									40	25th-Apr	40	25th-Apr	40	25th-Apr	30	25th-Apr	30	25th-Apr
P need (kg P2O5 ha^{-1})	56		57		39		40		52		55		52		55		55	

R stands for rapeseed, W for winter wheat, B for spring barley, M for maize, P for dry pea or fodder crops.

The 7 considered crop successions ended up to a total of 1027 polygons. A dominant soil type as well as a dominant initial TOC value was assigned to each polygon. Finally, these 7 crop successions associated to one of the 4 soil types considered resulted to a total of 972 polygons covering 7498 ha.
In addition to the above-mentioned spatially-explicit data, climatic data recorded at two INRA-ECOSYS meteorological stations within the region were used: in Grignon (1998-2004) and in Feucherolles (2005-2011). Daily minimum and maximum temperature, rainfall, ETP Penman were measured. Climatic characteristics were considered as homogeneous over this small region, therefore these punctual meteorological data were used for the spatially-explicit temporal simulations.

3.3. Characterisation of regional EOM

The 15 EOM types that were available in the study zone were characterised through laboratory measurements (Noirot-Cosson et al., 20XX). Their potential C and N mineralisation were determined through incubations of soil+EOM mixtures conducted as described in the standard XPU44-163 (AFNOR, 2009). Total organic C and N of EOM were measured (AFNOR, 1995a for C and AFNOR, 1998 for N) and total organic C/N ratio calculated (Table 3). Their total P_2O_5 contents were measured following NF ISO 11263 standard method (AFNOR, 1995b).
Two main categories of EOM were considered: (i) “fertilising” EOM characterised by large potentially available mineral N, composed with highly degradable slurries (including Humival a dried pig slurry), sludges, litters, and (ii) “amending” EOM that are usually used to increase SOC contents, composed with all composts or manures of our panel except for composted poultry litter (belonging to fertilising EOM category because of high degradability.

Table 3. Exogenous Organic Matter characteristics (Noirot-Cosson et al., 20XX)

	EOM type	Mineral N content (mg N kg^{-1} DM)	OC content (g C kg^{-1} DM)	Organic C/N ratio	P_2O_5 (g P kg^{-1} DM)	C_r (%)	k_l (d^{-1})	k_r (d^{-1})
Fertilising	Dried Pig Slurry ("Humival")	696	407	7.8	47.2	40	0.376	0.00159
	Dried Sewage Sludge (SSd)	2897	397	5.6	51.6	17	0.376	0.00209
	Limed Sewage Sludge (SSl)	171	263	5.9	84.8	29	0.376	0.00209
	Dried Poultry Litter (PL)	2108	344	8.8	43.6	0	0.376	0.00209
	Composted Poultry Litter (PLc)	661	239	9.5	55.0	13	0.114	0.00159
Amending	Municipal Solid Waste compost (MSW)	2631	308	16.2	8.0	44	0.114	0.00159
	Biowastes Compost (Bio)	1247	208	8.6	11.0	92	0.114	0.000971
	Green Wastes compost "Sepur" (GWa)	23	334	23.1	5.2	45	0.114	0.000971
	Green Waste compost "BYS" (GWb)	231	268	10.6	6.5	87	0.114	0.000971
	Green Waste and Sewage Sludge compost 2009 (GWS)	4018	266	12.1	29.7	87	0.114	0.000971
	Green waste and bone meal ("Fertylis")	1438	257	16.7	19.7	79	0.114	0.000971
	Horse manure composted (HMc)	624	194	12.8	12.8	86	0.114	0.000971
	Horse manure with straw (HMs)	110	406	22.8	9.3	14	0.114	0.00159
	Horse manure with wood shavings (HMw)	71	428	29.3	10.2	21	0.114	0.00159
	Cattle manure (CM)	508	400	13.6	19.2	54	0.114	0.000971

DM stands for dry matter, C_r for the proportion of resistant carbon of the EOM, k_r and k_l for the decomposition rates of the EOM resistant and labile carbon respectively, C_r, k_r and k_l are models parameters defined in Noirot-Cosson et al. (20XX).

3.4. The CERES-EGC model: overall presentation, parameterisation and running

3.4.1. The CERES-EGC model

The CERES-EGC model (Gabrielle et al., 1995, 2005) is composed of three main modules: a so-called "physical" module describes heat, water and solute transfers in the soil, plant water uptake and evapo-transpiration governed by climatic demand; a second module addresses soil OM dynamics in the ploughing layer including microbiological mineralisation and immobilisation; and a third module addresses crop growth through crop net photosynthesis. Crop N uptake is computed through a supply/demand scheme, in which soil supply depends on nitrate and ammonium soil concentrations and root length density (Gabrielle et al., 2005). The CERES-EGC model uses Darcy's law to simulate water dynamics also adapted for fine textured soil (Gabrielle et al., 1995) and the modelling of OM dynamics is assured by the coupling with the NCSOIL model (see section 2.4.2), allowing environmental outputs such as nitrate leaching. Effects of clay and calcareous content on OM decomposition (Chaussod et al., 1986; Delphin et al., 1986; Brisson et al., 2009) was implemented in the OM dynamics module. CERES-EGC runs on a daily time step using daily weather data such as: solar radiation, rainfall, mean air temperature, potential evapotranspiration. It also requires water transfers parameters for the different horizons of soil, Soil Organic Matter (SOM) characteristics (for NCSOIL parameterisation) and crop management techniques as well as EOM adding characteristics when EOM are considered.

3.4.2. The NCSOIL model

The NCSOIL model (Barak et al., 1990; Hadas and Molina, 1993; Molina et al., 1983) simulates plant residue and EOM decomposition after incorporation into soil, nitrification, immobilisation and mineralisation of N, along with SOM decomposition and formation, by means of a system of controlling rates and decisional switches amongst a series of pools or compartments as described in Noirot-Cosson et al. (2016). The OM flow series implies 6 pools: (i) pool 0, termed "zymogenous biomass", (ii) pool I, also termed "microbial biomass", (iii) pool II, termed "mineralisable SOM", (iv) pool III, composed of highly-humified OM, (v) pools representing EOM, (vi) pools corresponding to crop residues. EOM and soil pools (except pool III) are divided into "fractions", one slowly decomposed, referred to as "resistant", and the second more labile and readily decomposed. Every fraction is characterised by its C content or size (mg C kg^{-1}), a decomposition rate (day^{-1}) for a first order kinetics of decomposition, an organic C/N ratio. The rate of N flows follows the C flows respecting the C/N ratio of the pool being decomposed, mineral N being added or withdrawn by the following pool to maintain its C/N ratio.

3.4.3. Model parameterisation

CERES-EGC was parameterised by means of different combined methods. The 20 years of meteorological inputs were obtained by averaging daily data over the 14 years of measurement (each

1st January of the 20 years of simulations were the same) to overstep the meteorological variability effects and focus on the other effects. The above-mentioned soil physico-chemical were directly used as soil parameters (Table 1).

For SOM and EOM parameterisation in NCSOIL, a set of typical parameter values from Corbeels et al., (1999), Hadas et al., (2004) and Nicolardot et al., (1994) was used and completed as described below. Size, C/N ratio, decomposition rate of soil pool II and EOM labile and resistant pools were adjusted.

For the SOM parameterisation in NCSOIL then CERES-EGC, the pool II decomposition rate was set to 0.002 as in Noirot-Cosson et al. (2016). Based on lab measurements of potential soil C and N mineralisation during incubation under controlled conditions, the pool II initial size (*CpII*) and its organic C/N ratio were optimised for each soil with the optimisation module of Barak et al., (1990) as described in Noirot-Cosson et al. (2016). Then, to prevent a quick and sharp *CpII* and TOC decrease over time through CERES-EGC simulations that would be obtained by directly using *CpII* optimised with lab measurements, we carried out a "spin up run" approach, a method often used for long-term SOC simulations (Coleman et al., 1997; Mondini et al., 2012; Exbrayat et al., 2014; Fang et al., 2015), resulting in steadier state for SOM. We used this method for each soil type to re-parameterise the *CpII*, using the optimised *CpII* and *CNpII* into CERES-EGC model through simulations of 40 years without EOM applications. This allowed determining a more stable *CpII* that we used as the initial pool II parameter for the regional simulations. The spin up run was carried out in the conditions of crop residues management and intercropping as defined section 2.5.1.

EOM parameterisation were based on Noirot-Cosson et al. (20XX) results. Decomposition rates of EOM labile and resistant pools (k_l and k_r) were determined, first using C mineralisation curves then using classification method to build coherent EOM clusters and assign the mean cluster k_l or k_r values to all EOM of the cluster. The EOM labile and resistant pool sizes (C_l and C_r) were simultaneously optimised as described in Barak et al. (1990). For the organic C/N ratio of the 2 pools we directly used total organic C/N ratio as optimising them did not improve much NCSOIL simulation of N mineralisation.

3.5. Building scenarios of EOM application for each cropping system

Considering the various EOM available for use over the region, and in conjunction with realistic management choices referring to crop residues or intercrops, scenarios of EOM application practices, including no EOM application for control scenario, were constructed for each of the 7 cropping systems. These scenarios comprised EOM types, EOM application dates, EOM quantity and number of application within a crop succession, and the order of priority of the crop to be amended within a crop sequence. They took into account constraints on P and N brought through amendment to prevent pollutions and comply with the EU regulations.

Table 4. Exogenous Organic Matter amendment scenarios

		Humival	SSd	SSl	PL	PLc	MSW	BIO	Gwa	GWb	GWS	Fertylis	HMc	HMs	HMw	CM
Crop succession	**P needs (kg P2O5 ha-1)**	**N from EOM (kg N ha^{-1}): Maximum application within the crop succession when constrained on P**														
succ1-RWWB	202	222	281	106	184	102	503	335	531	551	170	153	224	334	267	312
succ2-MWWB	203	223	282	107	185	102	505	336	533	553	171	153	225	346	268	313
succ3-RWMW	224	246	310	118	203	112	556	370	587	609	188	169	247	380	295	345
succ4-RWWRWB	314	345	435	165	286	158	780	520	823	854	263	237	347	534	413	484
succ5-MWW	161	177	223	85	146	81	400	267	422	438	135	121	178	274	212	248
succ6-MWPWW	260	297	360	136	236	130	645	430	680	706	278	196	287	441	342	400
succ7-RWPWW	256	287	354	134	232	128	635	423	670	695	214	193	282	434	336	394

N from EOM for each application (kg N ha-1)	**Scenarios of EOM applications**																	
Constraint on N (kg N.ha^{-1})	**<170**	**<250**	**<170**	**<250**	**<170[1]**	**<170**	**<250**	**<170[1]**	**<250**	**<250**	**<250**	**<250**	**<250**	**<250**	**<250**	**<250**	**<250**	**<250**
Nb of applications	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**
succ1-R*WWB	170	222	170	250	106		184	102	250	250	250	250	170	153	224	250	250	250
succ2-M*WWB	170	223	170	250	107		185	102	250	250	250	250	171	153	225	250	250	250
succ3-R*WMW	170	246	170	250	118	170	203	112	250	250	250	250	188	169	247	250	250	250
succ4-R*WWRWB	170	250	170	250	165	170	250	158	250	250	250	250	250	237	250	250	250	250
succ5-M*WW	170	177	170	223	85	146		81	250	250	250	250	135	121	178	250	212	248
succ6-M*WPWW	170	250	170	250	136	170	236	130	250	250	250	250	218	196	250	250	250	250
succ7-R*WPWW	170	250	170	250	134	170	232	128	250	250	250	250	214	193	250	250	250	250
Nb of applications	**2**		**2**	**2**		**2**			**2**	**2**	**2**	**2**	**2**	**2**	**2**	**2**	**2**	**2**
succ1-R*WWB*	111		140			92			250	168	250	250				172	133	156
succ2-M*WWB*	112		141			92			250	168	250	250				173	134	157
succ3-R*WM*W	123		155			102			250	185	250	250				190	147	172
succ4-R*WWR*WB		172	170	218		143			250	250	250	250	132		174	250	207	242
succ5-M*WW*			112						200	133	211	219				137		
succ6-M*WPWW*	149			180		118			250	215	250	250			143	221	171	200
succ7-R*WPWW*	144			177		116			250	211	250	250			141	217	168	197
Nb of applications	**3**		**3**						**3**	**3**	**3**	**3**				**3**		
succ1-R*WW*B*									168		177	184						
succ2-M*WW*B*									168		178	184						
succ3-R*W*M*W									185		195	203						
succ4-R*WWR*WB*	115		145						250	173	250	250				178		
succ5-M*WW*																		
succ6-M*WPW*W*									215		227	235						
succ7-R*W*PWW*									212		223	232						
Nb of applications									**4**		**4**	**4**						
succ4-R*WW*R*WB*									195		206	214						

[1]First constraint on N not binding, second not tested. C* stands for crop amended**.** RWWRWB, RWWB, RWMW, RWPWW, MWW, MWWB and MWPWW are the crop successions where R stands for rapeseed, W for winter wheat, B for spring barley, M for maize, P for dry pea or fodder crops. Among EOM Humival stands for the dried pig slurry, SSd for dried Sewage Sludge, SSl for limed Sewage Sludge, PL stands for Poultry Litter, PLc for Poultry Litter composted, MSW for Municipal Solid Waste compost, BIO for Biowaste compost, GWa for Green Waste compost "Sepur", GWb for Green Waste compost "BYS", GWS for co-compost of green waste and sewage sludge, Fertylis for the Green Waste and bone meal compost, HMc for Horse Manure composted, HMs for Horse Manure with straw, HMw for Horse Manure with wooden shavings, CM for Cattle Manure.

3.5.1. Crop residues and intercrops

Referring to crop residue management, as it is a common practice among farmers concerned by the SOC contents of their soils, a systematic "leaving on field and incorporating" was set. Concerning intercrops, a systematic mustard was set before spring crops such as maize, barley and pea, as mandatory in vulnerable zones such as the Versailles Plain region as defined by the "nitrate directive", referring to the Council Directive 91/676/EEC aiming to protect waters against pollution caused by nitrates from agricultural sources.

3.5.2. EOM applications

Concerning the EOM applications time, two dates were determined depending on EOM type: (i) as fertilising EOM may result in a high N crop supply and are likely to replace the synthetic fertiliser application, these EOM were applied one day before the first synthetic fertiliser application (Table 2), while (ii) amending EOM were applied end of August (20th) as farmers usually do (Dhaouadi, 2014). EOM were always incorporated into the first 30 cm depth of ploughing layer after application.

Despite the lack of regulations on EOM amendment regarding P fertilisation, and because growing concerns about P soil contamination (Némery et al., 2005), we decided to restrict the P input originating from EOM applications to the cumulated crop needs over a crop succession-time scale (Castillon et al., 2009). A second constraint was taken into account based on the French application of "nitrate directive". It limits the quantity of N brought through EOM application (N organic + mineral N) to 170 kg N ha^{-1} of utilised agricultural area at the farm scale in vulnerable zones. Therefore, for spring applications of fertilising EOM we set the constraint to 170 kg N ha^{-1} of maximum annual EOM N input but we also tested scenarios with a constraint set to 250 kg N ha^{-1} to explore higher application quantities. For EOM applied in September, we only tested the constraint set to 250 kg N ha^{-1} in order to decrease the number of scenarios simulated, assuming that N leaching would be limited with these EOM for which only a low proportion of N is readily available.

The maximum quantities of EOM that could be applied over a given crop succession (as restrained by the limitation rule due to P needs) were applied either once in the crop succession, or split into 2, 3 or 4 applications (one per year maximum), the maximum number of applications in crop successions being restricted by crop succession calendars (see section 2.5.3) and defined by the binding on the N constraint: as long as the quantity at each application was above the N limit (before applying this latter limit) and the crop calendar constraint allowed other application in the crop succession, one more application within this crop succession was created and thus another scenario was considered, the N limit considered for this being 170 kg N ha^{-1} for the fertilising EOM and 250 kg N ha^{-1} for the amending EOM. Two scenarios of 3 applications of dried Sewage Sludges (SSd) on the succ6 and succ7 successions with pea were not tested because once divided into 3, quantities applied were too low (< 2 tons raw material per ha). Some scenarios were not tried because too similar to others. EOM amendments scenarios are summarised in Table 4.

3.5.3. Crop succession constraints for EOM application

On the basis of former surveys (Dhaouadi, 2014) and some agronomical expertise, an order of priority of the crop to be amended was defined for each crop succession. Additional constrains for EOM application due to crop calendar were also taken into account as it could limit the number of possible applications: for instance, EOM application was not possible between maize and wheat because of a too short time lag between maize harvest and soil preparation before sowing, and furthermore because of the interdiction to spread EOM after November 15th in vulnerable areas.

3.6. Insertion of mineral N saving in the modelling

3.6.1. Principle

The objective was to consider the decrease of synthetic mineral N needs for crops due to the progressive increase of mineral N availability ("extra N min") in scenario simulations with EOM application. Both the direct effect related to an EOM application, and the increased mineral N available due to SOM increase after repeated EOM applications were considered (Fig. 3).

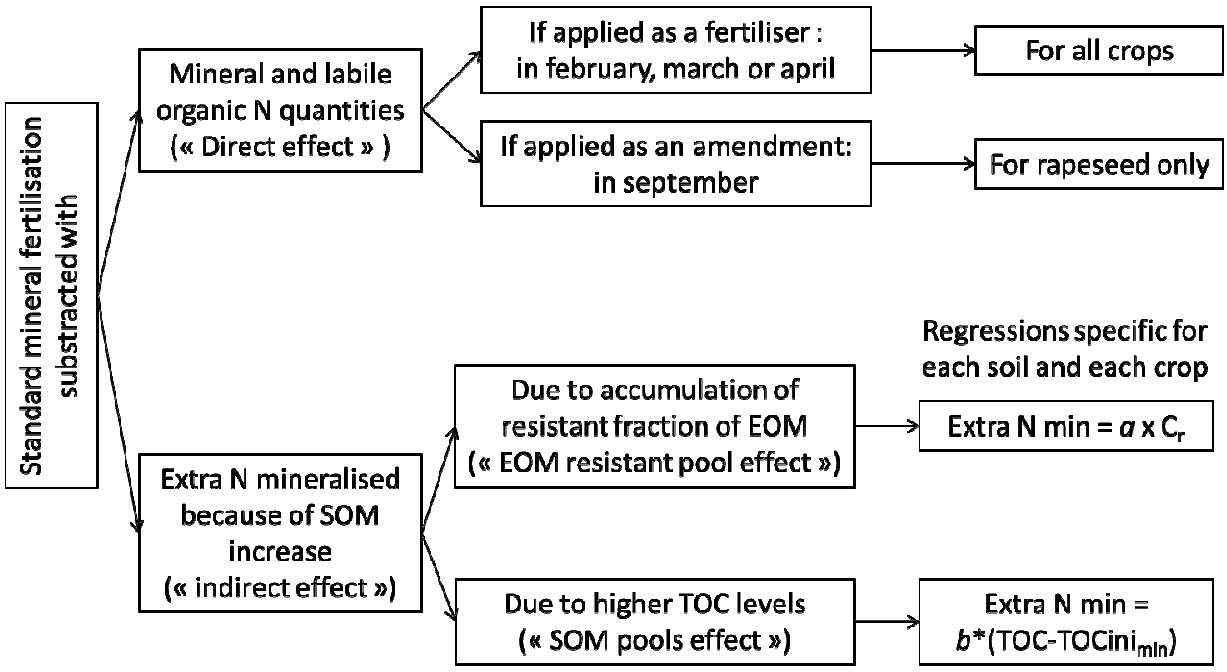


Figure 3. Scheme of the rules for nitrogen saving calculation.
"Extra N min" stands for surplus of mineral N stocks (kg N ha^{-1}) in soil when EOM is applied or when the organic carbon content in soil pools (TOC, g C kg^{-1}) is higher than the lowest initial SOC content ($TOCini_{min}$, g $C.kg^{-1}$) specific of a soil type, C_r (g $C.kg^{-1}$) stands for EOM resistant pool size, *a* and *b* are regression coefficients (kg N ha^{-1}/ g C kg^{-1}), details of their calculations are given in figure 4.

Direct effect

For **EOM considered as organic fertilisers** and applied in spring, the readily available N after EOM application, composed of both EOM-mineral N and organic N from the EOM labile pool, was directly substituted from standard mineral fertilisation (Table 2). Indeed we considered that this N would be

totally available for the crop supply, while losses through leaching would be limited because crop was already sown and N uptake started.

For EOM considered as organic amendments and applied in September, a direct effect on N available as previously described was only considered for rapeseed.

Extra N mineralised because of increased SOM

The total SOM increase was due (i) to accumulated resistant organic fraction of EOM in soil and (ii) to increased soil organic pools. Both origins were considered through separate simulations (Fig. 4). First we considered that the standard mineral fertilisation as defined in Table 2 corresponded to a soil with no EOM and with the lowest TOCini value for each soil type. The fertilisation could thus be reduced when total OM in soil has increased from this baseline.

For predicting the surplus of mineral N originating from accumulating EOM resistant pool (so-called "EOM resistant pool effect", Fig. 4a) or originating from increased SOM contents (so-called "SOM pools effect"), Fig. 4b), regression models relating "Extra N min" to either accumulation of EOM resistant carbon pool (C_r) in soil or increase of total organic carbon in soil pools (TOC) i.e. EOM pools excluded, were constructed on the basis of 20 y-CERES-EGC simulations of mineral N release (Fig. 4). For the "EOM resistant pool effect", 2 scenarios of EOM amendment with applications of either horse manure (HMs) or green waste compost (GWb) and crop successions 3 (RWMW), 4 (RWWRWB), and 6 (MWPWW) were considered. For the "SOM pools effect", the simulations concerned each of the 4 soil types, 3 of the 7 crop successions, and each of the 4 TOCini. No application of EOM was considered in these scenarios. The crop successions 2 (MWWB), 4 (RWWRWB), and 6 (MWPWW) were considered in order to retrieve simulations for all existing pairs of a given crop and its preceding crop. A total of 24 and 48 combinations were thus considered for the "EOM resistant pool effect" and the "SOM pools effect", respectively resulting in a total of 1440 years of simulations used for regressions buildings.

For the "EOM resistant pool effect", the mean predicted values of soil mineral N over the 120 cm depth was calculated ($\overline{Nsoil}$ in kg N ha^{-1}) and the mean predicted C_r ($\overline{Cr}$ in g C kg^{-1}) over the plant uptake period for each year of simulation. Regressions were constructed between these means, $\overline{Nsoil}$ and $\overline{Cr}$, for each crop of each crop succession, each soil type and each EOM scenario (examples given fig.1S). The slopes a obtained were averaged for each crop and each soil type, the intercept being set to 0 to calculate the extra mineral N available compared to a reference without EOM. The equations allowing the calculation of the quantity of mineral N fertiliser to be subtracted to the standard fertilisation were implemented in CERES-EGC in the forms of:

$$Extra\ N\ min = a_{crop,\ soil} \times C_r \qquad [1]$$

With "Extra N min" in kg N ha^{-1}, $a_{crop,soil}$ in kg N ha^{-1} / g C kg^{-1}, C_r in g C kg^{-1}.

For the "SOM pools effect", the mean predicted values of soil mineral N over the 120 cm depth as previously ($\overline{Nsoil}$ in kg N ha^{-1}) and the mean predicted TOC ($\overline{TOC}$ in g C kg^{-1}) were calculated over the plant uptake period for each year of simulation. For each of the 4 initial TOC, the $\overline{Nsoil}$ and $\overline{TOC}$ values were averaged over the 20 years of simulations in way that for each soil and each crop of each crop succession, 4 points of ($\overline{Nsoil}$;$\overline{TOC}$) allowed to define a trend of extra mineral N released by increased TOC (examples given in Fig. 2S). The slopes *b* obtained were then averaged for each crop and each soil type. The equations allowing the calculation of the quantity of mineral N fertiliser to be subtracted to the standard fertilisation were implemented in CERES-EGC in the forms of:

$$Extra\ N\ min = b_{crop,\ soil} * (TOC - TOCini_{min}) \quad [2]$$

With *Extra N min* in kg N ha^{-1}, $b_{crop,soil}$ in kg N ha^{-1} / g C kg^{-1}, *TOC* and $TOCini_{min}$ in g C kg^{-1}. The initial extra N min related to the different classes of TOCini was thus considered.

The plant uptake period was defined as follows: (i) the beginning of uptake was defined as the fertilisation date as such date should be based on the beginning of crop supply; (ii) the end of uptake was determined as 1 month before harvest (Comifer, 2013).

3.6.2. Some additional rules

Some additional rules were also adopted to deduct the surplus of available N to the standard mineral N fertilisation.

For the direct effect of EOM considered as organic fertiliser and applied in spring, the substitution was done for the first application of mineral N fertiliser and eventually the second one. For EOM considered as organic amendments and applied in September, a direct effect on N available was only considered for rapeseed and distributed between the first 2 mineral fertiliser applications.

For the extra N min coming from "SOM pools effect" and "EOM resistant pool effect", the quantity of N subtracted to the standard mineral N fertilisation was split between the 2 first fertiliser applications.

When the mineral N application became lower than 30 kg N ha^{-1}, we considered that a farmer would avoid such low application and therefore, the remaining quantity was added to the second mineral N application. For wheat, the N quantity of the third application was not decreased as it is better for grain N filling, unless the second mineral N application became lower than 30 kg N ha^{-1}; in this case the third application was cancelled and the remaining N was added to the second one, this second being considered more important regarding plant needs.

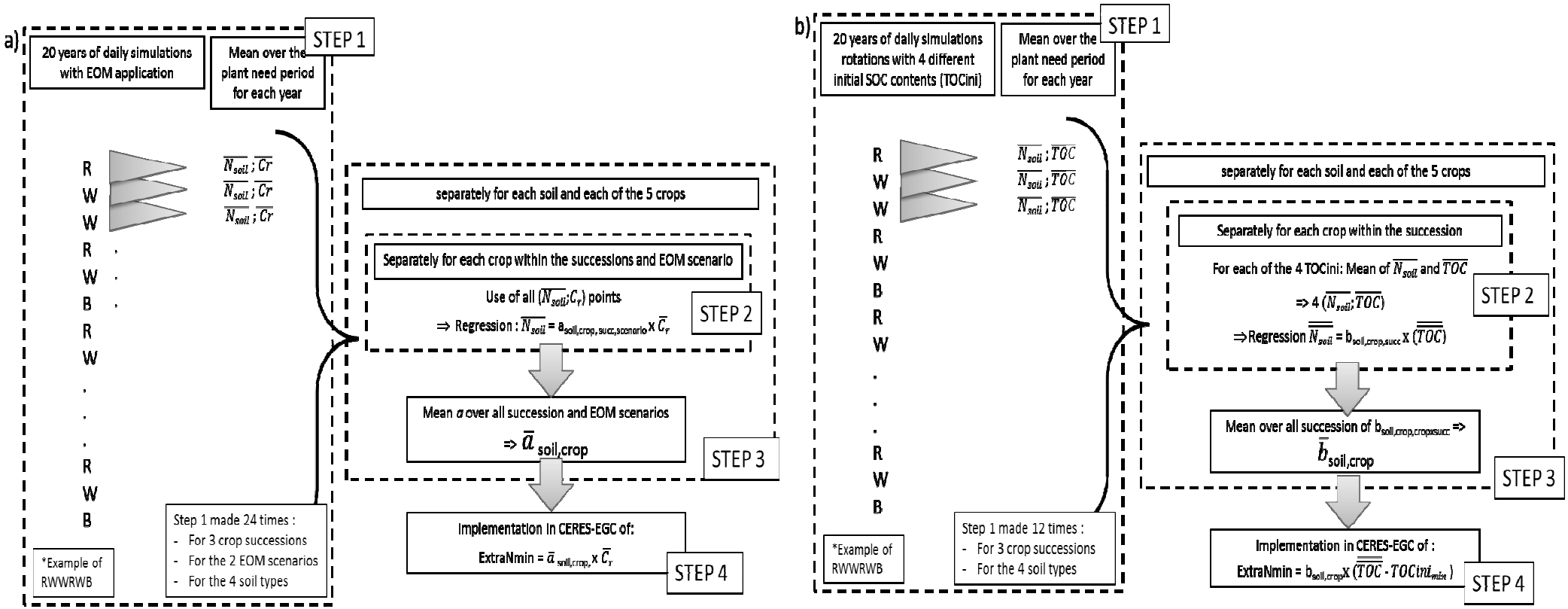


Figure 4. Calculation of regression coefficients both expressed in kg N ha^{-1}/ g C kg^{-1}: *a* for the "EOM resistant pool effect" and *b* for the "SOM pool effect". TOC stands for organic carbon contents in soil pools, $TOCini_{min}$ for the lowest initial TOC.

3.7. Running the model

Simulations of all scenarios corresponding to all crop successions, all soil types and all TOCini were performed with Python2.5©. The outputs considered were carbon storage (CS) i.e. the increase of total soil organic carbon (EOM pools included) after 20 years of simulations, nitrogen saving (NS) as previously calculated and nitrogen leaching (NL) cumulated over the 20 y-period, either the absolute values mostly for control scenarios (without EOM applications) or the specific effect of EOM applications, i.e. net from control, were considered. These net EOM outputs were related with C or N brought with EOM applications or with soil type or crop within the crop successions on which they were applied.

3.8. Analysis of the factors driving carbon storage, nitrogen saving and nitrate leaching

ANalysis of COVAriance (ANCOVA) were performed in order to assess the relative importance of the different factors driving CS, NS and NL, considered as the explained variables. Two categories of determining variables were considered: (i) characteristics of crop plots i.e. “soil type”, “crop succession” and “initial SOC content”, and (i) characteristics of EOM scenarios such as “EOM type”, “EOM application frequency within the crop succession” and others specific to each explained variable. Among the other determining factors were: “total C applied” in 20 years and “C per application” for CS analysis, “total N from EOM applied” and “N from EOM per application” for NS and NL analysis, and also “total N from EOM+synthetic” for NL analysis. The 16 “initial SOC contents” (Fig. 2) were merged into 8 classes the boundaries of which were based on a 1 g C kg^{-1} increment from 12 to 18 g C kg^{-1}, in order to assess the initial SOC content effect independently to soil type. In order to appreciate (i) the significance of each determining factors in the variability of the explained variables, and (ii) their relative importance, we considered: (i) the p-value of the Fischer test performed for each factor and (ii) the associated Sum of Squares (SS), the higher SS, the more determining the factor.

4. Results

4.1. EOM characteristics and consequences on their insertion in scenarios

Table 3 summarises EOM characteristics and parameters used in NCSOIL. Two categories of EOM, fertilising and amending, could be discriminated by their composition and their NCSOIL parameters based on OM mineralisation dynamics (Noirot-Cosson et al., 20XX).

EOM categories differed by their labile/resistant composition and the degradability of their OM: the C_r value of fertilising EOM was lower than 40% while that of amending C_r was greater than 44% except for the two horse manures (HM) with C_r values of 14 and 21% for HMs and HMw respectively. Finally, fertilising EOM had higher k_l (0.376 d^{-1}) and k_r (0.00159 or 0.00209 d^{-1}) than amending EOM, k_l being 0.114 d^{-1} and k_r 0.00159 or 0.00114 d^{-1} for HM or the other amending EOM respectively.

These EOM NCSOIL parameters determined in Noirot-Cosson et al. (20XX) resulted from high C and N mineralisation from the fertilising EOM, low for the stable amending EOM. For the less stable amending EOM such as the manures, C mineralisation was high while N dynamics showed early immobilisations before re-mineralisation.

Fertilising EOM were also characterised by high P content, P_2O_5 content greater than 43.6 g kg^{-1} dry matter (DM), low organic C/N ratio, (<9.5). The amending EOM were characterised by P_2O_5 content lower than 29.7 g kg^{-1}DM and organic C/N ratio greater than 8.6, especially green waste compost (GWa), horse manure with straw (HMs) or with wooden shavings (HMw), having organic C/N greater than 20.

Since the rates of EOM application are based on P needs of a crop succession, the total amount of EOM potentially added during a crop succession, thus the total added OC and N varied among the EOM (Table 4). Higher N quantities could be applied when EOM with lower P_2O_5 contents, or higher OC quantities could be applied when EOM with higher OC contents or higher organic C/N ratios were used. Fertilising EOM such as limed Sewage Sludge (SSl) or composted Poloutry Litter (PLc) and amending EOM such as Fertylis and the green waste and sewage sludge co-compost (GWS) could only be applied once within the succession (except GWS for 6-y succ4). The other fertilising EOM mostly allowed 2 applications, but with quantities lower than 170 kg N ha^{-1} for the Poultry Litter (PL), or with larger quantities for SSd and Humival, allowing also a third application on succ4. Concerning the other amending EOM, manures, composted horse manure (HMc) and Biowaste compost (Bio) allowed 2 applications with higher quantities for Bio and HMs allowing a third application on succ4, while the other composts allowed up to 4 applications.

4.2. C storage

4.2.1. C storage drivers

Table 5. Carbon Storage (CS) factors, sorted according to their decreasing Sum of Squares (SS) i.e. their relative importance in CS variability

Factors	DoF	SS	F	P-value
Soil type	3	18586037.445	15052.424	< 0,0001
EOM type	13	6012826.483	1123.767	< 0,0001
Crop succession	6	2177493.110	881.752	< 0,0001
Total C from EOM applied	1	824112.883	2002.293	< 0,0001
Initial SOC content	1	177446.819	431.131	< 0,0001
C from EOM per application	1	86492.423	210.145	< 0,0001
Application frequency	1	62827.870	152.649	< 0,0001

DoF stands for degrees of Freedom, F for Fisher, P-value for the Fisher test

For CS, all factors studied had significant effects as shown by very low p-values for the Fisher test (Table 5). "Soil type" was the factor that explained the most CS variability, more than the other crop plots characteristics "crop succession" and "initial SOC content". The "EOM types" was the second most determining factor, and explained more CS variability than "total C applied", "C per application" and "application frequency".

4.2.2. Differences in C storage

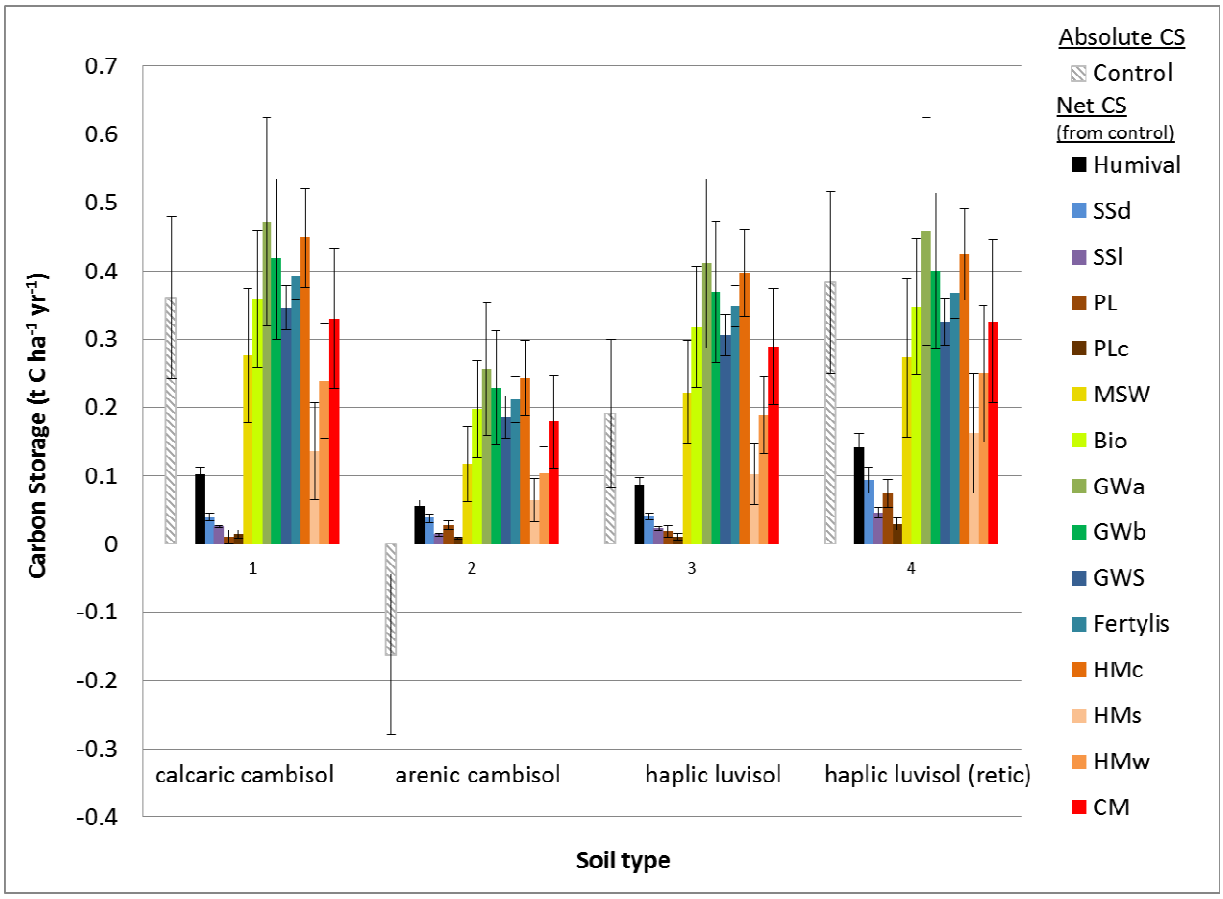


Figure 5: Soil Carbon Storage (CS) for control and CS from EOM (net control), for scenarios with one application limited to 250 kg N ha^{-1}, yearly averaged, each bar represents CS averaged for the 7 crop rotations and the 4 initial SOC contents.
EOM labels: "Humival" a dried pig slurry, SSd for dried Sewage Sludge, SSl for limed Sewage Sludges, GWS for Green Waste and Sludge compost, PL for dried Poultry Litter, PLc for Poultry Litter composted, MSW for Municipal Solid Waste compost, Bio for Biowaste compost, GWa Green Waste "Sepur", GWb for Green Waste compost "BYS", "Fertylis" a green waste compost and bone meal, HMc for Horse Manure composted, HMs for Horse Manure with straw, HMw for Horse Manure with wood shavings, CM for Cattle Manure.

Influence of soil type

Soil type appeared to be the variable better explaining CS variability. Examples of results are presented in Fig. 5 showing average results of scenarios corresponding to one application of each EOM within a crop succession and each soil type. For control scenarios, soils 1 and 4 stored respectively 0.36 and 0.38 t C ha^{-1} yr^{-1} on average over 20 years, allowing more storage than soil 3, 0.14 t C ha^{-1} yr^{-1}, and soil 2, -0.16 t C ha^{-1} yr^{-1}. In these control scenarios, simulated C storage could be

due to the incorporation of all crop residues in the scenarios. Another explanation could be the non-equilibrium of the TOC when starting the scenarios. Differences can primarily be explained by clay and calcareous soil contents, two factors lowering OM decomposition rates. The relatively high CS for soil 4, despite lower clay and calcareous contents than soil 3 and 1, could be due to its low TOCini, as shown in Fig.2, the lower the TOCini, the higher the potential CS.

The differences in CS from EOM (net from controls) between soils were generally lower than differences observed among the control scenarios, emphasising the strongest driving effect of soil type on CS. CS from EOM showed the same ranking by soil as CS for control scenarios, therefore factors discriminating CS between soils for control scenarios, i.e. clay, calcareous and TOC, might have the same effect on the EOM. Additionally, the ranking of CS from EOM was similar in each soil. However some slight soil-EOM interactions were observable. For instance, slightly higher CS from amending EOM was simulated in soil 1 compared to soil 4 except for the two HM, and less for all the fertilising EOM and the two HM. On the other hand, soil 2 caused less CS than soil 3 for amending EOM (except HM), while differences remained very low with fertilising EOM and HM. The protecting effect of clay and calcareous, stronger in soil 1 and 3 compared to soil 4 and 2, might be especially efficient with stable EOM, maybe because they remain longer in soil.

Influence of EOM-category

Regarding CS as a result of EOM use and their CS efficiency, i.e. CS from EOM divided by total C applied, potential CS appeared to be grouped by EOM types (Fig. 6).

Fertilising EOM such as PL, PLc, SSd, SSl, Humival led to the lowest CS ranging from 0.0 and 0.1 t C ha^{-1} yr^{-1} obtained with PL and Humival respectively. This was due to both limited quantity applied (maximum of 0.5 t C ha^{-1} yr^{-1}, because of high P content > 40 g DM kg^{-1}) and low potential CS from EOM because of rather labile carbon (C_r < 40%) and high decomposition rates. This also resulted in low CS efficiencies comprised between 2% and 25% (Table 6) increasing in the order litters < sludges < Humival.

Among the amending EOM, HM gave rise to rather low CS ranging from 0.1 to 0.3 t C ha^{-1} yr^{-1}despite high C quantity applied (1.0 to 2.3 t C ha^{-1} yr^{-1}). Indeed, because of their low P content (9.0 and 10 g DM kg^{-1}) and high C/N ratios (22.8 and 29.3), large C amounts could be applied while respecting constraints on P and N. However, their low C stability (C_r of 14 and 21% for HMs and HMw, respectively) explained these quite low CS efficiencies for HM, comprised between 7 and 15%. Most compost scenarios led to higher CS between 0.2 and 0.6 t C ha^{-1} yr^{-1} with applied C quantities of medium-level generally comprised between 0.5 and 1.5 t C ha^{-1} yr^{-1} because of their high C stability, high C_r values ranging between 44 and 87% and low decomposition rates. Cattle manure (CM) behaved more like composts rather than HM, due to its more stable carbon (C_r of 54% and low decomposition rates).

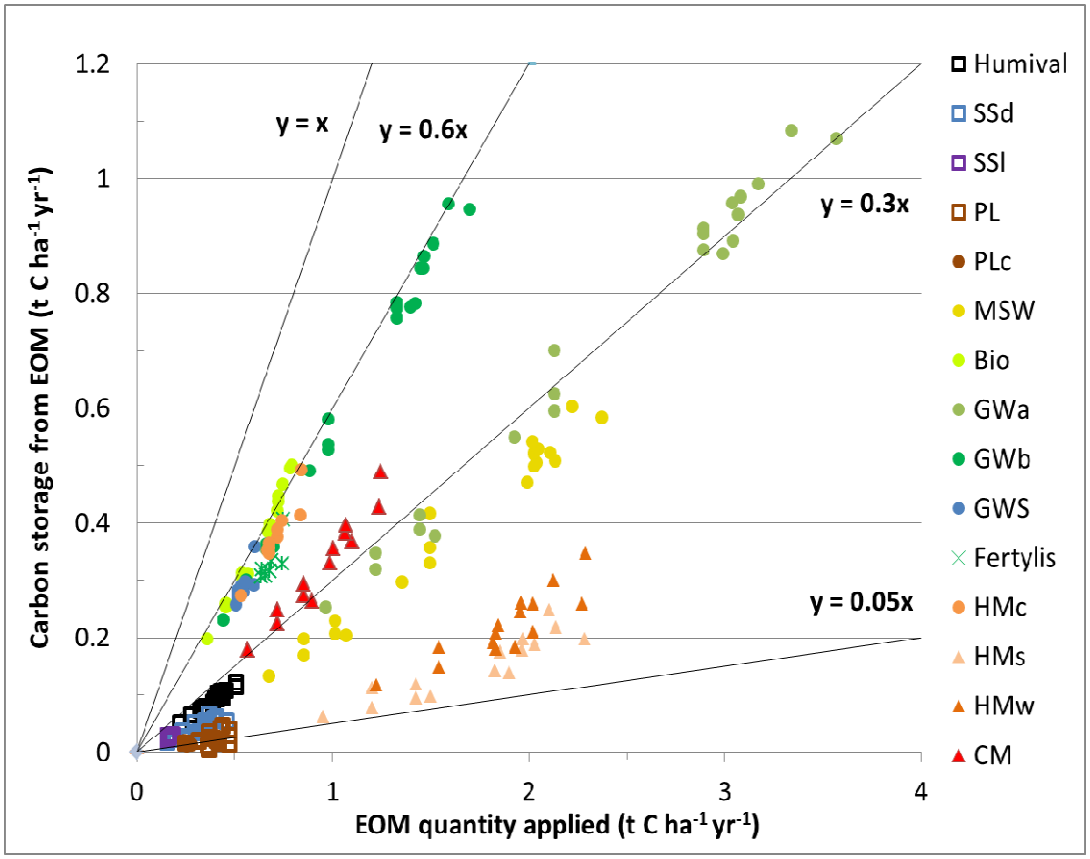


Figure 6. Mean annual Carbon Storage (CS) from EOM (net control). Each dot represents CS for one scenario i.e. 1 crop rotation x 1 EOM type x 1 nb of application x 1 quantity applied, obtained by averaging CS for 16 simulations (scenario simulations for 4 soil types and 4 initial SOC contents i.e. TOCini).

EOM types	CS efficiency	
	Mean	*Sd*
Humival	23%	*8%*
SSl	16%	*8%*
SSd	13%	*8%*
PL	6%	*6%*
PLc	6%	*4%*
MSW	24%	*7%*
Bio	59%	*12%*
GWa	29%	*7%*
GWb	56%	*12%*
GWS	53%	*12%*
Fertylis	48%	*11%*
HMc	53%	*12%*
HMs	9%	*3%*
HMw	12%	*4%*
CM	34%	*8%*

Table 6. Carbon Storage (CS) efficiencies for the various Exogenous Organic Matters (EOM) types. Mean and *standard deviation (Sd)* of all simulated scenarios over all soil types, crop successions and initial SOC contents

Among EOM Humival stands for the dried pig slurry, SSd for dried Sewage Sludge, PL for Poultry Litter, SSl for limed Sewage Sludge, PLc for Poultry Litter composted, HMs for Horse Manure with straw, HMw for Horse Manure with wooden shavings, Fertylis for the Green Waste and bone meal compost, GWb for Green Waste compost "BYS", GWa for Green Waste compost "Sepur", CM for Cattle Manure, HMc for Horse Manure composted, GWS for co-compost of green waste and sewage sludge, BIO for Biowaste compost and MSW for Municipal Solid Waste compost.

The two green waste composts (GWa and GWb) gave rise to the highest CS up to 1.1 t C ha^{-1} yr^{-1}: (i) GWb with relatively low quantity applied reached high CS efficiencies (up to 59%) mostly because of its stable C composition, (ii) the P and N contents of GWa made possible to apply higher C amount up to 3.6 t C ha^{-1} yr^{-1} despite lower CS efficiencies (maximum 33%) than GWb mostly due to lower *Cr*.
CS tended to linearly increased with the total C amounts applied of an EOM within the different scenarios (Fig. 6) leading to CS efficiencies quite constant for a given EOM (Table 6). However for some EOM, such as the manures and GWa, the CS efficiencies tended to increase with the total applied quantities of C. High EOM amounts may have slowed down the rate of EOM degradation and this can be explained by a limitation of C decomposition induced by a lack of available N, a limitation which is actually modelled in NCSOIL.

4.3. N savings

Evaluation of the N saving calculation

The calculation was tested on the same scenarios and same crop successions as these used to build the regressions, but was also tested on succ1 (RWWB) and succ2 (MWWB). The modelling was considered as validated because the new N fertilisation did not impact crop yields by less than 0.5 Mg ha^{-1}, which represents from 5 to 11% of crop yields for maize and rapeseed, respectively.

4.3.1. N savings drivers

According to the very low p-values for the Fisher test, the effects for all tested NS factors were significant (Table 7). “Crop succession” was the most determining factor followed by “EOM types”. They explained better the NS variability than the other factors.

Table 7. Nitrogen Savings (NS) factors, sorted according to their decreasing Sum of Squares (SS) i.e. their relative importance in NS variability

Factors	DoF	SS	F	P-value
Crop succession	6	183509.873	376.440	< 0,0001
EOM type	11	176934.849	197.974	< 0,0001
Soil type	2	78856.507	485.283	< 0,0001
Initial SOC content	1	12348.412	151.984	< 0,0001
Application frequency	1	5312.686	65.389	< 0,0001
N from EOM per application	1	3020.640	37.178	< 0,0001
Total N from EOM applied	1	326.778	4.022	0.045

DoF stands for degrees of Freedom, F for Fisher, P-value for the Fisher test

4.3.2. Differences in N savings

Examples of regressions (STEP 2 in fig. 4) obtained for wheat on soil 1 are showed in fig. 1S for the "EOM resistant pool effect" and fig 2S for the "SOM pools effect". The regression coefficients introduced in the CERES-EGC model to calculate the additional mineral N release related to increased OM in soil are shown in Table 8 for "EOM resistant pool effect" (*a coefficient in eq 1)* and Table 9 for the "SOM pools effect" (*b coefficient in eq 2*). The *a* coefficients varied with the soil type and decreased in the order: soil 2 > soil 4 > soil 3 > soil 1, inversely related to soils clay and calcareous contents. The a coefficient also varied with the crop type increasing in the order barley < wheat < maize < rapeseed, in relation with the duration of crop presence in the field. The *a* coefficients ranged between 6.9 kg N ha^{-1}/g C kg^{-1} for barley on soil 3 to 67.7 kg N ha^{-1}/g C kg^{-1} for rapeseed on soil 2. They showed low variability, with σ values lower than of 55% exhibited for rapeseed on soil 3.

Concerning the b coefficient (Table 9) used to calculate the extra N min related to TOC increase, they ranged from 0.9 to 4.7 kg N ha^{-1}/g C kg. The soil 2 had the highest *b* coefficients whatever crop except for wheat while soil 3 had the lowest *b* coefficients for barley and maize, and soil 1 for wheat and rapeseed. The *b* coefficients presented larger variabilities than the *a* coefficients specially in soil 1 and 3 and for barley. The extra N min release from soils not only depends on each soil's *b* coefficient but also on the absolute difference of TOCini compared to the $TOCini_{min}$. For instance, the maximum initial extra N releases were calculated for each type of soils, i.e. *b* coefficients multiplied by the "$TOCini_{max}$-$TOCini_{min}$" difference and varied from 2.4 to 23.2 kg N ha^{-1}, and generally sorted as follow: soil 2 > soil 1 > soil 4 > soil 3 (Table 9). Initial additional N releases related to differences in TOCini, were maximum on maize mostly because of a longer period of N uptake.

Table 8. "*a*" coefficient: Extra mineral N supply from the increasing resistant pool of EOM. Mean *a* and *standard deviation* calculated for each soil and each crop over the different successions.

a **(kg N ha^{-1}/g C kg^{-1})**	**Soil 1 calcaric cambisol**		**Soil 2 arenic cambisol**		**Soil 3 haplic luvisol**		**Soil 4 haplic luvisol (retic)**	
Wheat	10.6	*3.2*	30.7	*14.8*	11.8	*6.4*	26.5	*10.6*
Rapeseed	17.0	*4.7*	67.7	*26.5*	17.1	*6.7*	31.4	*9.6*
Barley	7.4	*0.0*	65.3	*3.5*	6.9	*1.3*	13.7	*0.4*
Maize	14.1	*1.9*	56.3	*20.5*	17.1	*7.1*	28.8	*11.1*

Table 9. "*b*" coefficient: Extra mineral N supply by soil with increased SOC content (TOC) compared to the lowest initial SOC content ($TOCini_{min}$). Mean *b* and *standard deviation* calculated for each soil and each crop over the different successions. Initial maximum extra mineral N supply in soil with the highest initial SOC content ($TOCini_{max}$)

b **(kg N ha^{-1}/g C kg^{-1})**	**Soil 1 - calcaric cambisol**		**Soil 2 – arenic cambisol**		**Soil 3 – haplic luvisol**		**Soil 4 – haplic luvisol (retic)**	
Wheat	1.0	*0.6*	1.5	1.0	1.2	*0.7*	1.9	*0.8*
Rapeseed	0. 9	*0.1*	3.3	*0.2*	1.0	*0.8*	2.0	*0.1*
Barley	1.7	*1.8*	3.2	*0.3*	0.9	*1.0*	1.6	*0.8*
Maize	2.7	*0.0*	4.7	*2.6*	1.4	*0.0*	2.3	*1.5*
$TOCini_{max}$-$TOCini_{min}$ (g C kg^{-1})	5.74		4.94		2.65		2.72	
Maximum initial Extra N min (kg N ha^{-1})	**Soil 1 - calcaric cambisol**		**Soil 2 – arenic cambisol**		**Soil 3 – haplic luvisol**		**Soil 4 – haplic luvisol (retic)**	
Wheat	5.9		7.6		3.1		5.0	
Rapeseed	4.9		16.1		2.5		5.3	
Barley	9.8		15.8		2.4		4.4	
Maize	15.5		23.2		3.7		6.1	

A comparison of total NS in scenarios corresponding to one application limited to 250 kg N within a crop succession is presented in Fig. 7. The differences of TOCini in the different soils led to average NS ranging from 3.5 to 5.4 kg N ha^{-1} an^{-1} without EOM application. When EOM were applied, NS varied with EOM types (Fig. 7). In crop successions starting with maize, fertilising EOM, especially SSd and PL, gave rise to significantly higher NS than amending EOM. The stable amending EOM such as GWb, HMc, and Bio and to a lower extent CM resulted in higher NS than less stable amending EOM such as HM and municipal solid waste compost (MSW) which contained more mineral and labile N. Such readily available mineral N was considered as not usable by spring crops such as maize, when amending EOM were applied in September but potentially leached during winter season. On the other hand, for more stable amending EOM, higher proportion of stable N (higher *Cr*), not yet degraded 7 months after application, leads to more potential mineralised N when maize can uptake mineralised N. Thus, the lowest NS was calculated for the least stable amending EOM (HMs). In crop successions starting with rapeseed, fertilising EOM resulted in less NS than the most labile amending EOM. The lower P content of these latter amending EOM allowed higher amount of OM and thus organic N applied, which is usable by rapeseed because of high proportion of labile N (low *Cr*) mostly mineralised during the following autumn. Hence, for crop successions differing in their first crop only, such as succ2 (MWWB) and succ1 (RWWB), NS was higher with SSd and PL fertilising EOM in succ2 (MWWB) and this was due to fertilisation need higher for maize (160 kg N ha^{-1}) than rapeseed (150 kg N ha^{-1}) totally supplied by these two EOM. For PLc, despite a very labile composition (C_r, 13%), N inputs from this EOM were not sufficient to cover neither maize nor rapeseed needs, leading to equivalent NS whatever the first crop of the succession.

Observing all scenarios (Fig. 8), NS was more variable within each EOM categories than CS. For instance, NS due to HM ranged between 5.2 and 55.0 kg N ha^{-1} yr^{-1}, SSd between 25.0 and 73.1 kg N ha^{-1} yr^{-1}, GWa between 18.6 and 73.7 kg N ha^{-1} yr^{-1}. Consequently, a similar NS level could be obtained through fertilising EOM, manures and composts depending on scenarios. HM allowed the lowest NS but for some scenarios with HMw they went up to 55.0 kg N ha^{-1} yr^{-1} without increased quantity of applied N from EOM. Higher NS were obtained when HM were applied on rapeseed, because the “direct effect” (mineral N and labile N) was considered useable and thus substracted to initial fertilisation. This readily available mineral N from HM was not accounted for other crops leading to lower NS when applied on other crops, that is why NS efficiency i.e. NS from EOM divided by N applied with EOM, ranged from 9% to 79% for HM.

Fertilising EOM and composts allowed similar range of NS but with different NS efficiencies. Indeed fertilising EOM caused NS from 20.3 to 73.1 kg N ha^{-1} yr^{-1} and composts from 11.2 to 73.7 kg N ha^{-1} yr^{-1}. However, NS efficiencies ranged from 55 to 99% for fertilising EOM while from 16 to 90% for composts. High NS efficiencies from fertilising EOM was related to their large “direct effect” and their application date synchronised with plant needs. With composts, high NS could be obtained when large amounts were applied allowed by low P content, but high NS efficiencies were obtained when

quantities applied were relatively low. Indeed, increasing quantity applied generally led to loss of NS efficiency. However, the differences in NS efficiency were rather due to the number of applications within the crop succession along with the type of crop amended. The best NS efficiencies were obtained when compost was applied on rapeseed in the case of only one application during the succession, while the lowest efficiencies occurred when compost was applied more than once and on maize first. For instance, NS efficiency was 81% when GWa was applied once on succ1 (RWWB) while it was 31% for succ2 (MWWB). When GWa was applied 3 times, NS efficiency decreased to 41% for succ1 (RWWB) and 29% for succ2 (MWWB). This might be primarily due to the fact that barley uptakes no more than 100 kg N ha^{-1} yr^{-1} and might be responsible for the loss of NS efficiency for rapeseed more than wheat.

A few EOM were applied only in small amounts: this was the case for SSl, PLc, Fertylis and GWS, because of high P contents for SSl and PLc or conjunction of moderate P contents, high OC contents and high C/N ratios as for Fertylis and GWS. Therefore these latter EOM were applied only once within the crop succession (except GWS applied twice on RWWRWB) and reached relatively high NS efficiency.

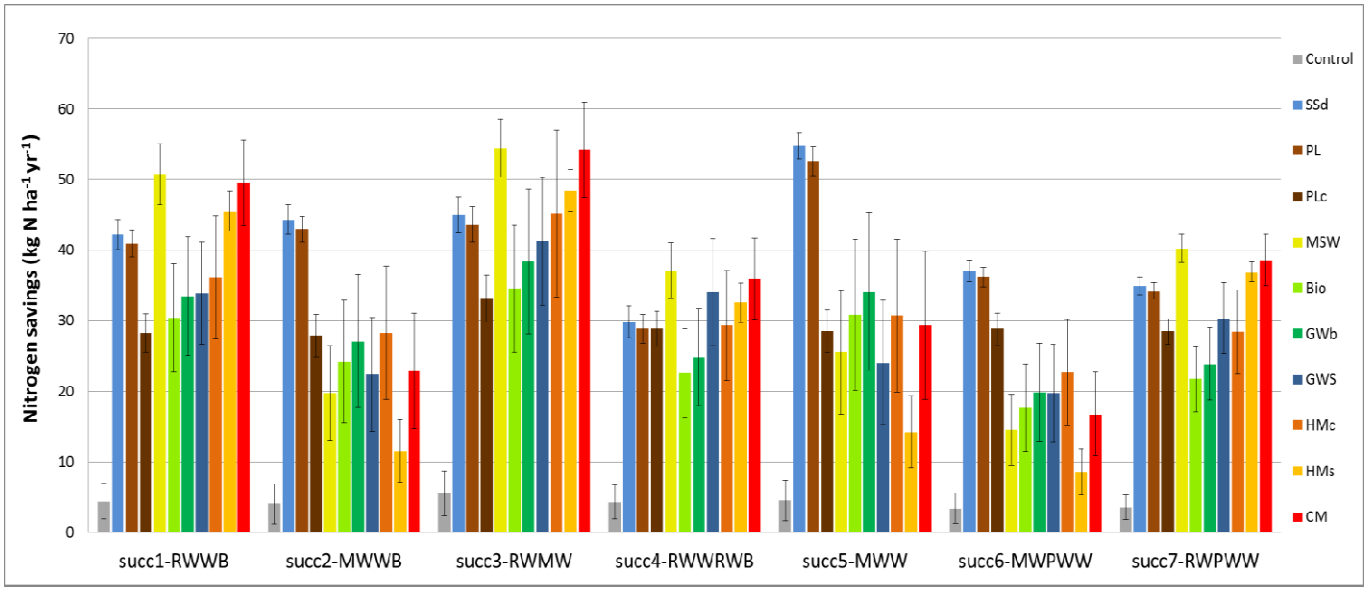


Figure. 7. Mean annual Nitrogen Savings (NS) for control and scenarios with one application limited to 250 kg N.ha^{-1} for contrasting EOM, i.e. with application frequencies depending on succession length, each bar represents NS averaged for the 4 soil types and the 4 initial SOC contents.

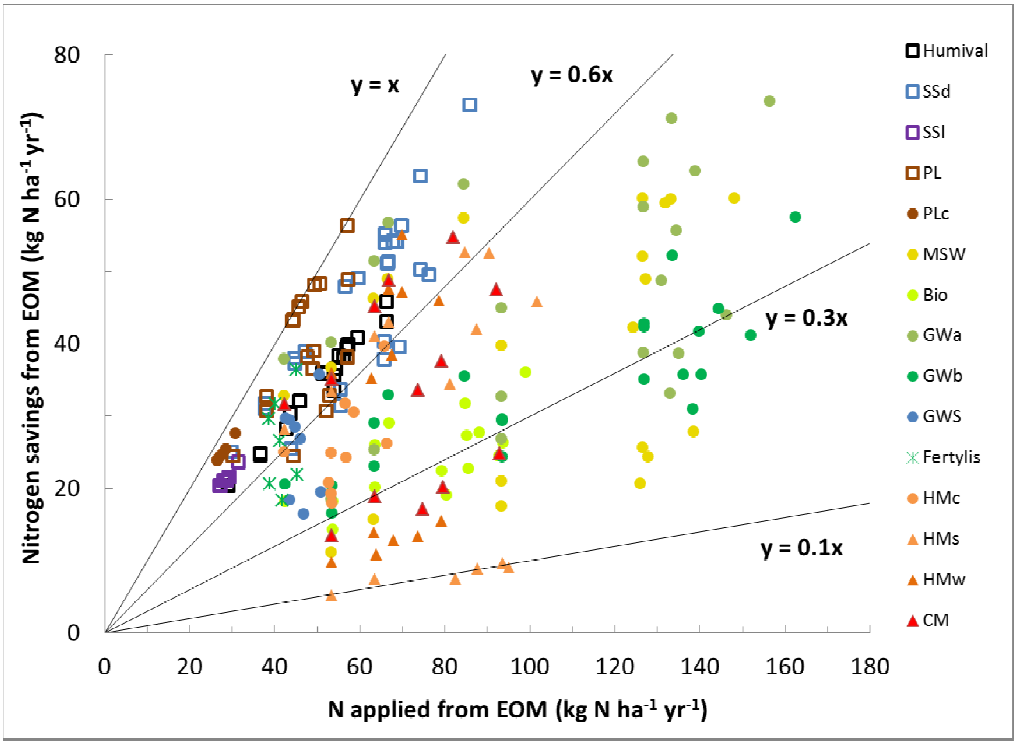


Figure 8. Mean annual Nitrogen Savings (NS) from EOM (net control). Each dot represents NS for one scenario i.e. 1 crop rotation x 1 EOM type x 1 nb of application x 1 quantity applied, obtained by averaging NS for 16 simulations (scenario simulation on 4 soil types and 4 initial SOC contents i.e. TOCini).

EOM labels: "Humival" a dried pig slurry, SSd for dried Sewage Sludge, SSl for limed Sewage Sludges, GWS for Green Waste and Sludge compost, PL for dried Poultry Litter, PLc for Poultry Litter composted, MSW for Municipal Solid Waste compost, Bio for Biowaste compost, GWa Green Waste "Sepur", GWb for Green Waste compost from "BYS", "Fertylis" a green waste compost and bone meal, HMc for Horse Manure composted, HMs for Horse Manure with straw, HMw for Horse Manure with wood shavings, CM for Cattle Manure.

4.4. Nitrate leaching

4.4.1. Nitrate leaching drivers

According to the low p-values for the Fisher test, all NL factors tested presented significant effects on N leaching (Table 10). NL variability was first explained by "soil type" and then by "crop succession". Among EOM scenarios characteristics, "EOM types" was the most determining factor.

Table 10. Nitrate Leaching (NL) variable factors, sorted according to their decreasing Sum of Squares (SS) i.e. their relative importance in NL variability

Factors	DoF	SS	F	P-value
Soil type	3	3922863.953	26875.064	< 0,0001
Crop succession	6	176068.628	603.112	< 0,0001
EOM type	12	108587.444	185.980	< 0,0001
Total N applied from EOM+synthetic	1	30301.909	622.784	< 0,0001
Initial SOC content	1	8044.935	165.345	< 0,0001
Total N from EOM applied	1	2463.372	50.629	< 0,0001
N per application	1	2365.352	48.614	< 0,0001
Application frequency	1	1383.410	28.433	< 0,0001

DoF stands for degrees of Freedom, F for Fisher, P-value for the Fisher test

4.4.2. Differences in N Leaching

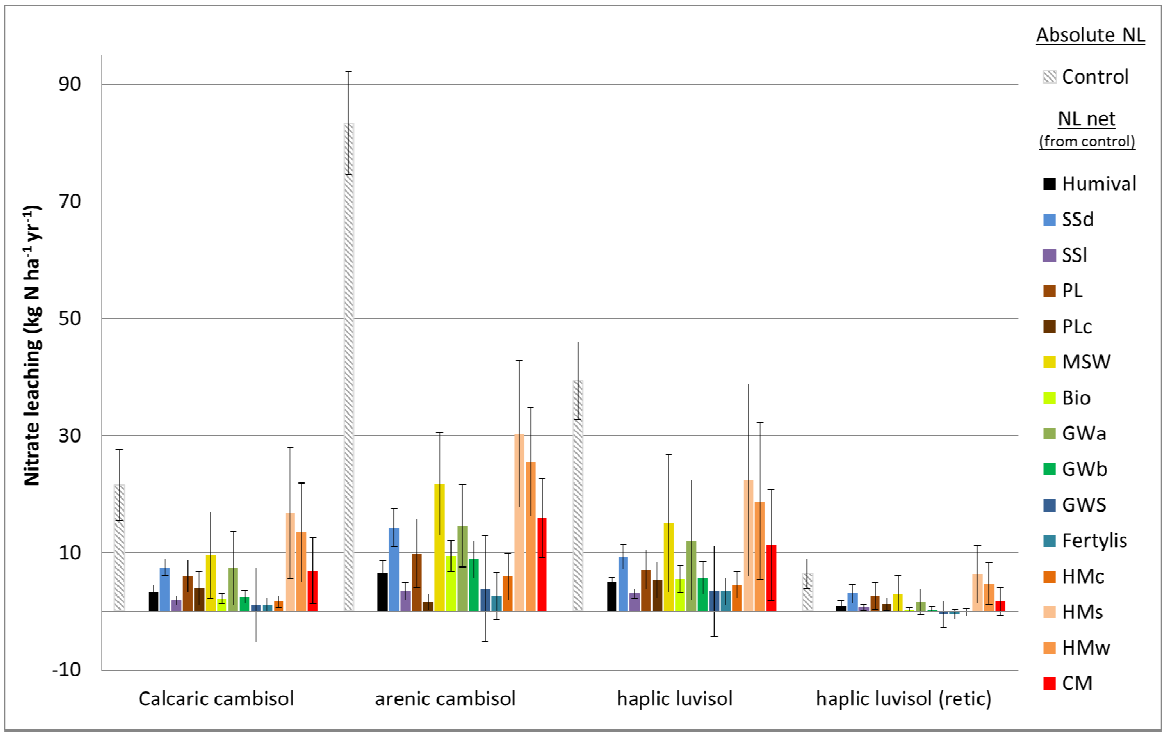


Figure 9. Nitrate Leaching (NL) for control and NL from EOM (net control) for scenarios with one application limited to 250 kg N ha-1, yearly averaged, each bar represents CS averaged for the 7 crop rotations and the 4 initial SOC contents (TOCini).
EOM labels: "Humival" a dried pig slurry, SSd for dried Sewage Sludge, SSl for limed Sewage Sludges, GWS for Green Waste and Sludge compost, PL for dried Poultry Litter, PLc for Poultry Litter composted, MSW for Municipal Solid Waste compost, Bio for Biowaste compost, GWa Green Waste "Sepur", GWb for Green Waste compost from "BYS", "Fertylis" a green waste compost and bone meal, HMc for Horse Manure composted, HMs for Horse Manure with straw, HMw for Horse Manure with wood shavings, CM for Cattle Manure.

NL largely varied with the soil type (Fig. 9), and was the highest for soil 2, the arenic cambisol, with 83 kg N ha^{-1} yr^{-1} on average in the control scenarios and decreased in the order: soil 2 > soil 3 > soil 1 > soil 4. The ranking could be explained by the potential SOM mineralisation inversely related to SOM storage potential in the upper horizon for nitrate production and the soil hydraulic conductivity Ks in the deep soil layers for nitrate leaching.

When EOM were added, soil ranking for NL remained unchanged except for PLc and Fertylis for which NL was larger in soil 3 than soil 2. The least stable amending EOM (HMs, HMw, MSW, CM and GWa) generally generated more NL than fertilising EOM, while the most stable amending EOM (GWb, GWS, Fertylis, Bio) caused less.

Fertilising EOM caused a maximum of additional 11 kg N ha^{-1} yr^{-1} representing NL rates (i.e. NL from EOM divided by quantity of N applied with EOM) ranging from 1 to 21%. For amending EOM, NL was more variable, especially for the least stable ones depending on crop succession and on numbers of application (Fig 10.). With Fertylis, GWS, Bio, HMc, NL was always lower than 9 kg N ha^{-1} yr^{-1} and NL rates remained lower than 1% because of low quantities of N applied and high OM stability. Compost such as GWb, GWa, MSW for which higher applied quantities were allowed by constraints, caused up to 16, 36, 40 kg N ha^{-1} yr^{-1} respectively and NL rate up to 12, 29 and 39% with the less stable the EOM, the more NL and the higher NL rate. All manures caused NL ranging from 7 to 39 kg N ha^{-1} yr^{-1} with intermediate quantities of N applied, therefore reaching NL rate higher than 50%. The NL and NL rate varied inversely to NS and NS efficiency depending on EOM types and scenarios.

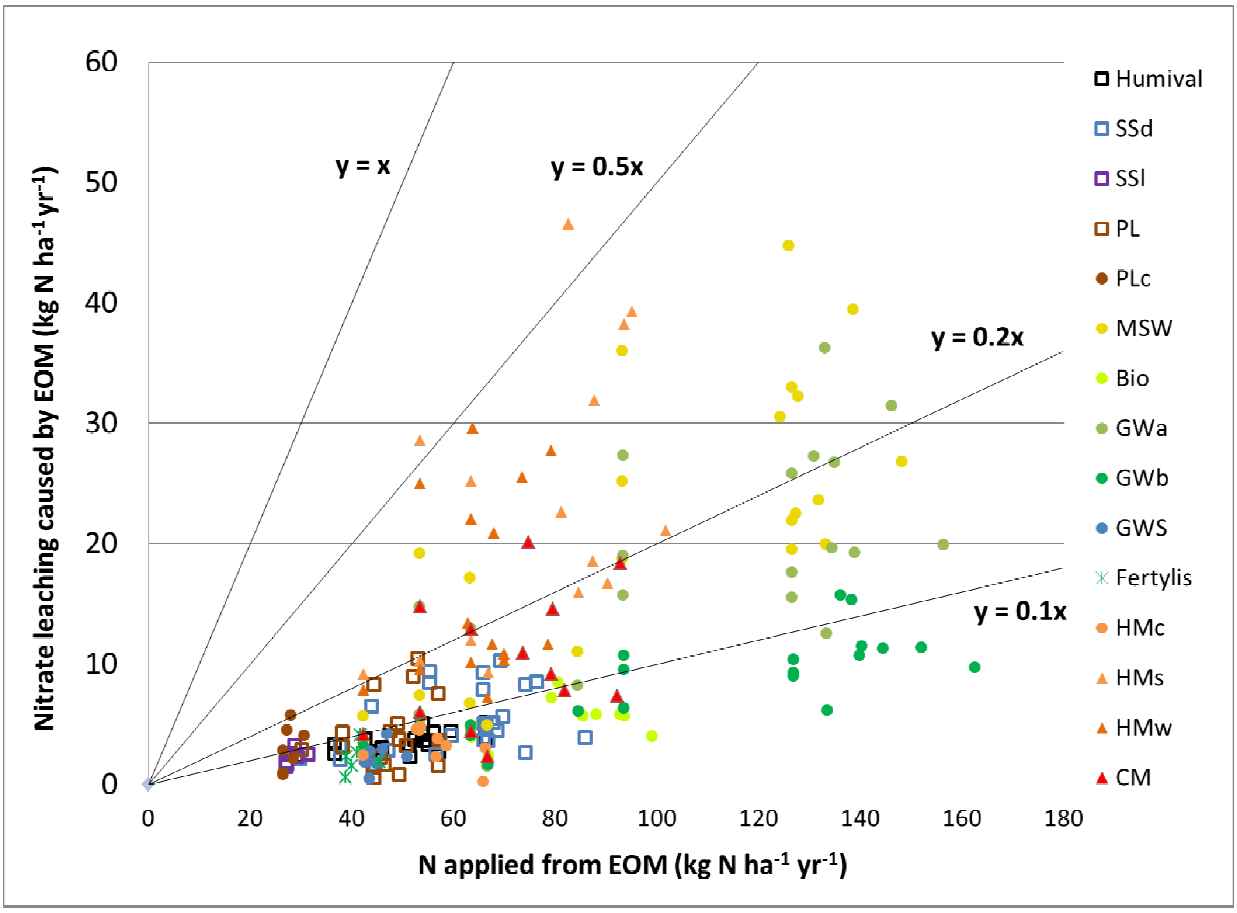


Figure 10. Nitrate Leaching (NL) caused by EOM (net control). Each dot represents NL for one scenario i.e. 1 crop rotation x 1 EOM type x 1 nb of application x 1 quantity applied, obtained by averaging NS for 16 simulations (scenario simulation on 4 soil types and 4 initial SOC contents i.e. TOCini).

EOM labels: "Humival" a dried pig slurry, SSd for dried Sewage Sludge, SSl for limed Sewage Sludges, GWS for Green Waste and Sludge compost, PL for dried Poultry Litter, PLc for Poultry Litter composted, MSW for Municipal Solid Waste compost, Bio for Biowaste compost, GWa Green Waste from "Sepur", GWb for Green Waste compost from "BYS", "Fertylis" a green waste compost and bone meal, HMc for Horse Manure composted, HMs for Horse Manure with straw, HMw for Horse Manure with wood shavings, CM for Cattle Manure.

5. Discussion

5.1. Simulations of C and N cycles

This study was indeed original because the EOM impacts on both C and N cycles were studied depending on the crop plot characteristics, including soil type, crop succession, initial SOC contents and EOM scenarios of application in the studied area. Added to the study of all the combinations of drivers of CS, NS and NL and their relative impacts on these variables, different options of EOM applications were explored in the scenarios of use. The EOM use could allow CS, NS but also generate risk of N and P water pollutions that we considered as well in our work. Both C and N cycles were studied here on the same time scale. When changes in SOC are hardly observable on a short time scale (Goidts and van Wesemael, 2007) and can be studied over up to 150 years (Tornquist et al., 2009), N cycle needs to be well simulated on a shorter time scale as crop growth and NL are sensitive to soil mineral N content which can quickly vary according to crop growth or rainfall events. An intermediate time span of 20 years was therefore chosen for our simulations. Such period also allowed to include several crop succession cycles lasting between 3 and 6 years in order to better take into account outputs variabilities caused by the various crop successions.

The CERES-EGC model was used to simulate different scenarios of EOM applications. Within CERES-EGC, the dynamics of C and N biological transformations in soil are described and simulated by the NCSOIL sub-model, parameterised based on laboratory experiments of soil-EOM incubations during which both C and N mineralisation were measured. Such parameterisation has been shown as relevant for the haplic luvisol and some amending EOM in Noirot-Cosson et al. (2016). In this previous work, CS efficiencies were on average 65, 65, 50 and 39% for GWS, Bio, CM and MSW, respectively, over a period of 13 years. In the present study, CS efficiency obtained for the same EOM on the same soil type and initial SOC content were on average 60, 67, 40 and 30%, respectively for all crop successions and all scenarios. This same order of magnitude suggests that the EOM parameterisation conducted was appropriate. The CS efficiencies simulated with the different EOM in the different soils were similar to other reported studies. Using the RothC model and over 20 years of simulation, Peltre et al. (2012) found rates of C accumulation ranging from 0.2 to 0.5 t C ha^{-1} yr^{-1} for doses of manure and compost application of 1 t C ha^{-1} yr^{-1}, respectively, where we found CS between 0.1 and 0.6 for the same application quantities of a larger panel of manures and composts. Also with the RothC model, Mondini et al. (2012) considering one scenario of 1.1 t C ha^{-1} yr^{-1} of compost amendment on Italian arable soils over 30 y-simulations, found CS potential of 0.3 t C ha^{-1} yr^{-1}, which fell in the range of the CS potential found here for input ranges of 1-1.5 t C ha^{-1} yr^{-1} of MSW or GWa (0.2 to 0.4 t C ha^{-1} yr^{-1}, for MSW; 0.3 - 0.4 t C ha^{-1} yr^{-1} for GWa) but was lower than CS obtained with the more stable compost GWb, with CS potential ranging from 0.5 to 0.8 with application of 0.9-1.3 t C ha^{-1} yr^{-1}. Yokozawa et al. (2010) studied 25 y-application of between 1 and 1.5 t C ha^{-1} yr^{-1} of farmyard manure and found averaged simulated CS of 0.3 t C ha^{-1} yr^{-1} which was close to our CM-

simulations (CS, 0.3 - 0.5 t C ha^{-1} yr^{-1}). Koga et al., (2011) with inputs of 1.1 t C ha^{-1} yr^{-1} of composted cattle manure mixed with straw found CS of 0.2 t C ha^{-1} yr^{-1} but on 39 years of simulation. Therefore, we could validate our results on CS efficiencies after EOM addition to soil.

In our study, soil type appeared to be the main explaining variable of CS variability. Wiesmeier et al. (2015) found that land use effect on SOC storage were strongly affected by soil types. We suggested that soil type effect on CS was mainly due to clay and calcareous limiting SOC decomposition in CERES-EGC. The importance of clay on CS is well known and is nowadays used to calculate the maximum potential SOC level for a soil through the concept of C-saturation (Angers et al., 2011; Six et al., 2002). Here, we found the CS would be better if EOM were applied on soils with high clay and calcareous contents or on soils with low SOC; the clay and calcareous protecting effect being even higher with stable amending EOM.

5.2. Nitrogen Savings and other impacts

An original result of this study was the attempt to estimate NS due to direct availability of EOM nitrogen when applied and to indirect effect of regular EOM application on SOM stocks. The calculation of NS was based on simulations of crop successions taking therefore into account all interactions between EOM, SOM, soil characteristics, climatic data, cropping systems. Some decision rules were used and assumptions were made related to the calculation of NS and distribution within the N mineral fertilisation practices. Regressions of surplus of N mineralised due to increase of organic matter in soil were specifically made for each crop and each soil type. Despite, such specificity, regressions parameters remained quite variable, depending actually also of the previous crop or the whole crop succession. This latter source of variability was not taken into account to limit the number of regressions however it could lead to some biases. Nevertheless, the NS modelling allowed to maintain simulated crop yields of the control scenarios.

Some tools already exist to calculate NS. For instance, coefficients related to direct effect of EOM application are proposed in decision-making tools available to calculate mineral N fertilisation taking into account a large panel of EOM types, the application timing and the crop on which they are applied (Comifer, 2013). The indirect effect of regular EOM application is also taken into account through the overall SOM mineralisation increased because of EOM application, considering clay and calcareous content. Despite similarities with our method, we proposed a more generic approach since based on EOM parameters that can be calculated for every EOM (see Noirot-Cosson et al., 20XX), and can thus take into account an even larger EOM panel. Secondly, our method distinguished the "SOM pools effect" from the "EOM resistant pool effect". The increase of OM in soil related with the accumulation of EOM resistant fraction would cause larger NS than the NS related with the increase in SOC (see NS coefficients table 8 and 9). This questions the composition of OM in soil and the proportion of past EOM amendments that are accumulated in the soil. In our study, EOM in soil was initially set to 0 g C kg^{-1} soil, but most of OM increase in soil was caused by EOM resistant pool

accumulation (not shown). Using the RothC model, Peltre (2010) also found that C accumulation in soil in case of regular EOM addition was largely related to accumulated residual fraction of EOM C.

Some interesting results emerged from this modelling. The highest NS were obtained with fertilising EOM especially when applied in spring on maize and with amending EOM with low stability when applied in autumn before rapeseed sowing. Moreover, the best NS efficiencies were obtained when EOM were applied once within the crop succession on rapeseed or maize crops. In case of several applications within a crop succession, barley were amended rather than wheat (in crop succession including barley), as farmers would do, according to expertise and farm polls. However, applying EOM on wheat rather than barley could be better in order to gain NS efficiency and probably to decrease NL. As well as maintaining yields, farmers pay attention to maintaining protein content in grains especially in wheat that can be obtain with maintaining the 3rd application of fertiliser as we did unless the two first application were reduced to less than 30 kg N ha^{-1} yr^{-1}.

Concerning NL, the soil type was also the main explaining variable considering all simulations were performed using the same climatic scenarios. The largest NL were simulated for the arenic cambisol with a sandy texture and were actually within the range of NL observed for sandy soils after EOM application (Rusakova, 2015). However, water fluxes in soil are simulated without considering capillary rise of the water table (Gabrielle et al., 1995) which could probably explained the endostagnic features observed in this soil and could likely impact NL. The lowest NL were simulated for the retic haplic luvisol, however for this soil Ks was adjusted using water fluxes measurements, leading to a very low Ks below 30cm depth (Chalhoub, 2010) while pedotransfer functions (Gabrielle et al., 2002) were used for the other soils leading to rougher Ks estimates.

The most important drivers of CS, NS and NL were crop plot characteristics (mainly soil types and crop successions), however, this was true at a regional scale. These characteristics may be less variable at a farm level and EOM amendments characteristics may become more discriminant

5.3. Uncertainties and how to reduce them

The description and parameterisation of EOM characteristics and scenarios of use, of soil types and SOM parameters, of climatic data and crop successions may create uncertainties in the C and N simulations in scenarios of EOM use. The potential CS related with EOM applications could become ambiguous when taking into account these uncertainties (Koga et al., 2011) as shown by performing Monte Carlo simulations. In our study, we showed which factors were the most discriminating for CS, NS or NL outputs. Soil type was the factor that most clearly accounted for CS and NL. However a limited number of soil types characterised the region; nevertheless they displayed very distinct parameters, among which clay and calcareous contents appeared to be key drivers of CS. Would a more detailed map deriving from a digital soil mapping approach and focused on these soil properties such as that made for SOC contents by Hamiache et al. (2012) result in decreasing CS and NL simulation uncertainties ? However NL is driven by hydric and hydrodynamic parameters the

assessment of which relies on time-consuming lab measurements after soil profile description and sampling and spatial variability may not be caught by detailed bottom-up spatial approaches because of dramatic uncertainties in water fluxes (Baveye and Laba, 2015). Similarly, crop successions took an important role in NS variability. As for soil types, using a simplified crop successions map, and then standard cropping systems may lead to some uncertainties. A further refinement in our modelling would consist in using crop information retrieved at the scale of all single fields in the study area, for instance as obtained from very high resolution Pléiades images in the western part of our study zone (Vaudour et al., 2015). Another important driver of SOC evolution is climate, this is why a lot of authors included different climate scenarios for SOC evolution predictions (Mondini et al., 2012; Yagasaki and Shirato, 2014). Global warming might decrease SOC storage (Lefèvre et al., 2014; Stockmann et al., 2013) because of a boost of SOC mineralisation related to temperature increase. However this latter effect is ambiguous as it could also be lowered by soil water content decrease. However, authors usually simulate on a longer period like 100 years, indeed, for 20 years as in our simulations, climate change effect may not change relative scenario results. Soil erosion could also play an important role in SOC evolutions as it can result in loss of 0.193 t C $ha^{-1}.yr^{-1}$ for croplands (Doetterl et al., 2012). It was not taken into account here but it may be rather limited as soil is rarely left bare in this study.

5.4. Feasibility of EOM use scenarios at crop plot and regional scale

In this study we explored scenarios of EOM use constrained by P and N inputs from EOM. Applications of EOM are actually mainly regulated based on N and for some EOM such as sludge and urban composts, trace elements (TE) quantities they brought. In our scenarios, the calculation of EOM quantities based on P needs of the crop successions led to lower potentially applied EOM amounts than if limited by TE as required by French regulations (NFU 44 051, NFU 44 095 or sludge regulation, depending on EOM) except for GWb and MSW for which the highest number of applications exceeded the maximum lead limit by only by 2-13% depending on the succession. The EOM quantities applicable were also always lower than if limited by K plant needs. This emphasises the environmental feasibility of our scenarios.

The amendment scenarios were tested without taking into account the EOM availability in the region. A regional scenario where all the crop plots would be amended with the highest number of applications possible (specific of each crop plot characterised by its surface and its crop succession) and with the highest quantity possible (< 250 kg N ha^{-1}) as considered in our scenarios (Table 4) would be possible according to the available EOM quantities (Dhaouadi, 2014) which could cover 149% of this need (table 11). For this purpose, regional requirements were exceeded because of a relatively high EOM availability in the region and because of amendments scenarios highly restricted by the P and N constraints. Using mean CS, NS and NL efficiencies despite potentially high variability depending on soil types or crop successions amended, and if covering only 100% of the regional need,

2135 t C yr^{-1} could be stored, 308 179 kg N yr^{-1} could be saved while a surplus of 57 269 kg N yr^{-1} could be lost by leaching. This potential NS would represent 29% of the standard mineral fertilisation over the region without EOM application. The potential surplus of NL would be 26% for a surplus of total N inputs of 52%. Considering mean efficiencies only allows to roughly estimate the regional potentials as efficiencies highly depended on soil type and crop succession where EOM were applied. Results in term of CS or NS and NL would actually depend on the EOM regional distribution. Mathematical tools should be considered in order to allocate the available EOM over the crop plots of the region.

CS potential in soil through organic amendment is a way to offset greenhouse gas (GHG) emissions. The regional CS of 2135 t C yr^{-1} would represent 7828 t CO_2 yr^{-1} sequestrated in soil.The use of synthetic N fertilisers also implies significant energy use and indirect GHG emissions (Sutton et al., 2013), it should be taken into account as well. Considering CO_2 emissions for Urea-Ammonium Nitrate (as used on the region) of 3.13 kg CO_2-eq kg^{-1} N (Nemecek and Kägi, 2007), 965 t CO_2 would not be emitted thanks to these NS. However, it is not taking into account neither differences in GHG emissions such as N_2O and fuel consumption of the different practices nor P and K savings whose conventional productions would also cause direct and indirect CO_2 emissions.

Table 11. Regional potentials of Carbon Storage (CS), Nitrogen Saving (NS) and Nitrate Leaching (NL), according to EOM disponibility in the region

EOM type	territory requirements[a] (t DM yr^{-1})	Available on the region (t DM yr^{-1})[b]	% of regional need covered	Mean CS efficiency	Regional CS potential (t C yr^{-1})[c]	Mean NS efficiency	Regional NS potential (kg N yr^{-1})[c]	Mean NL rate	Regional NL potential (kg N yr^{-1})[c]
Humival	7737	700	9%	23%	66	68%	25297	7%	2557
SSd	7117	3204	45%	16%	208	75%	178066	8%	18694
SSl	4617	311	7%	13%	11	75%	10491	9%	1193
PL	8739	375	4%	6%	8	83%	12836	9%	1352
PLc	7838	35	0%	6%	1	90%	813	11%	97
MSW	51060	0	0%	24%	0	41%	0	21%	0
BIO	26719	4080	15%	59%	499	33%	34153	7%	7496
GWa	70993	5050	7%	29%	497	49%	36030	16%	11890
GWb	42224	5020	12%	56%	753	34%	43636	8%	10182
GWS	15204	0	0%	53%	0	56%	0	4%	0
Fertylis	19529	6480	33%	48%	802	64%	69796	4%	4407
HMc	28668	0	0%	53%	0	47%	0	6%	0
HMs	45276	3672	8%	9%	127	38%	25252	30%	19715
HMw	35158	468	1%	12%	23	48%	3289	25%	1721
CM	20607	1450	7%	34%	198	48%	20974	15%	6295
		Total	149%		3192		460 632		85 599
Scenario « 100% of regional needs » covered		Total	100%		2135		308 179		57 269

EOM labels: "Humival" a dried pig slurry, SSd for dried Sewage Sludge, SSl for Limed Sewage Sludge, PL for Dried Poultry Litter, PLc for Poultry litter composted, MSW for Municipal Solid Waste compost, BIO for Biowaste Compost, GWa Green Wastes "Sep", GWb for Green Waste compost "BYS", GWS for Green Waste and Sludge compost, "Fertylis" a Green waste compost mixed with bone meal, HMc for Horse manure composted, HMs for Horse Manure with straw, HMw for Horse manure with wood shavings, CM for Cattle manure.

CS efficiency represents CS from EOM divided by C quantity applied, NS and NL efficiencies are NS and NL from EOM divided by quantity of N from EOM applied.

[a]need calculated for scenarios with maximum number of applications within the crop succession and with the highest quantity possible.

[b]from inventory (Dhaouadi, 2014).

[c]using all EOM quantity available on the territory and mean CS, NS and NL efficiencies and rate.

6. Conclusion

Simulation of EOM use scenarios on all the potentially amended crop plots of a region were used to study the impacts of insertion of EOM in the fertilisation practices, on C and N cycles taking into account CS, NS and NL. The factors which influence were considered were soil type, EOM characteristics, crop succession. All 20-y based simulations were realised under constant climatic and cropping systems conditions. The soil type was one of the most important drivers of CS and NL while NS was rather driven by crop succession. All effects were highly dependent on EOM types but also on quantity applied and frequency of application. EOM types had different effects depending on the quality of their OM, the amending EOM being composed with higher proportions of stable C and lower quantities of readily available N. However, taking also into account application constraints on P and N, quantity that can be applied also differ among EOM types enhancing or limiting these first differences. Amending EOM can generally be applied in higher quantity, increasing CS potentials compared to fertilising EOM, and limiting potential NS differences, primilarly caused by lower proportion of readily decomposable OM for amending EOM. Indeed, the green compost GWb allowed CS of up to 1.1 t C ha^{-1} yr^{-1}, while CS were generally lower than 0. 1 t C ha^{-1} yr^{-1} for fertilising EOM and NS could reach up to 74 kg N ha^{-1} yr^{-1} for both the green compost GWa and the dried sewage sludge. In this context, higher quantities applied through higher number of applications also meant lower N substitution and higher N leaching relatively to the quantity of N applied with EOM. CS was maximum in soils with low SOC contents or in clayey and calcareous soils, the OM protection effect of which being even more efficient with stable amending EOM. Arenic cambisol caused the largest NL especially when amended with manures or unstable composts which caused the highest potential surplus NL up to 47 kg N ha^{-1} yr^{-1}. The NS efficiency i.e. relatively to the amount of N brought with EOM was the highest when EOM were applied once in the succession, fertilising EOM on maize and amending EOM on rapeseed, especially these with low OM stability. This allowed NS efficiencies of up to 99% with the poultry litter and up to 90% with the compost MSW. Such results could be used in order to improve regional EOM use.

A large panel of EOM was studied here and laboratory characterisation seemed to have led to correct simulations. Regional simulations could be improved as suggested by a better spatially explicit characterisation of soil and crop production systems. However, such a study already constitutes a consequent source of information for policy-makers to improve regional EOM management. At a regional scale, many factors and potential scenarios of EOM use are combined in addition to a limited quantity of available EOM. Mathematical optimisation methods should be a helpful tool to manage this complexity. Depending on the objectives of EOM use i.e. carbon storage or mineral fertiliser substitution, the use of the available EOM over the region may be different, therefore multi-criteria and trading-off approaches should be led and more attention should be paid to the definition of the EOM uses objectives.

Acknowledgment

This study was carried out in the framework of both the Prostock-Gessol3 project supported by the French Environment and Energy Management Agency (ADEME) and the SOCSENSIT project supported by the LabeX BASC of Campus Paris Saclay. It also benefited financial and technical support from the French Space Agency (CNES) in the framework of the PLEIADES-CO TOSCA program. Special thanks to Véronique Etiévant, Dalila Hadjar, Joël Michelin, Jean-Noël Rampon, Christophe Labat for technical assistance.

7. Supplementary data

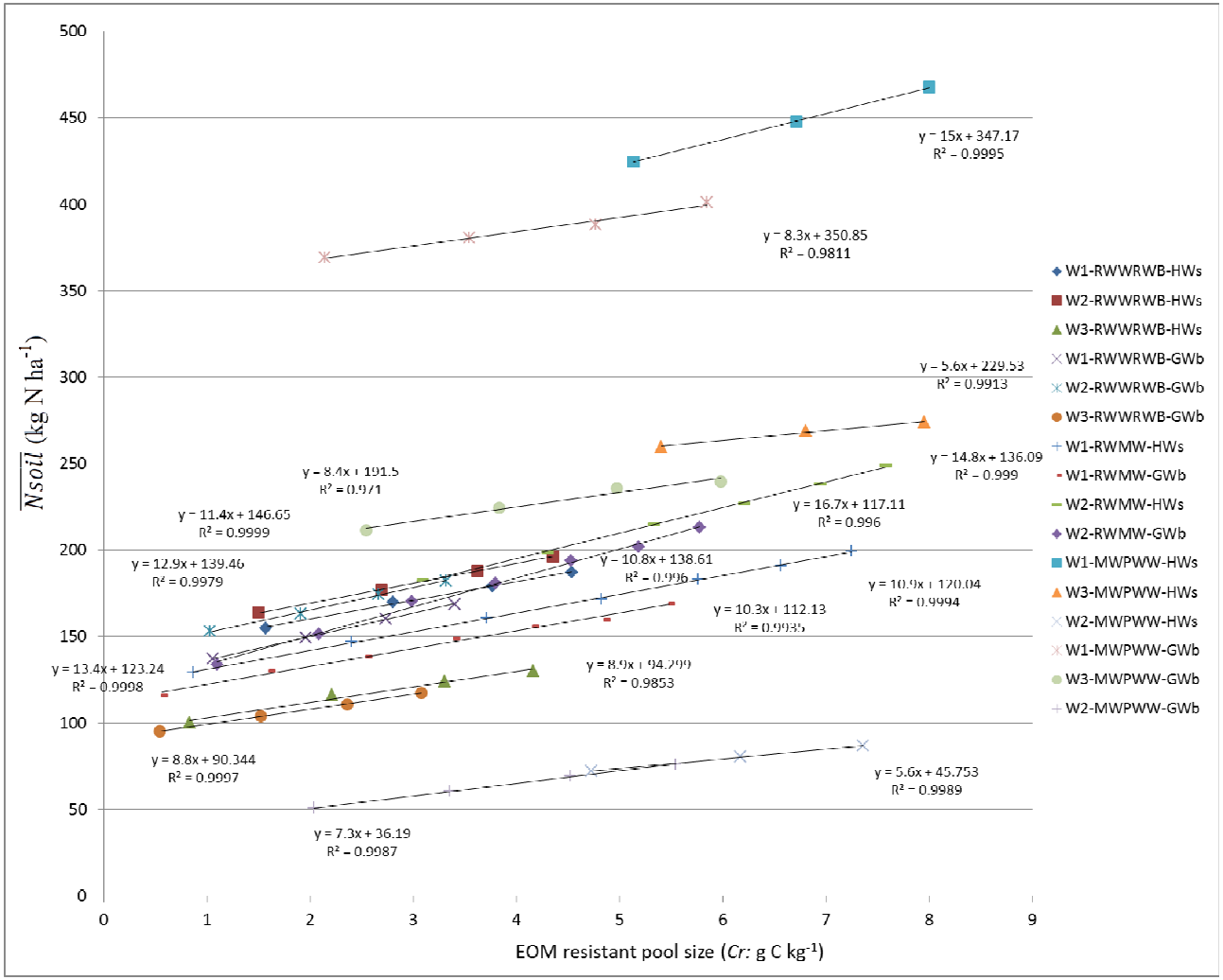


Figure 1S. Regressions for the "EOM resistant pool effect" of Nitrogen Savings calculation: examples of regressions made for wheat on soil 1.

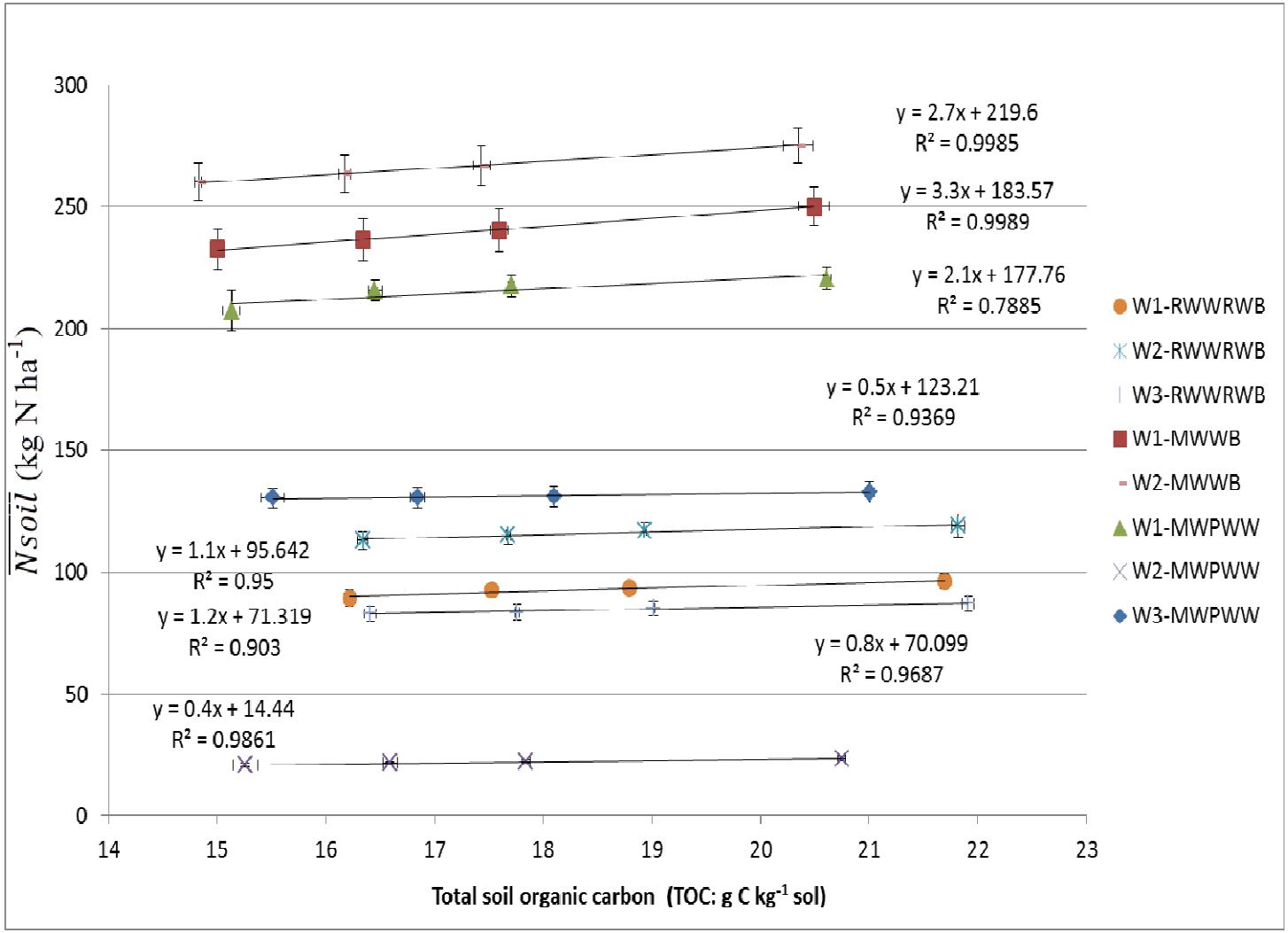


Figure 2S. Regressions for the "SOM pools effect" of Nitrogen Savings calculation: examples of regressions made for wheat on soil 1

8. References

AFNOR, 1995a. Norme NF ISO 10694. Qualité du sol - Dosage du carbone organique et du carbone total après combustion sèche, Qualité des sols, vol. 1. AFNOR, Paris.

AFNOR, 1995b. NF ISO 11263. Qualité du sol - Dosage du phosphore - Dosage spectrométrique du phosphore soluble dans une solution d'hydrogénocarbonate de sodium. AFNOR, Paris.

AFNOR, 1998. NF ISO 13878. Qualité du sol - Détermination de la teneur totale en azote par combustion sèche ("analyse élémentaire"), Qualité des sols, vol. 1. AFNOR, Paris.

AFNOR, 2009. Norme XP U 44-163. Amendements organiques et supports de culture - Caractérisation de la matière organique par la minéralisation potentielle du carbone et de l'azote. AFNOR, Paris.

Angers, D.A., Arrouays, D., Saby, N.P.A., Walter, C., 2011. Estimating and mapping the carbon saturation deficit of French agricultural topsoils. Soil Use Manag. 27, 448–452.

Ansari, A., Gill, S., Lanza, G., Rast, W., 2011. Eutrophication: Causes, Consequences and Control. Springer.

Barak, P., Molina, J.A.E., Hadas, A., Clapp, C.E., 1990. Optimization of an ecological model with the Marquardt algorithm. Ecol. Model. 51, 251–263.

Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. Eur. J. Soil Sci. 47, 151–163.

Baveye, P.C., Laba, M., 2015. Moving away from the geostatistical lamppost: Why, where, and how does the spatial heterogeneity of soils matter? Ecol. Model., Complexity of Soils and Hydrology in Ecosystems 298, 24–38.

Bhattacharyya, T., Pal, D.K., Easter, M., Williams, S., Paustian, K., Milne, E., Chandran, P., Ray, S.K., Mandal, C., Coleman, K., Falloon, P., Powlson, D., Gajbhiye, K.S., 2007. Evaluating the Century C model using long-term fertilizer trials in the Indo-Gangetic Plains, India. Agric. Ecosyst. Environ., Soil carbon stocks at regional scalesAssessment of Soil Organic Carbon Stocks and Change at National Scale, Final Project Presentation, The United Nations Environment Programme, Nairobi, Kenya, 23-24 May 2005 122, 73–83.

Brisson, N., Launay, M., Mary, B., Beaudouin, N., 2009. Conceptual basis, formalisations and parameterization of the STICS crop model. Quae.

Bruun, S., Christensen, B.T., Hansen, E.M., Magid, J., Jensen, L.S., 2003. Calibration and validation of the soil organic matter dynamics of the Daisy model with data from the Askov long-term experiments. Soil Biol. Biochem. 35, 67–76.

Castillon, P., Denoroy, P., Duval, R., Eveillard, P., Le Souder, C., Villette, C., 2009. Grille de calcul PKMG. Le groupe COMIFER 2007 et 2009.

Chalhoub, M., 2010. Effet de l'apport de composts sur la dynamique hydrique du sol, la disponibilité de l'azote pour la plante et le lessivage du nitrate : cas d'un sol limoneux cultivé du bassin parisien. Institut national agronomique paris-grignon - INA P-G.

Chalhoub, M., Garnier, P., Coquet, Y., Mary, B., Lafolie, F., Houot, S., 2013. Increased nitrogen availability in soil after repeated compost applications: Use of the PASTIS model to separate short and long-term effects. Soil Biol. Biochem. 65, 144–157.

Chaussod, R., Nicolardot, B., Catroux, G., Chretien, J., 1986. Relations entre les caractéristiques physicochimiques et microbiologiques de quelques sols cultivés. Sci. Sol 213–226.

Ciais, P., Wattenbach, M., Vuichard, N., Smith, P., Piao, S.L., Don, A., Luyssaert, S., Janssens, I.A., Bondeau, A., Dechow, R., Leip, A., Smith, P., Beer, C., Van Der Werf, G.R., Gervois, S., Van Oost, K., Tomelleri, E., Freibauer, A., Schulze, E.D., Carboeurope Synthesis Team, 2010. The European carbon balance. Part 2: croplands. Glob. Change Biol. 16, 1409–1428.

Coleman, K., Jenkinson, D.S., Crocker, G.J., Grace, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter, D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 29–44.

Comifer, 2013. Calcul de la fertilisation azotée Guide méthodologique pour l'établissement des prescriptions locales. Comifer.

Corbeels, M., Hofman, G., Cleemput, O.V., 1999. Simulation of net N immobilisation and mineralisation in substrate-amended soils by the NCSOIL computer model. Biol. Fertil. Soils 28, 422–430.

Crahet, M., 1992. Soil Map of the Versailles Plain. Scale 1:50 000 (Internal report). Institut National Agronomique Paris-Grignon.

Delphin, J.-E., Huck, C., Tiquet, J.-L., 1986. Evaluation du pouvoir minéralisateur de sols agricoles en fonction de leurs caractéristiques physico-chimiques. Agronomie 6, 453–458.

Dhaouadi, A.K., 2014. Insertion des Produits Résiduaires Organiques dans les systèmes de culture : Cas des systèmes céréaliers de la Plaine de Versailles et du Plateau des Alluets. Paris, AgroParisTech.

Doetterl, S., Six, J., Van Wesemael, B., Van Oost, K., 2012. Carbon cycling in eroding landscapes: geomorphic controls on soil organic C pool composition and C stabilization. Glob. Change Biol. 18, 2218–2232.

Edmeades, D.C., 2003. The long-term effects of manures and fertilisers on soil productivity and quality: a review. Nutr. Cycl. Agroecosystems 66, 165–180.

Exbrayat, J.-F., Pitman, A.J., Abramowitz, G., 2014. Response of microbial decomposition to spin-up explains CMIP5 soil carbon range until 2100. Geosci. Model Dev. 7, 2683–2692.

Fang, Y., Liu, C., Leung, L.R., 2015. Accelerating the spin-up of the coupled carbon and nitrogen cycle model in CLM4. Geosci. Model Dev. 8, 781–789.

Franko, U., Crocker, G.J., Grace, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter, D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using the CANDY model. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 109–120.

Gabrielle, B., Da-Silveira, J., Houot, S., Francou, C., 2004. Simulating Urban Waste Compost Effects on Carbon and Nitrogen Dynamics Using a Biochemical Index. J. Environ. Qual. 33, 2333.

Gabrielle, B., Da-Silveira, J., Houot, S., Michelin, J., 2005. Field-scale modelling of carbon and nitrogen dynamics in soils amended with urban waste composts. Agric. Ecosyst. Environ. 110, 289–299.

Gabrielle, B., Kengni, L., 1996. Analysis and field-evaluation of the CERES models' soil components: Nitrogen transfer and transformations. Soil Sci. Soc. Am. J. 60, 142–149.

Gabrielle, B., Laville, P., Duval, O., Nicoullaud, B., Germon, J.C., Henault, C., 2006. Process-based modeling of nitrous oxide emissions from wheat-cropped soils at the subregional scale. Glob. Biogeochem. Cycles 20, 1–13.

Gabrielle, B., Menasseri, S., Houot, S., 1995. Analysis and Field Evaluation of the Ceres Models Water Balance Component. Soil Sci. Soc. Am. J. 59, 1403–1412.

Gabrielle, B., Roche, R., Angas, P., Cantero-Martinez, C., Cosentino, L., Mantineo, M., Langensiepen, M., Henault, C., Laville, P., Nicoullaud, B., Gosse, G., 2002. A priori parameterisation of the CERES soil-crop models and tests against several European data sets. Agronomie 22, 119–132.

Goidts, E., van Wesemael, B., 2007. Regional assessment of soil organic carbon changes under agriculture in Southern Belgium (1955–2005). Geoderma 141, 341–354.

Hadas, A., Kautsky, L., Goek, M., Erman Kara, E., 2004. Rates of decomposition of plant residues and available nitrogen in soil, related to residue composition through simulation of carbon and nitrogen turnover. Soil Biol. Biochem. 36, 255–266.

Hadas, A., Molina, J. a. E., 1993. Simulation of nitrogen assimilation by heterotrophic soil microbial biomass. Physiol. Plant. 89, 664–668.

Hamiache, J., Bel, L., Vaudour, E., Gilliot, J.M., 2012. Spatial stochastic modeling of topsoil organic carbon content over a cultivated peri-urban region, using soil properties, soil types and a digital elevation model. Digit. Soil Assess. Beyond 161–164.

Houot, S., Pons, M.-N., Pradel, M., Caillaud, M.-A., Savini, I., Tibi, A., 2014. Valorisation des matières fertilisantes d'origine résiduaire sur les sols à usage agricole ou forestier. Impacts agronomiques, environnementaux, socio-économiques. Expertise scientifique collective. INRA-CNRS-Irstea. Synthèse 113pp (http://institut.inra.fr/Missions/Eclairer-les-decisions/Expertises/Toutes-les-actualites/Expertise-Mafor-effluents-boues-et-dechets-organiques#).

IPCC, 2006. Intergovernmental Panel on Climate Change 2006: Guidelines for National Greenhouse Gas Inventories, prepared by the National Greenhouse Gas Inventories Programme. Institute for Global Environmental Strategies (IGES), Japan.

Jandl, R., Rodeghiero, M., Martinez, C., Cotrufo, M.F., Bampa, F., van Wesemael, B., Harrison, R.B., Guerrini, I.A., Richter Jr., D. deB, Rustad, L., Lorenz, K., Chabbi, A., Miglietta, F., 2014.

Current status, uncertainty and future needs in soil organic carbon monitoring. Sci. Total Environ. 468–469, 376–383.

Jensen, L.S., Mueller, T., Nielsen, N.E., Hansen, S., Crocker, G.J., Grace, P.R., Klír, J., Körschens, M., Poulton, P.R., 1997. Simulating trends in soil organic carbon in long-term experiments using the soil-plant-atmosphere model DAISY. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 5–28.

Kelly, R.H., Parton, W.J., Crocker, G.J., Graced, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter, D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using the century model. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 75–90.

Koga, N., Smith, P., Yeluripati, J.B., Shirato, Y., Kimura, S.D., Nemoto, M., 2011. Estimating net primary production and annual plant carbon inputs, and modelling future changes in soil carbon stocks in arable farmlands of northern Japan. Agric. Ecosyst. Environ. 144, 51–60.

Lal, R., Follett, F., Stewart, B.A., Kimble, J.M., 2004. Soil carbon sequestration to mitigate climate change and advance food security. Soil Sci. 304, 1623–1627.

Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuriès, L., Chaussod, R., Guillotin, M.L., Linères, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S., 2010. Typology of exogenous organic matters based on chemical and biochemical composition to predict potential nitrogen mineralization. Bioresour. Technol. 101, 157–164.

Laville, P., Michelin, J., Houot, S., Gueudet, J.-C., Rampon, J.N., Labat, C., Vaudour, E., 2014. Soil N2O Emissions from Recovered Organic Waste Application in Versailles Plain (France): A Laboratory Approach. Waste Biomass Valorization 5, 515–527.

Lefèvre, R., Barré, P., Moyano, F.E., Christensen, B.T., Bardoux, G., Eglin, T., Girardin, C., Houot, S., Kätterer, T., van Oort, F., Chenu, C., 2014. Higher temperature sensitivity for stable than for labile soil organic carbon – Evidence from incubations of long-term bare fallow soils. Glob. Change Biol. 20, 633–640.

Lettens, S., Van Orshoven, J., van Wesemael, B., De Vos, B., Muys, B., 2005. Stocks and fluxes of soil organic carbon for landscape units in Belgium derived from heterogeneous data sets for 1990 and 2000. Geoderma 127, 11–23.

Marmo, L., Feix, I., Bourmeau, E., Amlinger, F., Bannick, C.G., De Neve, S., Favoino, E., Gendebien, A., Gibert, J., Givelet, M., Leifert, I., Morris, R., Rodriguez Cruz, A., Ruck, F., Siebert, S., Tittarelli, F., 2004. Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection, vol. 3, taskegroup 4 Exogenous Organic Matter.

Molina, J. a. E., Clapp, C.E., Shaffer, M.J., Chichester, F.W., Larson, W.E., 1983. NCSOIL, A Model of Nitrogen and Carbon Transformations in Soil: Description, Calibration, and Behavior. Soil Sci. Soc. Am. J. 47, 85–91.

Mondini, C., Coleman, K., Whitmore, A.P., 2012. Spatially explicit modelling of changes in soil organic C in agricultural soils in Italy, 2001-2100: Potential for compost amendment. Agric. Ecosyst. Environ. 153, 24–32.

Morvan, T., Nicolardot, B., 2009. Role of organic fractions on C decomposition and N mineralization of animal wastes in soil. Biol. Fertil. Soils 45, 477–486.

Morvan, T., Nicolardot, B., Péan, L., 2006. Biochemical composition and kinetics of C and N mineralization of animal wastes: a typological approach. Biol. Fertil. Soils 42, 513–522.

Mosier, A.R., 2001. Exchange of gaseous nitrogen compounds between agricultural systems and the atmosphere. Plant Soil 228, 17–27.

Nemecek, T., Kägi, T., 2007. Life cycle inventories of U.S. agricultural production system (No. 15b). Agrosope Reckenholz-Tänikon Research Station ART, Zürich and Dübendorf.

Némery, J., Garnier, J., Morel, C., 2005. Phosphorus budget in the Marne Watershed (France): urban vs. diffuse sources, dissolved vs. particulate forms. Biogeochemistry 72, 35–66.

Nicolardot, B., Molina, J.A.E., Allard, M.R., 1994. C and N fluxes between pools of soil organic matter: Model calibration with long-term incubation data. Soil Biol. Biochem. 26, 235–243.

Noirot-Cosson, P.E., Vaudour, E., Gilliot, J.M., Gabrielle, B., Houot, S., 20XXb. Parameterisation of the NCSOIL model to simulate short term C and N mineralisation of exogenous organic matters in different soils. Waste Manag. in prep.

Noirot-Cosson, P.E., Vaudour, E., Gilliot, J.M., Gabrielle, B., Houot, S., 2016. Modelling the long-term effect of urban waste compost applications on carbon and nitrogen dynamics in temperate cropland. Soil Biol. Biochem. 94, 138–153.

Paustian, K., Levine, E., Post, W.M., Ryzhova, I.M., 1997. The use of models to integrate information and understanding of soil C at the regional scale. Geoderma 79, 227–260.

Pellerin, S., Bamière, L., Angers, D., Béline, F., Benoît, M., Butault, J.P., Chenu, C., Colnenne-David, C., De Cara, S., Delame, N., Doreau, M., Dupraz, P., Faverdin, P., Garcia-Launay, F., Hassouna, M., Henault, C., Jeuffroy, M.H., Klumpp, K., Metay, A., Moran, D., Recous, S., Samson, E., Savini, I., Pardon, L., 2013. Quelle contribution de l'agriculture française à la réduction des émissions de gaz à effet de serre ? Potentiel d'atténuation et coût de dix actions techniques (Rapport d'étude). INRA, France.

Peltre, C., 2010. Potentialité de stockage de carbone dans les sols par apport de matières organiques exogènes. AgroParisTech, Paris.

Peltre, C., Christensen, B.T., Dragon, S., Icard, C., Katterer, T., Houot, S., 2012. RothC simulation of carbon accumulation in soil after repeated application of widely different organic amendments. Soil Biol. Biochem. 52, 49–60.

Petit, C., Aubry, C., 2014. Collecte de grandes cultures biologiques en Île-de-France. Économie Rurale 339-340, 33–47.

Rosen, C.J., Allan, D.L., 2007. Exploring the benefits of organic nutrient sources for crop production and soil quality. Horttechnology 17, 422–430.

Rusakova, I., 2015. Reducing nitrate leaching from soddy-pozolic sandy loam soil when applying poultry manure in combination with barley straw. Sci. Pape Rs Ser. Manag. Econ. Eng. Agric. Rural Dev. 15.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. Plant Soil 241, 155–176.

Sleutel, S., De Neve, S., Beheydt, D., Li, C., Hofman, G., 2006. Regional simulation of long-term organic carbon stock changes in cropland soils using the DNDC model: 1. Large-scale model validation against a spatially explicit data set. Soil Use Manag. 22, 342–351.

Sleutel, S., De Neve, S., Németh, T., Tóth, T., Hofman, G., 2006. Effect of manure and fertilizer application on the distribution of organic carbon in different soil fractions in long-term field experiments. Eur. J. Agron. 25, 280–288.

Sleutel, S., Neve, S.D., Hofman, G., 2007. Assessing causes of recent organic carbon losses from cropland soils by means of regional-scaled input balances for the case of Flanders (Belgium). Nutr. Cycl. Agroecosystems 78, 265–278.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J., 2008. Greenhouse gas mitigation in agriculture. Philos. Trans. R. Soc. Lond. B Biol. Sci. 363, 789–813.

Stockmann, U., Adams, M.A., Crawford, J.W., Field, D.J., Henakaarchchi, N., Jenkins, M., Minasny, B., McBratney, A.B., Courcelles, V. de R. de, Singh, K., Wheeler, I., Abbott, L., Angers, D.A., Baldock, J., Bird, M., Brookes, P.C., Chenu, C., Jastrow, J.D., Lal, R., Lehmann, J., O'Donnell, A.G., Parton, W.J., Whitehead, D., Zimmermann, M., 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. Agric. Ecosyst. Environ. 164, 80–99.

Stockmann, U., Padarian, J., McBratney, A., Minasny, B., de Brogniez, D., Montanarella, L., Hong, S.Y., Rawlins, B.G., Field, D.J., 2015. Global soil organic carbon assessment. Glob. Food Secur. 6, 9–16.

Sutton M.A., Bleeker A., Howard C.M., Bekunda M., Grizzetti B., de Vries W., van Grinsven H.J.M., Abrol Y.P., Adhya T.K., Billen G.,. Davidson E.A, Datta A., Diaz R., Erisman J.W., Liu X.J., Oenema O., Palm C., Raghuram N., Reis S., Scholz R.W., Sims T., Westhoek H. & Zhang F.S., with contributions from Ayyappan S., Bouwman A.F., Bustamante M., Fowler D., Galloway J.N., Gavito M.E., Garnier J., Greenwood S., Hellums D.T., Holland M., Hoysall C., Jaramillo V.J., Klimont Z., Ometto J.P., Pathak H., Plocq Fichelet V., Powlson D., Ramakrishna K., Roy A., Sanders K., Sharma C., Singh B., Singh U., Yan X.Y. & Zhang Y., 2013. Our nutrient world: the challenge to produce more food and energy with less pollution.

Glob. Overv. Nutr. Manag. Cent. Ecol. Hydrol. Edinb. Behalf Glob. Partnersh. Nutr. Manag. Int. Nitrogen Initiat.

Thuriès, L., Pansu, M., Larré-Larrouy, M.-C., Feller, C., 2002. Biochemical composition and mineralization kinetics of organic inputs in a sandy soil. Soil Biol. Biochem. 34, 239–250.

Tornquist, C.G., Gassman, P.W., Mielniczuk, J., Giasson, E., Campbell, T., 2009. Spatially explicit simulations of soil C dynamics in Southern Brazil: Integrating century and GIS with i_Century. Geoderma 150, 404–414.

Vaudour, E., Baghdadi, N., Gilliot, J.M., 2014. Mapping tillage operations over a peri-urban region using combined SPOT4 and ASAR/ENVISAT images. Int. J. Appl. Earth Obs. Geoinformation 28, 43–59.

Vaudour, E., Bel, L., Gilliot, J.M., Coquet, Y., Hadjar, D., Cambier, P., Michelin, J., Houot, S., 2013. Potential of SPOT Multispectral Satellite Images for Mapping Topsoil Organic Carbon Content over Peri-Urban Croplands. Soil Sci. Soc. Am. J. 77, 2122–2139.

Vaudour, E., Noirot-Cosson, P.E., Membrive, O., 2015. Early-season mapping of crops and cultural operations using very high spatial resolution Pléiades images. Int. J. Appl. Earth Obs. Geoinformation 42, 128–141.

Viaud, V., Angers, D.A., Walter, C., 2010. Toward Landscape-Scale Modeling of Soil Organic Matter Dynamics in Agroecosystems. Soil Sci. Soc. Am. J. 74, 1847–1860.

Wiesmeier, M., Lützow, M. von, Spörlein, P., Geuß, U., Hangen, E., Reischl, A., Schilling, B., Kögel-Knabner, I., 2015. Land use effects on organic carbon storage in soils of Bavaria: The importance of soil types. Soil Tillage Res. 146, Part B, 296–302.

Xu, S., Shi, X., Zhao, Y., Yu, D., Wang, S., Tan, M., Sun, W., Li, C., 2012. Spatially explicit simulation of soil organic carbon dynamics in China's paddy soils. Catena 92, 113–121.

Yagasaki, Y., Shirato, Y., 2014. Assessment on the rates and potentials of soil organic carbon sequestration in agricultural lands in Japan using a process-based model and spatially explicit land-use change inventories - Part 2: Future potentials. Biogeosciences 11, 4443–4457.

Yokozawa, M., Shirato, Y., Sakamoto, T., Yonemura, S., Nakai, M., Ohkura, T., 2010. Use of the RothC model to estimate the carbon sequestration potential of organic matter application in Japanese arable soils. Soil Sci. Plant Nutr. 56, 168–176.

Dans ce troisième article, nous avons réalisés des scenarios d'amendements organiques se voulant réalistes : techniquement, car tenant compte des contraintes des agriculteurs et s'inspirant de leurs pratiques, environnementalement, leurs quantités d'apports étant contraintes pour prévenir pollutions azotées, phosphorées et aux éléments traces métalliques. Une modélisation a également été réalisée pour réduire la fertilisation azotée de synthèse compte tenu de l'azote apportée via les amendements organiques, réellement disponible pour la plante. De plus, le territoire a été cartographié en termes de types de sols, successions culturales, teneurs en carbone organique. Ainsi, les impacts sur le stockage de carbone, les économies en engrais de synthèse et les lixiviations de nitrates de nombreux scenarios d'amendements organiques ont pu être évalués dans les toutes les conditions de sol et de productions culturales du territoire.

Les PRO constituent une ressource limitée à l'échelle d'un territoire, car difficilement transportable sur de longues distances, et les conditions variées à cette même échelle de leur utilisations permettent des bénéfices ou des coûts variés, ainsi une distribution intelligente des PRO disponibles sur le territoire pourrait accroitre ces bénéfices ou limiter ces coûts.

Chapter 5. Optimisation of Exogenous Organic Matter use by agriculture at the regional scale: case study of a peri-urban area near Paris, France

P.E. NOIROT-COSSON[1], VAUDOUR E.[1], GILLIOT J.M.[1], GABRIELLE B.[1], LUBBERS M.[2], BAMIERE L.[3], HOUOT S.[1]

[1] UMR ECOSYS, AgroParisTech, INRA, Université Paris-Saclay, 78850, Thiverval-Grignon

[2] Plant Production Systems, Wageningen University, P.O. Box 430, 6700 AK Wageningen

[3] INRA - UMR210, Economie Publique Research Unit, 78850, Thiverval-Grignon

1. Abstract

The use of Exogenous Organic Matter (EOM) in crop production might be a promising way to improve agriculture sustainability as it may enhance crop productivity while improving resource use efficiency. It may also cause negative impacts on the environment, that should be limited. The aims of this paper were (i) to explore whether there is room for benefit improvement or damage limitation related with EOM use in agriculture, thanks to an optimised regional distribution of EOM resource, and (ii) to understand the drivers of this optimised distribution in order to facilitate future improvements in regional management. Based on simulated impacts of EOM application scenarios, a Mixed Integer Linear Programming (MILP) model was developed in order to select the best EOM uses for all crop plots within a 221 km²-region. Maximising carbon storage (CS) and synthetic nitrogen saving (NS) at the regional scale required to apply EOM on all crop plots but the former would need to apply "amending" EOM preferentially on soils with high CS potential, and the latter would need to apply "fertilising" EOM preferentially on crop successions with maize and less stable amending EOM on crop successions with rapeseed. These two optimisations respectively led to storage of 0.47 t C yr^{-1} ha^{-1} and savings of 53 kg N yr^{-1} ha^{-1} of synthetic fertiliser compared to control (no EOM application). The minimal additional Nitrate Leaching (NL) was reached with occasional use of EOM which only decreased NL by less than 1 kg N yr^{-1} ha^{-1} compared to control. This study showed that higher benefits can be reached thanks to such optimised management and it also showed how EOM should be distributed in the region depending on the benefits to be improved.

2. Introduction

Soils constitute the largest terrestrial reservoir of carbon with 1500 to 2400 Gt (Ciais et al., 2013), and therefore represent a determinant driver for climate change mitigation or enhancement through C storage or destocking in/from soil (Lal et al., 2004). Over the last fifty years, agriculture intensification has led to a number of environmental consequences including soil organic matter (SOM) declining (Matson et al., 1997). This is the case in the large Paris area characterised by croplands with no livestock activity which effluents and manures could maintain SOM through their use as organic fertilisers or amendments (Petit and Aubry, 2014). The SOM decrease has become a major threat for soil quality (European Commission, 2006) as it leads to the reduction of soil fertility by impacting chemical, biological and physical soil properties (Ciais et al., 2010; Jandl et al., 2014).

Exogenous Organic Matter (EOM), originating from crop residues or animal manures from agriculture, industries and urban areas can be applied on cultivated land and favour accumulation of Soil Organic Carbon (SOC) (Marmo et al., 2004). In addition, applications of EOM provide Nitrogen (N) and Phosphorus (P) supply which is otherwise brought from mineral fertilisers. Mineral N fertiliser is produced thanks to a fossil-fuel consuming process converting atmospheric N into plant available N forms, while P needs to be extracted from mines which is also energy consumptive and additionally issued from non-renewable ressources (Matson et al., 1997). EOM application is a mean to enhance soil fertility and in overall soil quality (Davies and Lennartsson, 2005; Diacono and Montemurro, 2010). For all these reasons, EOM use by agriculture as amendments or fertilisers is encouraged in Europe (European Commission, 2010), and particularly in France, through the Grenelle 2 law for environmental protection (article 204-I), the energy transition law for green growth (article 7-V) and international research program called “4 per 1000” launched in 2015. In peri-urban areas such as in the Plain of Versailles, EOM and especially from urban activities are abundant and can be used instead of the rarefied livestock effluents for SOC enhancement and mineral fertiliser substitution. Despite these benefits, EOM applications, may also damage the environment and especially water quality in case of excessive N leaching and P runoff (Edmeades, 2003) and also air quality because of NH_3, N_2O emissions. Heavy metals are generally more concentrated in EOM than in soil, therefore repeated amendments may also result in metals accumulations in soil (Zhang et al., 2006) but fluxes are low enough and there is no tangible evidence demonstrating negative impacts of heavy metals applied to soil (Diacono and Montemurro, 2010). These impacts depend on EOM characteristics, which vary according to the initial products the EOMs are made from, their composition, their proportion in the mixture, and the transformation process (Francou et al., 2008; Askri, 2015). Composting is one of the existing processes used to transform EOM. It presents numerous advantages: it reduces volume and masse becoming easier to transport, it stabilises

OM, thus enhancing the potential for carbon storage in soils, it concentrates mineral elements, it destroys pathogens (Francou, 2003). These element concentration and OM stabilisation also impact N outputs through decreasing crop availability, potential leaching, and use of composted EOM rather than fresh products should be assessed for these various impacts and integrated in the whole crop production system.

Benefits and costs of the EOM application practice depend on several other factors, all spatially variable at a regional scale: (i) environmental factors such as soil characteristics and climate (Valé, 2006; Rousk et al., 2009) (ii) cropping system characteristics such as crop successions, fertilisation, intercrops, residue management, soil management that influence SOM (Alvarez, 2005; Ogle et al., 2005; Ladha et al., 2011) and (iii) the EOM type applied, the quantity and frequency of application (Bastida et al., 2008 Val-Moraes et al., 2011). Moreover, EOM production is generally localised, within livestock farms, composting plants, waste-water treatment plants whereas areas of application may be widely spread over thousands hectares. This is why EOM distribution management must be thought and assessed at a regional level.

Some of the various drivers of EOM impacts on environment may be taken into account through mechanistic models which simulate C and N fluxes in the soil-plant-atmosphere system after EOM application in variable conditions and for a wide range of EOM types (Smith et al., 1997; Franko et al., 1997; Kelly et al., 1997; Gabrielle et al., 2004). Provided their calibration, these models can also quantify these impacts and therefore help to assess the practice and decision-making towards an improved management. While many previous studies focused on organic carbon fate in soil using the models Roth-C (Guo et al., 2007; Peltre et al., 2012), Century (Falloon et al., 2002), DAISY (Bruun et al., 2003), few of them studied the N cycle (Chalhoub et al., 2013), but to the best of our knowledge, none simultaneously took P issues into account. Models are mostly used for the purpose of better understanding OM dynamics, but quite seldom to explore potentials and risks of innovative practices.

Indeed, managing EOM at a regional scale i.e. distributing a limited quantity of EOM to many farm plots characterised by their soil type, their soil organic matter, their crop succession, their surface, with various objectives such as carbon storage or nitrogen saving, constitutes a very-complex problem that can be solved with model-based tools composing a decision support system using linear programming model(s) as backbone. According to Makowski et al. (2000), linear programming model is defined by an ‘objective function’ that is maximised or minimised taking into account ‘constraints’ and determining optimal solutions among ‘decision variables’. In linear programming models developed for exploring land use or land management options to support re-orientation of farming systems, the objective function can represent an economic, an agricultural, or an environmental objective and several agricultural and environmental objective functions may be maximised or minimised, within a multi-objective modelling approach (de Wit et al., 1988; Ten Berge et al., 2000; Castelán-Ortega et al., 2003; Pacini et al., 2004; Dogliotti et al., 2005). ‘Constraints’ can be related to land capabilities (through soil types) and resources (water for irrigation, labour) or product balances (e.g. self-

sufficiency for agricultural products). Decision variables are typically areas allocated to production activities such as crops or animal husbandry: this article was original in considering allocation of available EOMs on crop plots of a region.

The aim of this paper was to evaluate the potentials and risks of various strategies of EOM distribution at the regional scale over the Versailles Plain covering 221 km², for various objectives : Carbon Storage (CS), the substitution of mineral N fertiliser from industrial origin or "Nitrogen Saving" (NS) and Nitrate Leaching (NL). This assessment was made for the actual EOM availability and for a second EOM availability situation for which some EOM were composted. This paper also aimed at understanding the drivers of the obtained distribution patterns.

3. Material & methods

3.1. Study area

Located about 30 km west from Paris, the Versailles plain and the Alluets plateau area (48°46'-48°56' N; 1°50'-2°07' E) represents over 22 145 ha which 9907 ha are cropped. Crop successions are mainly composed with Rapeseed (R), winter Wheat (W), spring Barley (B), Maize (M) or sometimes dry Pea (P) or fodder crops and mostly conventionally managed (Vaudour et al., 2015). These crops are grown on haplic luvisols formed on plateaus, calcaric, rendzic cambisols developed on plateau flanks and colluvic cambisols (stagnic) originated from loessic colluvium and/or sands on lower slopes and at valley bottoms (Crahet, 1992; Vaudour et al., 2014, 2013) according to the WRB classification (World Reference Base for Soil Resources, 2014). The SOC contents range from 4 to 32 g $C.kg^{-1}$ (Hamiache et al., 2012).

3.2. Spatially-explicit simulations of EOM applications

Spatially-explicit simulations of EOM application were carried out over the study area in the timespan of a 20y period and full detail of these simulations can be found in Noirot-Cosson et al. (20XXb).

3.2.1. Spatially-explicit data for model parameterisation

A set of several data layers were gathered and structured into ArcMap10.3®. Four main soil types were extracted from the soil map of Crahet (1992): (1) the calcaric, rendzic cambisol, (2) the arenic, podzolic cambisol, (3) the haplic luvisol from loess deposits over limestone, (4) the haplic luvisol (retic) from loess deposits over clay material. For each soil type, four classes of SOC content were based on the quartiles defined from the map of SOC content (Hamiache et al. 2012). Seven most representative crop successions were selected from the map of crop successions obtained by intersecting four land parcel register layers between 2007 and 2011. The crop successions selected over the 4 main soil types represented 7498 ha (76% of the total cropped surface). Regional results

will correspond to integrated results over this surface. Each crop plot was characterised both by its area and by one of the 112 triplet combinations of soil type, SOC content, crop succession retrieved thoughout the study region.

3.2.2. Scenario simulations of EOM use

The scenarios of EOM use are summarised in Table 1S (Noirot-Cosson et al., 20XXb). These scenarios considered 14 EOM types available or potentially available in the region: the C and N dynamics of each EOM were characterised and parameterised thanks to laboratory incubations and biochemical fractionations (Table 2S; Noirot-Cosson et al. 20XXa). Scenarios were defined by EOM types, EOM application dates, EOM quantity and their number of applications within the crop succession, and also included control scenario where no EOM was applied but standard mineral fertilisation. They were set differently according to two distinct groups of EOMs: the "fertilising EOMs" such as litters, slurries, sludges characterised by a large availability of N for crops immediately after EOM application and the "amending" EOM comprising composts and manures used as organic amendments for their potential efficiency at restoring or increasing SOM. Fertilising EOMs were applied in spring, one day before the mineral fertilisation in February, March or May for wheat, barley or maize, respectively while amending EOMs were applied end of August. Quantity and maximum number of applications within the crop succession were specific to each crop succession and only one application per year of cultivation was tested. In order to create realistic scenarios and avoid pollutions, amendment quantities were constrained based on quantities of N and P applied. First the total quantity that could be applied on a crop succession was limited to the crop needs in P for the whole crop succession. Based on the 91/676CEE "nitrate directive", the rate of EOM applications should not exceed 170 kg N ha^{-1} per application. However,, 2 rates of EOM applications were considered for fertilising EOMs (170 kg N ha^{-1} or 250 kg N ha^{-1}) and only the 250 kg N ha^{-1} constraint was tested for amending EOMs. All possible numbers of applications within the crop succession were tested, and the total EOM quantity that could be applied (based on the P constraint) were equally split between the different applications, the N constraint being applied after this sharing. The maximum number of applications was reached when N per application as defined by total N potentially applied according to P constraint and divided by the number of applications was below the maximum rate of N per application chosen as 170 kg N ha^{-1} for fertilising EOMs and 250 kg N ha^{-1} for amending EOMs (Table 1S).

3.2.3. Simulations with the CERES-EGC crop model

The CERES-EGC (Gabrielle et al., 1995, 2002) crop model was chosen to simulate the impacts of 20 years of EOM applications on C and N cycles. The NCSOIL (Molina et al., 1983; Barak et al., 1990; Hadas and Molina, 1993) model has been coupled to the CERES-EGC crop model in order to better

simulate OM fate in soil in terms of C and N dynamics. Environmental outputs such as nitrate leaching and gaseous emissions of ammonia, N_2O and N_2 could be calculated (Gabrielle et al., 2005, 2006)

A new module in CERES-EGC allowing to calculate the quantity of synthetic N fertiliser that could be saved consequently to the increase of SOM and/or to EOM applications was implemented (Fig 1S; Noirot-Cosson et al., 20XXb). This module distinguished direct and indirect effects of EOM application. The direct effect consisted in a direct substitution of a quantity of synthetic N fertiliser equal to the quickly available EOM-N when applied in spring for fertilising EOMs or if applied in september for amending EOMs (only on rapeseed). This quickly available N corresponds to mineral N and the labile i.e. the quickly degradable EOM organic pool defined in the EOM parameterisation (Table 3S; Noirot-Cosson et al., 20XXa). The indirect effects of SOM increase and accumulated resistant fraction of EOM were modelled by means of regressions of a surplus of N supply over a given depth of 120 cm related to the surplus of OC in soil compared to a reference situation: either the surplus of TOC (defined as SOC content in soil pools besides the EOM pools) compared to the lowest initial TOC ($TOCini_{min}$, specific of a soil type) for the SOM increase effect, or the quantity of exogenous OC in the resistant pool for the indirect EOM effect. Each regression was specific to a given 'soil type x crop type' doublet.

3.3. Optimisation of EOM distribution

3.3.1. Preparation of optimisation entries

Simulations of all possible EOM use scenarios for every 'soil type × TOCini × crop succession' triplet were performed. By using Microsoft ACCESS2010®, the dataset characterising all 972 crop plots (area, triplet) of the region was completed with the simulated outputs, resulting in a new dataset with 31205 rows each composed with 'crop plot and characteristics', 'scenario characteristics', 'EOM input quantity', 'ouput impacts'). The outputs integrated over the plot area for each plot were: Carbon Storage (t C) and Nitrogen Saving, Nitrate Leaching, Ammonia (NH_3), Nitrous oxides (N_2O) and Gaseous N (N_2) emissions (kg N). Data concerning EOM input for each plot and each scenario were also included: total quantity applied in Mg DM. This new dataset of crop plots thus associated inputs and outputs for all EOM use scenarios and constituted entries for optimisation.

3.3.2. Optimisation model

Following a number of regional objectives and limitations imposed by EOM availability in the region and constraints for EOM application in the scenarios, the optimisation procedure was designed to select the most appropriate EOM amendment scenario for each crop plot of the region. For this purpose, a Mixed Integer Linear Programming model (MILP), named "EOM attribution", was developed. The "EOM attribution" model optimally distributed available EOMs across the study area

in order to either maximise CS, or NS, or minimise NL at this regional scale. "EOM attribution" was defined by Eq. [1] to [3] as follows:

Min or Max{ $z = \sum_{p,j,k,m}(x_{p,j,k,m} \times O_{p,j,k,m})$ } [1]

$\sum_{p,j,k,m} (x_{p,j,k,m} \times I_{p,j,k,m}) \leq Q_{(j)}$ [2]

For all p, $\sum_{j,k,m}(x_{p,j,k,m}) = 1$ [3]

Scalar z is the objective function i.e. regional CS or NS or NL; x is a binary variables; O represents the output in terms of CS, NS or NL of the simulated scenario for the crop plot p, using the EOM type j, k times within the crop succession and bringing an EOM quantity m; I the inputs in terms of EOM quantity used for the related scenario, Q the constraint of the available EOM quantity in the region.
Two situations of EOM availability were considered: (A) actual annual EOM production, dominated by green waste composts, cattle and horse manures, sewage sludge (Dhaouadi, 2014); (B) anticipation of a possible further evolution with horse manures (HM), poultry litters (PL) and sewage sludges (SS) composted, SS being co-composted with green waste, implying a lower production of green waste compost. The MILP model was written, compiled and solved using GAMS24.2®.

3.3.3. EOM availability situations

Among the presently available EOMs, three EOMs could be composted, the remaining being already composted or matured products. Quantities of potential composts of HM or PL and co-compost of green waste and sewage sludges (GWS) were calculated from quantities of raw EOM and composting mass reduction factors. The calculation procedure is presented in Fig. 1 and factors either found in litterature or calculated are described in Table 1. Eventually, mixed proportions in DM of 13% of sewage sludge, 87% of green wastes were considered (Doublet et al., 2011), to make GWS compost.

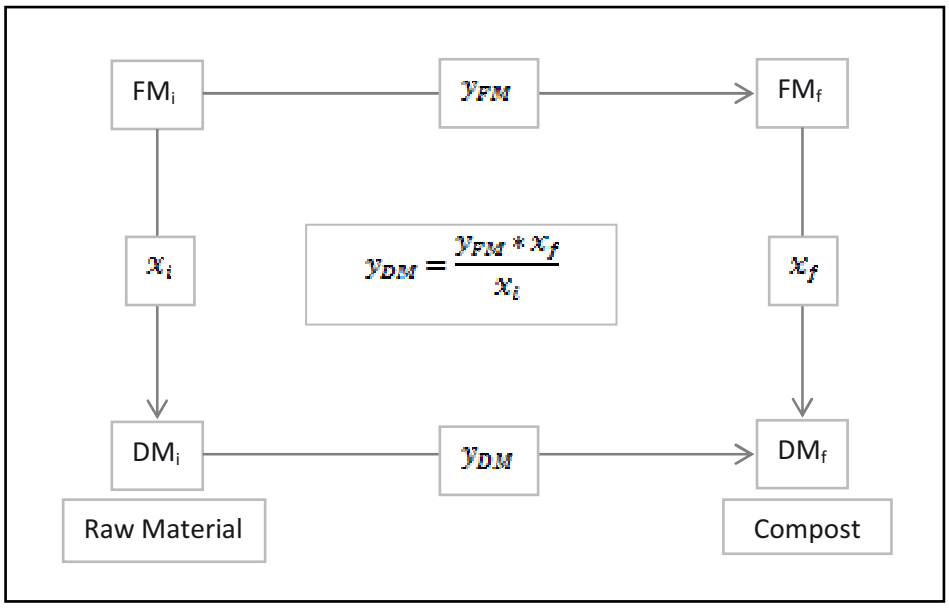


Figure 1. Scheme of calculation for the composting mass reduction factor
x stands for the percentage of Dry Material (DM) in Fresh Matter (FM), y for the composting mass reduction factor, i for 'initial', f for 'final'.

Table 1. Conversion factors between fresh and dry matter for x, and between raw matter and composted matter for y

Material	x_i	x_f	y_{FM}	y_{DM}
Horse Manures	0.35[1]	0.44[1]	0.5[2]	0.63
Poultry Litters	0.65[3]	0.63[3]	0.5[3]	0.48
Green Wastes and Sewage Sludge	0.35[4]	0.63[4]	0.26[4]	0.47
Green Wastes	0.44[5]	0.63[6,7,8]	0.32[9]	0.46

[1]Dhaouadi, 2014; [2]Leclerc et al., 2012; [3]ITAVI, 2001; [4]Doublet et al., 2011; [5]Mauboussin, 2010; [6]Biomasse Normandie and Trivalor, 2000 ; [7]Leclerc, 2014; [8]Mathieu, 2008; [9]Barrault, 2012

3.3.4. Optimisation results analysis

Each optimisation resulted in a database formed by the 972 crop plots, associated to one selected scenario and its related inputs and outputs. Optimisation results were mapped after joining the obtained database to the plots shapefile based on the plot identifyer.

As one of the main objectives was assessing the potentials/risks of EOM regional distribution on CS, NS, and N outputs, we first analysed optimisation results in terms of regional output summaries. Then we analysed the results in terms of scenarios favoured by optimisation in accordance to spatial characteristics, objective pursued and constraints applied. Therefore, an overall examination of the regional EOM distribution was performed, particularly spatial variations in EOM type distribution according to soil types and crop succession, then the frequency of applications for each EOM type was studied. Eventually maps of EOM distribution over the regional crop plots were observed. In Noirot-Cosson et al. (20XXb), simulations of all EOM scenarios for each 'soil type×TOCini×crop succession' triplets evidenced high pedological influence on CS, NS and NL and some influence of crop

succession (mostly NS and NL) and EOM types. Therefore, the optimised EOM distribution according to soil type was examined for all objectives while distribution according to crop succession was analysed for NS maximisation and NL minimisation. Optimisation procedures and results analysis are summarised in Fig. 2.

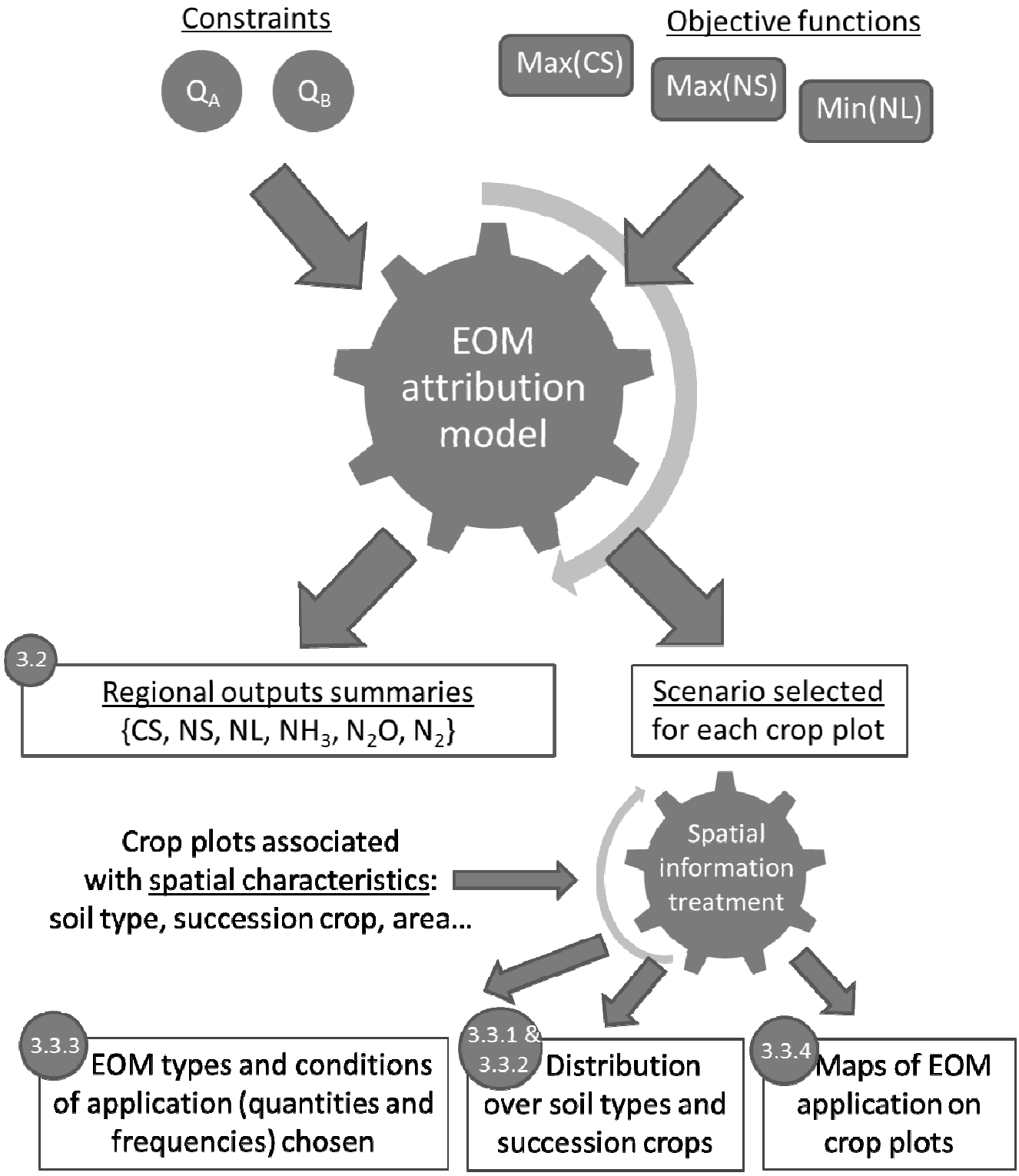


Figure 2. EOM attribution model use and results treatments.
Q_A and Q_B stand for the available quantity of EOM in situation A and B respectively, Max and Min for maximisation and minimisation procedures.
Among the regional outputs: CS for Carbon Storage, NS for Nitrogen Saving, NL for Nitrate Leaching, NH_3, N_2O and N_2 for ammonia, nitrous oxyde and nitrogen gas emissions respectively.

4. Results

4.1. EOM availability situations

For situation A, 78.3% of the available EOMs originated from urban activities while 21.7% were animal-derived EOMs (Table 2). Urban EOMs were mostly composts made with green wastes, such as GWa, GWb, Fertylis. The potential production of Biowaste compost (Bio) was calculated based on statistic of organic waste production by population and production rate of compost from the organic fraction of municipal solid wastes (Dhaouadi, 2014). Among EOMs derived from animal activities, HM (i.e. HMs and HMw) were the most abundant followed by cattle manure (CM). Their abundance was respectly explained by the important horse riding activity throughout the region and the existence of one big cattle experimental farm. In situation B, 12 396 t of GWS was made with 3429 t of SS and with green wastes instead of 10 070 t of GW composts, 2608 t of HMc from 4140 t of HM and 180 t of PLc from 375 t of PL. These transformations increased the proportion of urban EOM to 82.2%. Whatever situations, EOM mostly come from urban waste, although EOM derived from animal effluents were also abundant, an uncommon situation for a peri-urban area in the large Paris region.

Table 2. Available amounts of EOM for current situation (A) and with development of composting (B).

Available amounts (t DM yr^{-1})	Situation	
EOM types	A	B
Fertylis[a]	6480	6480
Dried Sewage Sludge (SSd)	3204	0
Limed Sewage Sludge (SSl)	311	86
Green Waste compost « BYS » (GWb)	5020	0
Green Waste compost « Sepur » (GWa)	5050	0
Green Waste and Sewage Sludge (GWS)	0	12396
Biowaste compost (Bio)	4080	4080
Municipal Solid Waste (MSW)	0	0
Urban EOM	24145	23042
Humival[b]	700	700
Poultry Litter (PL)	375	0
Poultry Litter Composted (PLc)	35	215
Horse Manures with Straw (HMs)	3672	0
Horse Manure with Wooden shavings (HMw)	468	0
Cattle Manure (CM)	1450	1450
Horse Manure Composted (HMc)	0	2608
EOM from animal origin	6700	4973
Total	30845	28015

[a] mixture of GW composts from the territory and imported bone meal.
[b] imported dried pig slurry.

4.2. Potential impacts related to EOM use at the regional scale

All impact results related with EOM uses for the overall region are summarised in Table 3 and the quantities of each EOM type used in Table 4. Control results, i.e. with no EOM application on any crop plot of the region led to a positive CS of 0.23 t C yr^{-1} ha^{-1}. This was mostly due to crop production systems which included systematic intercrop before spring crop, and residue incorporation after harvest. NS were also positive. First because NS calculation was based on a standard mineral N fertilisation considered adapted to N crop needs for the $TOCini_{min}$ in each soil, secondly the increase of CS led to indirect NS (Fig. 1S). Therefore, soil with higher TOCini led to positive NS.

Table 3. Regional annual results of Carbon Storage (CS), Nitrogen Savings (NS), Nitrate Leaching (NL), Ammonia emissions (NH3), Nitrous protoxyde emissions (N2O), di-nitrogen gas emissions (N2) issued from the simulations corresponding to different optimisation objectives and scenarios of available EOMs.

		Objectives	CS t C yr^{-1}	NS	NL	NH_3	N_2O	N_2
				kg N yr^{-1}				
Situation		Control	1756	2.83E+04	2.23E+04	3.02E+04	8.14E+03	1.38E+04
A		Max CS	5270	2.47E+05	2.64E+04	1.33E+04	9.09E+03	1.54E+04
		Max NS	3460	4.28E+05	2.67E+04	4.54E+03	8.74E+03	1.48E+04
		Min NL	2647	1.06E+05	2.21E+04	2.15E+04	8.20E+03	1.39E+04
B		Max CS	4508	2.13E+05	2.38E+04	1.51E+04	8.56E+03	1.45E+04
		Max NS	4168	2.49E+05	2.39E+04	1.30E+04	8.52E+03	1.45E+04
		Min NL	2566	9.84E+04	2.20E+04	2.20E+04	8.20E+03	1.39E+04
			t C yr^{-1} ha^{-1}	kg N yr^{-1} ha^{-1}				
		Control	0.23	3.77	29.74	4.03	1.0856	1.8412
A	relative to Control	Max CS	0.47	29.23	5.48	-2.25	0.1273	0.2143
		Max NS	0.23	53.32	5.90	-3.42	0.0796	0.1332
		Min NL	0.12	10.39	-0.29	-1.15	0.0080	0.0144
B		Max CS	0.37	24.63	1.99	-2.01	0.0566	0.0958
		Max NS	0.32	29.40	2.18	-2.30	0.0510	0.0863
		Min NL	0.11	9.35	-0.35	-1.10	0.0083	0.0150

When maximising CS or NS, 100% of the utilised area (7498 ha) received application of EOM while 65% of the surface (4863 ha) were left unamended when NL was minimised (Fig. 3) (situation A). Whatever the situation, maximising CS resulted in larger amounts of EOM used than maximising NS and than minimising NL, probably because of more amending EOM used. Indeed scenarios with amending EOMs always implied larger quantities, because less limited by P and N constraints.

Table 4. Quantities of available EOM over the territory for situations A and B, and proportions of EOM quantities used depending on objectives.

EOM	Total available EOM, situation A (t DM)	EOM used (% of available) situation A			Total available EOM, situation B (t DM)	EOM used (% of available) situation B		
		Max CS	Max NS	Min NL		Max CS	Max NS	Min NL
Humival	700	100%	34%	0%	700	0%	100%	0%
SSd	3204	13%	100%	2%	0			
PL	375	0%	100%	23%	0			
SSl	311	0%	0%	0%	86			
PLc	35	0%	0%	0%	215	0%	0%	0%
HMs	3672	100%	100%	0%	0			
HMw	468	100%	100%	0%	0			
GWb	5020	100%	100%	0%	0			
GWa	5050	100%	100%	0%	0			
CM	1450	100%	100%	0%	1450	100%	100%	0%
HMc	0				2608	100%	100%	0%
GWS	0				12396	43%	56%	28%
Bio	4080	100%	0%	0%	4080	100%	100%	68%
Fertylis	6480	100%	0%	94%	6480	100%	37%	30%
Total quantity used (t DM)		27327	19477	6256		23357	21344	8180

DM stands for Dry Matter. Max and Min for maximisation and minimisation procedure. CS, NS, NL for Carbon Storage, Nitrogen Saving and Nitrate Leaching respectively. Among EOM Humival stands for the dried pig slurry, SSd for dried Sewage Sludge, PL for Poultry Litter, SSl for limed Sewage Sludge, PLc for Poultry Litter composted, HMs for Horse Manure with Straw, HMw for Horse Manure with wooden shavings, GWb for Green Waste compost "BYS", GWa for Green Waste compost "Sepur", CM for Cattle Manure, HMc for Horse Manure composted, GWS for co-compost of green waste and sewage sludge, Bio for Biowaste compost and Fertylis a green waste compost mixed with bone meal.

4.2.1. Optimised distributions of current EOM (situation A)

When maximising CS, the additional storage relative to control was 0.47 t C yr^{-1} ha^{-1}. It was larger than the achieved storage when maximising NS (CS, 0.23 t C yr^{-1} ha^{-1}). All amending EOMs were used and among the fertilising EOMs 100% of Humival was used but only 13% of SSd, PL, PLc and SSl being not used (Table 4). CS maximisation favoured the use of either EOMs whose composition was characterised by more stable OM or of EOMs which enabled larger inputs of C than EOM not used.

When maximising NS, 53 kg N yr^{-1} ha^{-1} were saved compared to control. This was higher than NS obtained when maximising CS with 29 kg N yr^{-1} ha^{-1}. For NS maximisation, fertilising EOMs except SSl and PLc, and the amending EOMs except Bio and Fertylis were used (Table 4). These EOM were not chosen despite low C/N probably because of high P contents along with low OC contents especially for SSl and PLc, allowing only little quantity of N applicable considering the P contraint, and because of high stability concerning Bio and Fertylis, allowing only few N release and thus low NS.

When minimising NL, NL was similar but only slightly lower than control (-0.3 kg N yr^{-1} ha^{-1}). Some EOM applications unexpectedly caused less NL than synthetic fertilisation. NL minimisation implied the use of limited quantities of EOMs which were restrained to SSd, PL, and Fertylis (2%, 23% and 94% respectively) (Table 4). As other N losses (NH_3, N_2O, N_2) under the practice of EOM use were either lower or negligible compared to control, it might mean that N continuously released by these EOM decomposition was more efficiently used by plants than N applied in one, two or three times with synthetic fertilisation. The EOMs selected by the "EOM attribution" model were either very labile EOMs (PL) applied in late winter or in spring, or the most stable EOMs applied end of august. Fertylis was actually slighltly less stable than GWb however it was characterised by a higher C/N ratio that may result in N immobilisation likely to prevent from NL.

4.2.2. Impact of composting development (situation B)

Situation B resulted in lower quantities of EOM used: using HMc instead of HM or GWS instead of GW implied the use of lower quantities because of higher P and N contents in these new EOM and associated scenarios more constrained.

When maximising CS, storage relative to control was 0.37 t C yr^{-1} ha^{-1}, which was lower than in situation A by 0.10 t C yr^{-1} ha^{-1}. In this situation, 0% of the fertilising EOMs were applied, and among the amending EOMs, GWS only was not fully utilised (47%). Lower CS showed that increased stability of composts did not offset the lower potential amounts of EOM applicable.

When maximising NS, 29.4 kg N yr^{-1} ha^{-1} were saved, which represented only three-fifth of situation A. All EOMs were used except PLc while Fertylis and GWS were not totally used. The lower NS for situation B and the use of higher proportions of some EOMs such as Bio, Fertylis and Humival, already available in situation A, illustrated the lower capacity of the situation B EOMs set for NS. This might be due to EOM with overall higher stability and/or higher P content, and might be caused by some EOM composting (and mixing) when moving from situation A to situation B.

When minimising NL, NL (-0.35 kg N.yr^{-1}.ha^{-1}) were slightly lower than in situation A and GWS, Bio and Fertylis were used (28%, 68% and 30% respectively). These EOM were all very stable amending EOMs, releasing only little amounts of N which seemed to be used more efficiently by crops than synthetic fertilisers. Lower NL in sitation B might be allowed by the GWS not available in situation A.

4.2.3. Impacts on NH_3, N_2O and N_2

Surprisingly, for all objectives leading to EOM use instead of mineral N fertilisation, NH_3 volatilisation was reduced: from -1.1 to -3.4 kg N yr^{-1} ha^{-1}, compared to control emissions of 4.0 kg N yr^{-1} ha^{-1}. Such difference could be due to the chemical fertiliser type, which was composed of 25% of ammonium and 50% of urea, and therefore very subject to ammonia production and loss through volatilisation, while, the EOMs which partly substitute this synthetic fertiliser in the scenarios, were mostly composed of organic N more slowly transformed into ammonium, thus less subjected to

ammonia volatilisation. When ranking regional emissions among the different objectives and situations, increasing NH_3 emissions were perfectly related to decreasing NS, emphasising the hypothesis of synthetic fertiliser leading to more NH_3 emissions than EOM. The increase of composting in situation B led to slightly lower NH_3 emissions, and it might be due to lower ammonium contents of EOM used especially GWS compared to the initial non composted EOMs (sludge, PL and HM).

For both objectives corresponding to maximising CS and NS, NL was higher than for control. However, despite large quantities of EOM used, NL was only slightly higher. Indeed maximising CS and NS resulted in limited NL surplus ranging from 2.0 to 5.9 kg N yr^{-1} ha^{-1} which was quite low compared to 29.4 kg N yr^{-1} ha^{-1} calculated for control. This additional NL might be mostly due to the use of amending EOM end of summer before spring crops allowing losses before plant uptakes. However, the NS calculation limiting the overall amount of N applied might be the cause of the low NL surplus with EOM applications. Situation A led to higher NL than situation B whatever objective and this might be due to less stable amending EOMs applied. Except for the objective of NL minimisation, the more NS, the more NL at regional level. Indeed, both NS and NL were increased when the mineral N stocks increased especially during the plant growth period for NS and during late autumn for NL.

N_2O and N_2 emissions were higher when EOM were applied. Indeed, maximising CS led to surplus of N_2O emissions of 0.1273 kg N yr^{-1}, maximising NS to 0.0796 kg N yr^{-1} and minimising NL to 0.0144 kg N yr^{-1} compared to control, which represented surpluses of 11.7, 7.3, 0.7% respectively. For situation B, these N_2O emissions surpluses were much lower (5.2, 4.7, 0.8% respectively). As NS scores were greater for A, greater N_2O emissions for A could not be explained by higher quantities of synthetic fertiliser but rather by differences in EOM type applications. N_2O emissions were higher for A than for B (except for the NL minimisation objective) and then higher for CS than NS and NL. This could be explained by: (i) larger quantities of amending EOMs with low stability such as manures and GWa applied in late summer that could lead to higher N releases before winter when soils were filled with water, providing conditions leading to denitrification; (ii) higher overall EOM quantity applied (max CS>max NS>min NL) and therefore more N likely to be nitrified and denitrified. Similar results were obtained for N_2 emissions, with surplus of 11.6, 7.2 and 0.8% for situation A and 5.2, 4.7 and 0.8% for situation B compared to control for the same objectives respectively. The same reason can explained the ranking of regional N_2 emissions among objectives and situations.

4.3. EOM distributions

For this EOM distributions analysis, only situation A was studied, as EOM available formed a larger panel of EOMs with more different characteristics.

4.3.1. EOM distribution per soil types

Distribution of EOM according to soil types is presented Fig. 3. When maximising CS, soil 1 and 2 were opposed. All amending EOMs were applied on soil 1, while only fertilising EOMs were applied on soil 2 (Fig. 3a). Little amounts of the fertilising EOM, Humival, were applied on soils 3 and 4. Soil 3 was also amended with Bio and Fertylis, the most stable amending EOMs and soil 4 received GWa and HMs belonging to the least stable amending EOMs.

Such distribution may be due to the increasing potential of soil to store C, and especially from stable EOMs (Noirot-Cosson et al., 20XXb): soil 2 < soil 4 < soil 3 < soil 1. As a matter of fact, the "EOM attribution" model might favour the most stable EOMs on the soil having the highest potential of C storage, and preferentially distributed the least stable EOMs on the soil with the lowest potential.

Soil 4 received the largest quantities of EOM per unit area (5.2 Mg DM ha^{-1}), larger than soil 1 (4.6 Mg DM ha^{-1}), soil 3 (2.8 Mg DM ha^{-1}) and soil 2 (1.0 Mg DM ha^{-1}). This was related to the type of EOM it was amended with: the least stable amending EOM (HM and GWa) implied the largest potential rates of application.

When maximising NS, the most stable amending EOM used, GWb, was only applied on soil 2 and 4, while the least stable, HMs and HMw, on soil 1 and 3 (Fig. 3b). The least stable fertilising EOM used, PL, was also applied on soil 1 and 3, while the most stable fertilising EOM SSd was applied on every soil but soil 2.

This distribution was also consistent with the earlier study (Noirot-Cosson et al. 20XXb), in which potential of soil to save nitrogen because of indirect EOM effect related to accumulation of EOM resistant pool (expressed in kg $N.ha^{-1}/g\ C.kg^{-1}$) was sorted as follows: soil 2 > soil 4 > soil 3 > soil 1. EOMs that could bring large quantities of resistant OM i.e stable amending EOMs are prone to induce relatively higher NS if applied on soils 2 and 4 rather than on soil 3 and 1, the direct effect caused by mineral and labile N from EOM being independent from soil type.

For minimising NL, the largest quantities of EOM used were applied on soil 4 and 1, while soil 2 received only little amount of EOM and soil 3 none (Fig. 3c). This might be mostly explainable by the relative soil potential for extra NL because of EOM application which were sorted as follow: soil 4 < soil 1 < soil 3 < soil 2 (Noirot-Cosson et al., 20XXb).

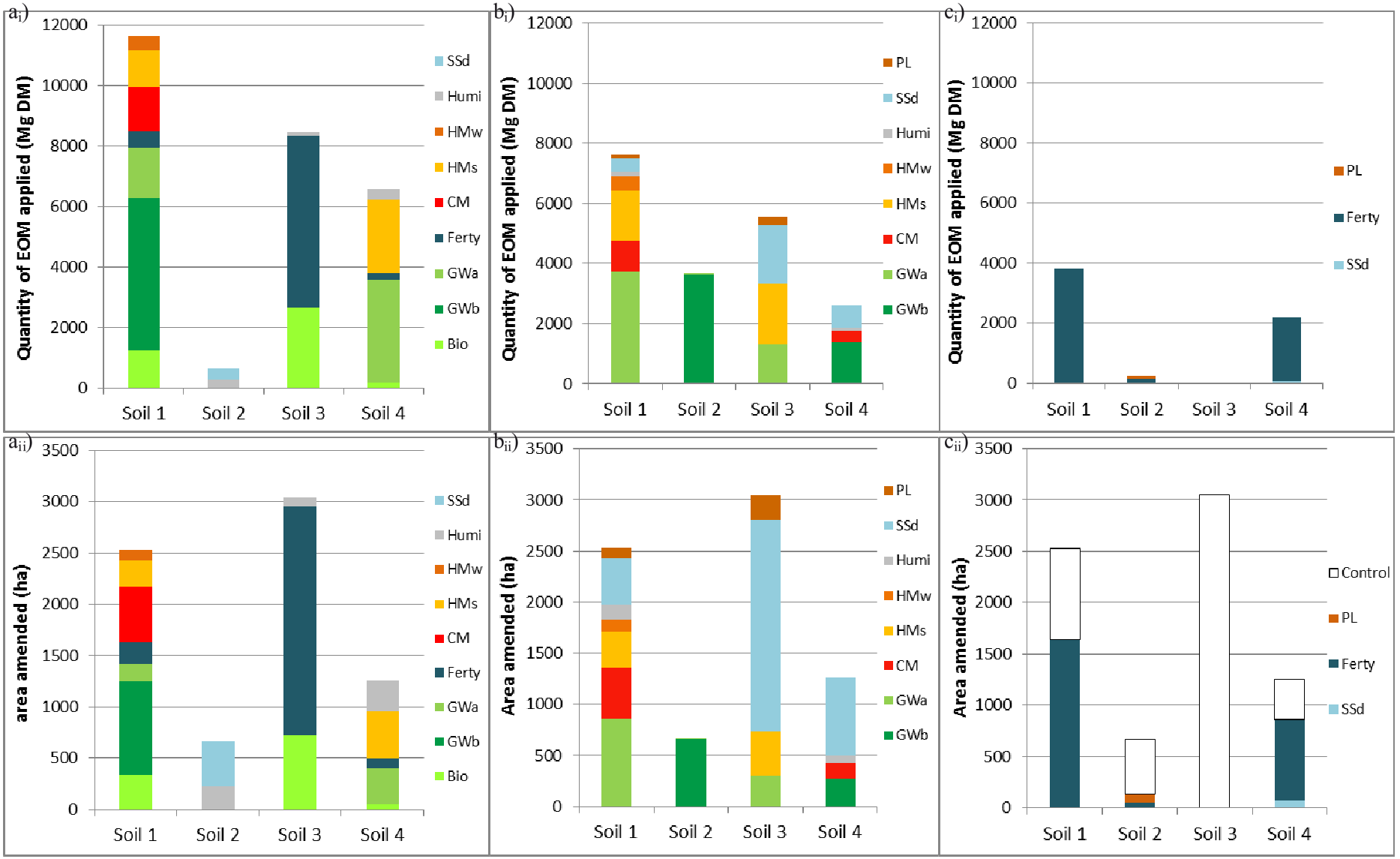


Figure 3. EOM applications per soil type when a) maximising CS, b) maximising NS, c) minimising NL, i) in terms of quantity applied, ii) in an terms area amended.

Soil 1 corresponds to the calcaric, rendzic cambisol, soil 2 to the areanic, podzolic cambisol, soil 3 to the haplic luvisol from loess deposits over limestone, soil 4 to the haplic luvisol (retic) from loess deposits over clay material. Among EOM, Bio stands for Biowaste compost, GWb for Green Waste compost "BYS", GWa for Green Waste compost "Sepur", Fertylis for the Green Waste and bone meal compost, CM for Cattle Manure, HMs for Horse Manure with Straw, HMw for Horse Manure with wooden shavings, Humival for the dried pig slurry, SSd for dried Sewage Sludge, PL for Poultry Litter.

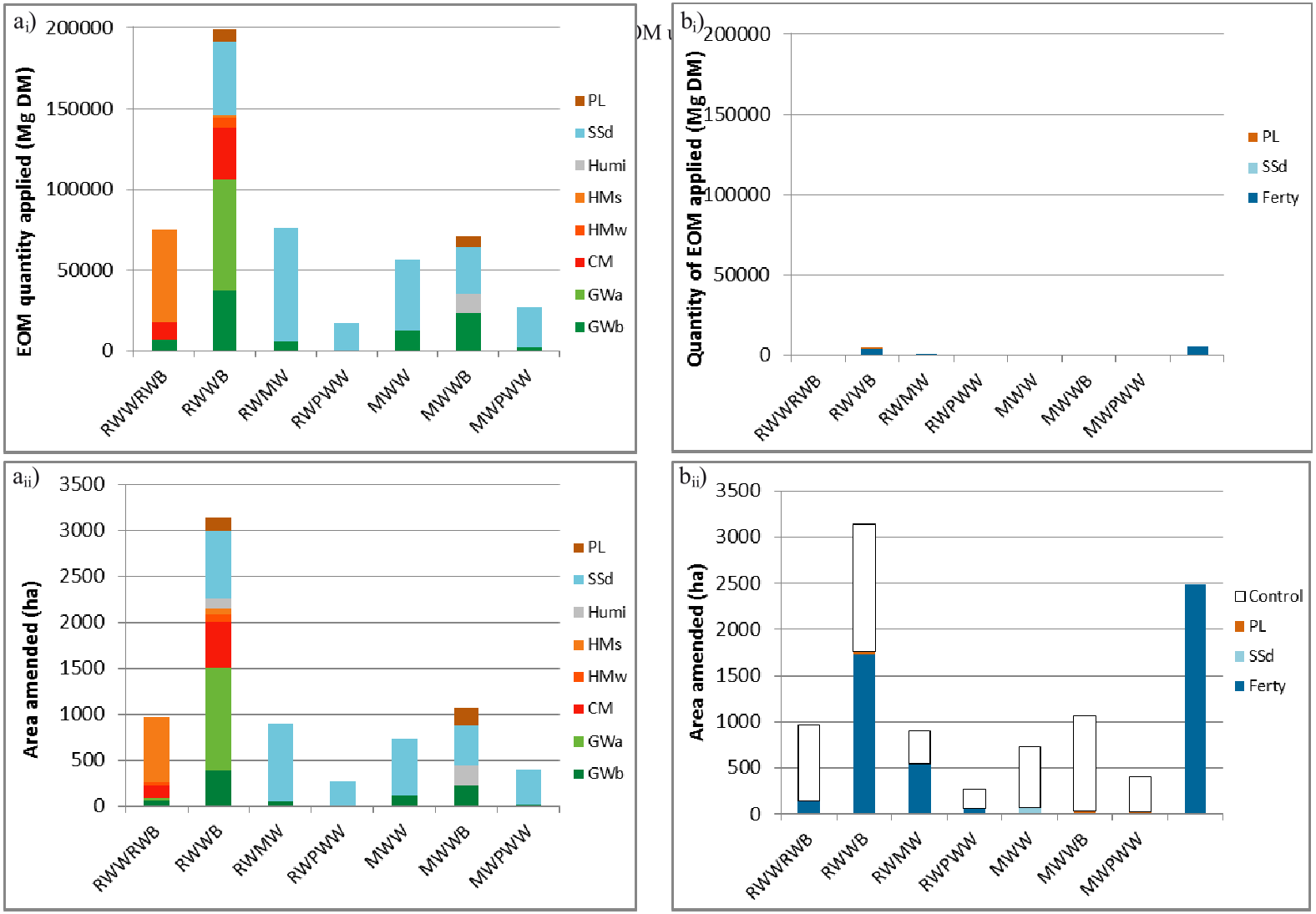


Figure 4. EOM application per crop succession area, a) when NS maximised, b) when NL minimised, i) in terms of EOM quantity applied, ii) in terms area amended.

RWWRWB, RWWB, RWMW, RWPWW, MWW, MWWB and MWPWW are the succession crops where R stands for winter oilseed rape, W for winter wheat, B for spring barley, M for maize, P for dry pea or fodder crops. Among EOM, PL stands for Poultry Litter, SSd for dried Sewage Sludge, Humival for the dried pig slurry, HMs for Horse Manure with Straw, HMw for Horse Manure with wooden shavings, CM for Cattle Manure, GWa for Green Waste compost "Sepur" , GWb for Green Waste compost "BYS", Fertylis for the Green Waste and bone meal compost.

4.3.2. EOM distribution per crop succession

For CS maximisation, crop succession effect on EOM distribution was not studied as no trend could be observed in accordance with our earlier study (Noirot-Cosson et al. 20XXb) where crop succession influence on CS was lower than either soil type or EOM types.

Crop succession influence on NS was higher. Distribution of EOM according to crop successions is presented in Fig. 4. When maximising NS (Fig. 4a), EOM applied on crop successions with maize were mostly fertilising EOM, especially SSd. On the contrary, amending EOM were only applied on crop successions with rapeseed (RWWB and RWWRWB) except GWb which was applied on 6 out of 7 crop successions. This was consistent with Noirot-Cosson et al. (20XXb), maize could slightly better used N from fertilising EOM than rapeseed but rapeseed could be largely more efficient than maize for using N from amending EOMs applied in September, even more from less stable amending EOMs such as manures. The GWb distribution might be due to its very stable composition which may lead to only little N supply in autumn and which may not give significant advantage to rapeseed compared to spring crops.

According to Noirot-Cosson et al. (20XXb), NL was rather influenced by soil characteristics as CS. However, it was also highly influenced by crop succession, probably because related to NS strongly driven by crop succession. For NL minimisation, no application was selected for most crop succession areas (Fig. 4b) however the little EOM quantities applied were mostly affected to crop successions with maize for SSd and PL, while, for Fertylis, the only affected crop successions included rapeseed. The selection of EOM applications for this purpose, meaning N supply more efficient than from the substituted synthetic N fertilisation, was in accordance with NS results, suggesting that crop succession with rapeseed better used N from amending EOMs while crop successions with maize better used N from fertilising EOMs.

4.3.3. EOM application concentration

By studying the EOM application frequencies in the scenarios selected by optimisation (Table 5), we could appreciate whether the EOM quantities applied were either: i) concentrated on the crop plots they amended, i.e. high frequencies of application selected or ii) widely spread, i.e. low frequencies selected.

Case (i) was observed under the objective of maximising CS for all amending EOMs while half of the fertilising EOM quantities was widely spread. High quantities of amending EOMs were therefore concentrated presumably on specific areas as on soil 1 and 3 to take advantage of their enhancing CS effect. The fertilising EOM were partly spread rather than being concentrated which would leave room for the remaining EOM The enhancement of OM degradation by concentrating already applied fertilising EOM would cause more C loss than C stored by applying the remaining EOM.

Table 5. Frequencies of EOM application within the crop succession for situation A, proportion of EOM used at each frequency.

			Max CS				Max NS				Min NL			
		Maximum Frequency	1	2	3	4	1	2	3	4	1	2	3	4
Fertilising	Humival	2	50%	50%	0%		100%	0%	0					
	SSd	2 (3 for RWWRWB and 1 for MWW)	57%	43%	0%		0%	100%	0		0%	100%	0%	
	PL	2 (1 for MWW)					0%	100%			0%	100%		
Amending	HMs	2 (3 for RWWRWB)	0%	100%	0%		3%	97%	0					
	HMw	2 (1 for MWW)	0%	100%			100%	0%						
	GWb	3 (4 for RWWRWB and 2 for MWW)	0%	0%	63%	37%	0%	63%	30%	7%				
	GWa	3 (4 for RWWRWB and 2 for MWW)	0%	63%	37%	0%	100%	0%	0	0				
	CM	2 (1 for MWW)	0%	100%			79%	21%						
	Bio	2 (3 for RWWRWB)	0%	57%	43%									
	Fertylis	1	100%								100%			

Max and Min for maximisation and minimisation procedures. CS, NS, NL for Carbon Storage, Nitrogen Savings and Nitrate Leaching respectively. RWWRWB, RWWB, RWMW, RWPWW, MWW, MWWB and MWPWW are the succession crops where R stands for winter oilseed rape, W for winter wheat, B for spring barley, M for maize, P for dry pea or fodder crops. Among EOM Humival stands for the dried pig slurry, SSd for dried Sewage Sludge, PL stands for Poultry Litter, SSl for limed Sewage Sludge, PLc for Poultry Litter composted, HMs for Horse Manure with Straw, HMw for Horse Manure with wooden shavings, Fertylis for the Green Waste and bone meal compost, GWb for Green Waste compost "BYS", GWa for Green Waste compost "Sepur", CM for Cattle Manure, HMc for Horse Manure composted, GWS for co-compost of green waste and sewage sludge, Bio for Biowaste compost and Fertylis a green waste compost mixed with bone meal.

Conversely, when maximising NS, fertilising EOM SSd and PL were concentrated (case i), while amending EOMs and Humival were more widely spread tending towards case (ii), especially GWa, CM, HMw and Humival. Fertilising EOM, which were applied rather on maize, were concentrated and left room for other EOM application that could allow supplementary NS despite potentially lower NS efficiency. However, the selected amending EOM were not concentrated meaning surprisingly that concentrating these EOM on some plots and using the remaining PL, SSd, Bio and Fertylis on the new available surfaces would not allow higher NS. It might be due to the fact that concentrating the selected amending EOMs would mean applications on barley and wheat within crop successions, two crops not using the direct effect of nitrogen availability of amending EOM.

4.3.4. Spatial patterns of EOM distribution

Maps shown in Fig. 5 illustrates the EOM distribution analysed above, and especially differences in spatial distribution when pursuing different objectives, here, maximising CS or NS. Mainly because of their large quantities available, the EOM applications covering large areas were GWb, GWa, CM for both objectives, Bio and Fertylis for maximising CS, SSd for maximising NS. In terms of EOM distribution, the two objectives seemed to be opposed. Many plots amended with stable EOM Fertylis and Bio for maximising CS were amended by fertilising SSd or PL when maximising NS, while plots amended with SSd for CS were amended with the stable GWb for NS. Similarly, plots amended with the quite stable CM for CS were amended with the slighlty less stable GWa for NS, and on the contrary plots where CM were applied for NS, were amended with a large panel of EOMs for CS. These opposite distributions were explained by the strong inluence of soil type on CS and NS results (Noirot-Cosson et al., 20XXb) and the opposed effects of soil OM protection and soil capacity to mineralise EOM.

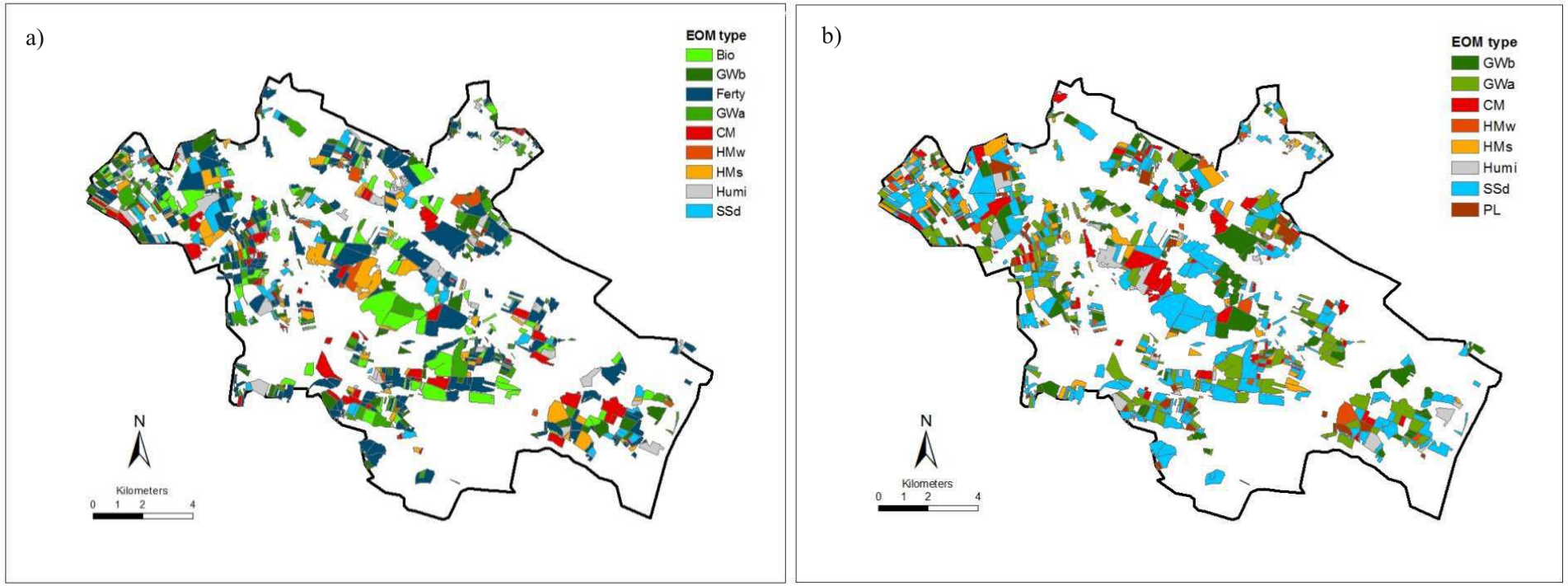


Figure 5. EOM distribution (situation A) under objectives of: a) maximising CS, b) maximising NS.
Bio stands for Biowaste compost, GWb for Green Waste compost "BYS", GWa for Green Waste compost "Sepur", CM for Cattle Manure, HMw for Horse Manure with wooden shavings, HMs for Horse Manure with Straw, Humi for the dried pig slurry, SSd for dried Sewage Sludge, PL for Poultry Litter "Humival".

5. Discussion

5.1. Regional strategies for EOM management

In this study we focused on allocation of the available EOM to crop plots of the region following various objectives. Makowski et al. (2000) pointed out the risk of high sensitivity of decision variables, i.e. scenarios selection for the crop plots in our study, despite robustness of the optimal value of the objective function, maximum regional CS for instance. For this reason, maps of optimal EOM distributions (Fig. 5) were given more as illustrations, and we mainly focused on drivers of these distribution patterns and associated regional results. The model-based study resulted in different recommendations of EOM distributions for maximising CS or NS or minimising NL at the regional scale. Indeed, the different objectives led to strong differences in the EOM types used, applied quantities, soil types and amended crop successions.

While maximising CS and NS led to use of total available EOM, minimising NL led to use very little EOMs. Maximising CS and NS led to use of different EOM types and to opposite distribution pattern with use of fertilising EOM especially SSd for maximising NS and larger quantities of stable amending EOM as Bio and Fertylis for maximising CS. However for both objectives, the model attributed 100% of available manures and green waste compost quantities for situation A, i.e. EOMs that had intermediate stability and/or could be applied in large quantities. The optimisation seemed to fully benefit from soil potentials either for protecting stable OM or for mineralising EOM or for preventing NL. For maximising CS, soil 1 and 3 with the highest clay and calcareous contents i.e. the highest OM protecting effect were dedicated to amending EOMs especially the most stable. This was nearly the contrary for maximising NS. Soil 2 and 4 with the highest capacity of mineralising the large proportion of resistant OM in EOM were dedicated to receive the most stable EOM among the selected EOMs. For minimising NL, EOM applications were avoided on the leaching arenic, podzolic cambisol, soil 2.

The optimisation seemed also to fully benefit from the highest NS efficiency of fertilising EOM on maize and of amending EOMs on rapeseed, in order to both maximise NS and minimise NL, while CS did not show any trend in EOM dedication on specific crop successions. This similarity in terms of pattern drivers for NS and NL suggested that NL could be decreased with EOM application only if it was associated to relatively important NS leading to more efficient use of N than in the control under conventional synthetic N fertiliser. Eventually, CS and NS could drastically increase with EOM amendments, and especially when optimally distributed, while associated NL increases could be quite reasonable by adapting synthetic fertilisation.

We also explored impacts on regional CS, NS or NL of composting the available manures, poultry litter and sludge (situation B). Composting EOMs, that could be interesting for decreasing EOM transportation costs, EOM sanitation, or stabilising carbon while decreasing readily available N, only resulted in slightly lower N outputs but also in consequently lower CS and NS. However, this was

mostly due to co-composting sludge with green wastes which would lead to a product with more P and more limited in its application than GW composts, rather than the composting effect by itself. Indeed, composting HM may eventually lead to higher CS.

5.2. Regional potential benefits and risks

The maximum CS that could be obtained was 0.47 t C yr^{-1} ha^{-1} on average for the region which represented surplus of about 200% compared to CS obtained without EOMs. Cropping systems defined here included agricultural practices already leading to consequent CS even without amendment, and this could have limited the benefits of EOM application. This maximum of CS was obtained by applying 3.6 t DM ha^{-1} and 7.8 t FM ha^{-1} on average. This was higher than the CS value of 0.29 t C.yr^{-1}.ha^{-1} found by Mondini et al. (2012) who explored the CS potential for composts application of 10 t FM yr^{-1} ha^{-1} on agricultural soils at the scale of whole Italy. It was also higher than the value of 0.32 t C yr^{-1} ha^{-1} found by Yokozawa et al. (2010) for 15 t FM ha^{-1} application of manures on Japanese uplands, and the value of 0.22 t C ha^{-1} yr^{-1} found by Koga et al. (2011) under application of 20 t FM yr^{-1} ha^{-1} of composted cattle manure on northern Japanese arable soils. Despite lower EOM quantities applied in our study, such higher CS was expectable, as the objective pursued was to maximise CS, which resulted in concentrating stable EOMs on soils with high storage potential of C from EOM. Moreover, Mondini et al. (2012) took into account the global warming and a temperature increase of 3.2 °C within the 100 years of simulation, while Yokozawa et al. (2010) considered soils with high initial SOC contents, an average of 89.5 t C ha^{-1} for uplands, two factors which would also limit CS potential. These results may emphasise the potential of optimising EOM distribution for drastically increasing CS.

In France, “4 per 1000” program has been launched to enhance C storage in soils. An annual increase of soil C by 4 ‰ would offset annual surplus of Greenhouse Gases (GHG) emissions. This would correspond to an increase of 0.16 t C ha^{-1} yr^{-1} for soils of the region containing on average 39 t C ha^{-1}. Using regional EOMs and distributed in an optimised manner, leading to 0 47 t C yr^{-1} ha^{-1}, would be an efficient way in order to reach and exceed the “4 per 1000” objective. It would increase soil C by 9.4 t C ha^{-1} on average within 20 years, likely to consequently enhance soil chemical, physical and biological fertility. This CS also corresponded to 258 000 tons of eq-CO_2 not emitted during these 20 years at the regional scale. Considering the associated NS (4 380 t N in 20 years) and CO_2 emissions for Urea-Ammonium Nitrate (as used on the region) of 3.13 kg CO_2-eq kg^{-1} N (Nemecek and Kägi, 2007), a total of 13 700 t eq-CO_2 would not be emitted thanks to NS due to EOM applications, ending up to a total of 271 700 t eq-CO_2.

Contrary to CS, for NS and N emissions, some authors explicitly spatialised N outputs but rarely studying EOM application effects. In our study, the average standard mineral fertilisation was 144 kg N ha^{-1} yr^{-1}. Maximising NS resulted in saving 40% of the total synthetic N fertilisation, 37% explained

by EOM applications and 3% by differences of TOCini and residues accumulation. In Noirot-Cosson et al. 20XXb, the non-optimised regional use of EOM for the same region led to 29% of mineral N savings, illustrating the optimisation interest. 25 030 t eq-CO_2 would not be emitted thanks to NS due to EOM applications when maximised. The associated CS represented 34 490 t C in 20 years or 126 500 t eq-CO_2, ending up to a total of 151 530 t eq-CO_2, 1,8 times lower than with CS maximised.

Dragosits et al. (1998) simulated ammonia emissions for UK and used a simple factor for croplands: 2.94% of the N fertilisation was lost through NH_3 volatilisation. Using the same emission factor, in our study without EOM application, NH_3 volatilisation would be 4.1 kg N ha^{-1} yr^{-1}, which was close to our control simulations corresponding to mineral N fertilisation (4.0 kg N ha^{-1} yr^{-1}). Gabrielle et al., (2014) simulated N losses through NO_3 leaching, NH_3 and direct N_2O for winter wheat and rapeseed of the whole Ile de France region: they found losses of 36.8 and 9.4 kg N-NO_3 ha^{-1} yr^{-1}, 4.10 and 2.31 kg N-NH_3 ha^{-1} yr^{-1}, 1.04 and 0.42 kg N-N_2O ha^{-1} yr^{-1} for wheat and rapeseed, respectively which fell in the range of our 29.7 kg N-NO_3 ha^{-1} yr^{-1}, 4.03 kg N-NH_3 ha^{-1} yr^{-1}, 1.09 kg N-N_2O ha^{-1} yr^{-1}. Duretz et al., (2011) studied N fluxes at the landscape scale (270 ha of croplands) and found 106.3 kg N-NO_3 ha^{-1} yr^{-1} of NO_3 leaching, 0.37 kg N-NH_3 ha^{-1} yr^{-1} of net NH_3 emissions, 4.07 kg N-N_2O ha^{-1} yr^{-1}, with mineral fertilisation of 240 kg N ha^{-1} for wheat and 120 kg N ha^{-1} for maize, the latter also receiving 120 kg N ha^{-1} of organic amendment. Taking into account a fertilisation of almost the double of our mean control fertilisation, N leaching and N_2O emissions were in the same order of magnitude as our results. The relatively low surplus of N_2O emissions with EOM applications is in accordance with Houot et al., (2014) who found only small differences of emissions between EOM and synthetic fertilisation. EOM applications can lead to large NH_3 emissions ranging from 0 to 100% of the N-$NH4^+$ from EOM depending on the EOM type and the context of application (Génermont, 2014). Here scenarios with EOM applications led to less NH_3 emissions than for control. It might be due low mineral N content in our EOMs compared to organic N which allowed large mineral N substitutions.

However our N emissions results should be taken cautiously. Indeed, despite CERES-EGC ability for NH_3, N_2O and N_2 emission simulations (Gabrielle et al., 2005, 2006), it was not dedicated for this purpose especially in case of EOM applications and should be validated against measurements.

5.3. Method reproducibility

Such regional study was intended to support policy makers by assessing potential of optimised EOM distribution management. GHG emissions limitations and offsetting by carbon sequestration are global issues that need to be thought at large scale. Following such objectives would incite to extend optimised EOM distribution to larger scale and therefore to apply such study to larger areas.

This study was possible thanks to initial precise descriptions of soil types distribution collected for the Plain of Versailles (Crahet, 1992) and soil types properties (Hamiache et al., 2012; Laville et al., 2014) and with some parameterisation (Chalhoub et al., 2013; Gabrielle et al., 2002), of crop successions and

cropping systems including EOM application practices (Dhaouadi, 2014), of quantities of available EOM in the region along with EOM laboratory characterisation (Dhaouadi, 2014). Then, this study was strengthened by the availability of a long-term field experiment on EOM application in the region (QualiAgro). Using such data was necessary for calibrating the model based on the simulations of long-lasting sets of data issued from long-term experiments. Efforts were made in this work to improve the method repoducibility. In Noirot-Cosson et al. (20XXa), a lot of EOM were already parameterised, through a parameterisation that might be applicable whatever the region studied; moreover, methods were defined to parameterise other EOM only based on few characterisitcs. In Noirot-Cosson et al. (20XXb), factors were sorted according to their influence on the outputs studied. This would help to prioritise efforts that should be made for intial region description, or for model parameterisation…

5.4. Multi-criteria analysis

For policy-making, the practice needs to be evaluated taking into account all possible impacts. Then they would need to assess relative importance of these different impacts, or even choosing among the impacts to be considered, and eventually trade-offs could be studied to help making decisions.

In this study, EOM applications impacts were studied based on several outputs of C and N, and also taking into account a constraint on P, which also led to respect regulations on trace-element inputs overcoming these other risks than N outputs (Noirot-Cosson et al., 20XXb). Even in terms of N outputs we were not exhaustive because of lack of measurements to correctly calibrate the model for all these N outputs, especially for gas emissions. However, the studied indicators appeared to be of great importance: CS because SOC are related to many other ecosystemic services; NS since it has a direct economic interest and farmers would be very sensitive to such parameter; NL was also important, especially in vulnerable areas as defined by the 91/676CEE “nitrate directive” such as the plain of Versailles. However trade-offs were not tested here.

5.5. Applicability

Besides its benefits for policy makers as it intends to deal with global issues, EOM use directly concerns farmers and it would be also constrained by farm economic issues. EOM use could mean money saving along with synthetic fertiliser savings, but also additional costs through EOM formulation, trading, transportation, or the purchase of application equipments. However many EOMs were by-products of animal origin or issued from waste treatments which cost are supported mainly by the waste producers and not the EOM user. EOM use would also mean additional time spent for amending when only part of the mineral fertilisation was substituted. Practice of EOM use is also constrained by social acceptability for some EOMs such as sewage sludges, municipal solid waste

composts or slurries for health fears or smell inconvenience. A lot of reasons set this study as a preliminary study for EOM distribution management.

6. Conclusion

This study aimed to assess the regional potential of optimising EOM distribution in terms of environmental impacts such as carbon storage in soil, savings of synthetic nitrogen fertiliser, nitrogen leaching and gaseous N emissions. For this purpose, it considered available EOMs over a region but it also tested the situation for which all EOMs were composted or matured. Our results showed that optimisation model could lead to mitigate negative impacts meanwhile fostering beneficial impacts of EOM amendments at the regional scale. Depending on the optimisation objectives or the EOMs available for use, EOM distribution as well as potential environmental impacts drastically differed. While maximising CS led to storage of 0.47 t C yr^{-1} ha^{-1} and synthetic fertiliser savings of 29.2 kg N yr^{-1} ha^{-1}, maximising NS led to 0.23 t C yr^{-1} ha^{-1} stored and 53.3 kg N yr^{-1} ha^{-1} compared to control without EOM application over the 7498 ha of the region. Following these two objectives, the optimisation model resulted in amending all crop plots. For CS, more amending EOMs were used, and they were rather concentrated on soils with high potential of CS. For NS, the use of higher fertilising EOM quantities were favoured, and rather concentrated on crop successions with maize while amending EOMs were spread on crop succession with rapeseed. These objectives led to opposite maps of EOM distribution. For NL minimisation few EOM application were selected, and probably those which implied more efficient use of N by plant than with synthetic fertilisation, and application of one amending EOM on rapeseed rotation and two fertilising EOM on maize rotation were favoured. When EOM were composted or co-composted, optimal distributions led to lower regional CS and NS because of lower quantities applicable, the composted EOMs containing more N and P (especilly the co-compost of green waste and sewage sludge) and being thus more constrained. Consequently, it also resulted in lower NL and emissions of NH_3, N_2O and N_2.

Such study set basis of regional management of EOMs to improve environmental benefits related to this practice. To consider such regional management, other factors would need to be taken into account, such as other environmental factors but also logistical constraints and socio-economic factors and especially at the farm scale. To help policy makers, impacts also need to be compared. However, this study offered promising results in terms of environmental benefits of such approach.

7. Supplementary data.

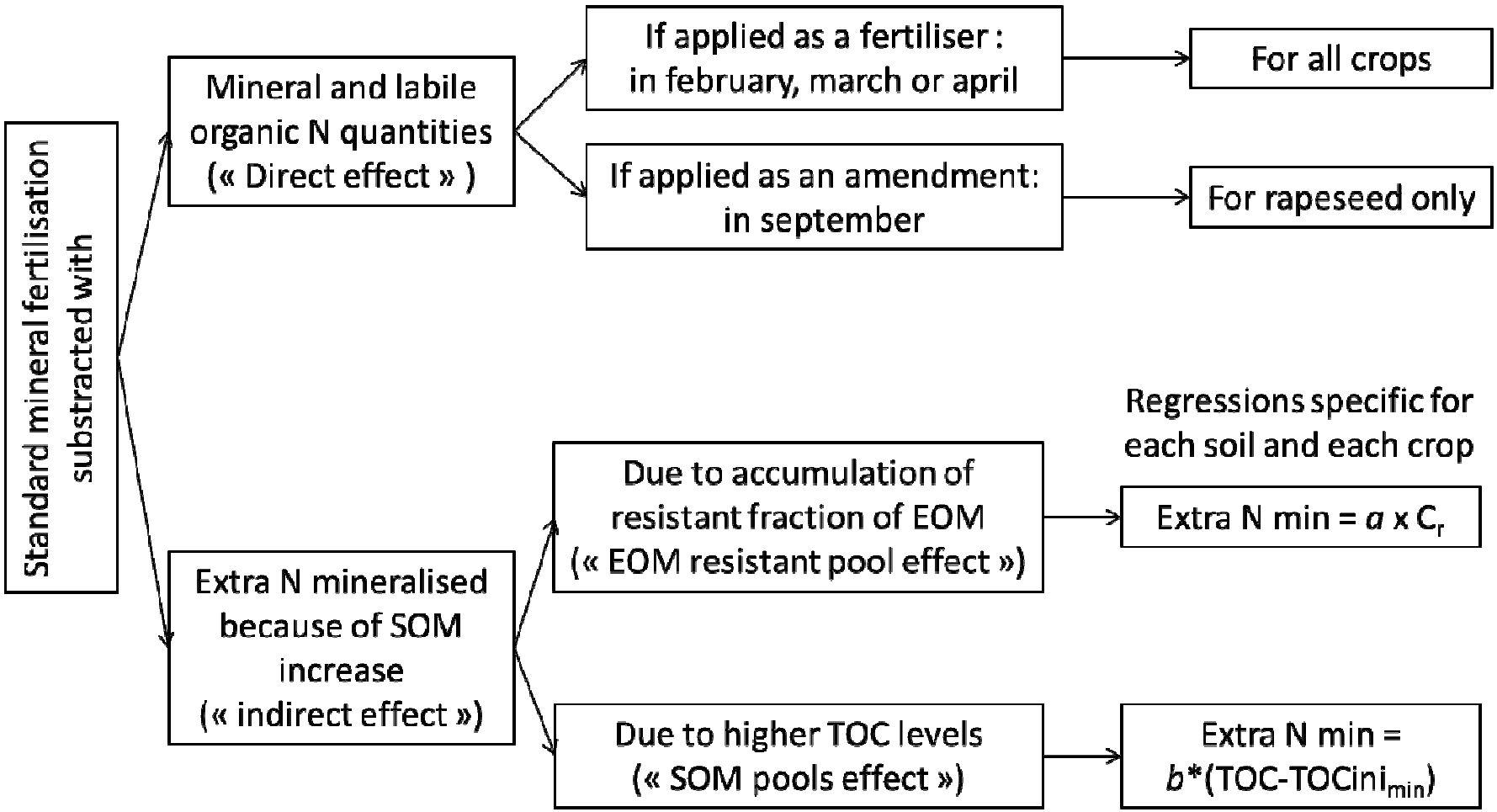


Figure 1S. (Noirot-Cosson et al. 20XXb.): Scheme of the rules for nitrogen saving calculation.
"Extra N min" stands for surplus of mineral N stocks (kg N ha^{-1}) in soil when EOM is applied or when the organic carbon content in soil pools (TOC, g C kg^{-1}) is higher than the lowest initial SOC content ($TOCini_{min}$, g $C.kg^{-1}$) specific of a soil type, Cr (g $C.kg^{-1}$) stands for EOM resistant pool size, *a* and *b* are regression coefficients (kg N ha^{-1}/ g C kg^{-1}).

Table 1S: EOM amendment scenarios

		Humival	SSd	PL	SSl	PLc	HMs	HMw	GWb	Gwa	CM	HMc	GWS	BIO	Fertylis
Crop succession	**P needs (kg P2O5 ha-1)**	**N from EOM (kg N ha^{-1}): Maximum application within the crop succession when contrained on P**													
succ1-RWWB	202	222	281	184	106	102	334	267	551	531	312	224	170	335	153
succ2-MWWB	203	223	282	185	107	102	346	268	553	533	313	225	171	336	153
succ3-RWMW	224	246	310	203	118	112	380	295	609	587	345	247	188	370	169
succ4-RWWRWB	314	345	435	286	165	158	534	413	854	823	484	347	263	520	237
succ5-MWW	161	177	223	146	85	81	274	212	438	422	248	178	135	267	121
succ6-MWPWW	260	297	360	236	136	130	441	342	706	680	400	287	278	430	196
succ7-RWPWW	256	287	354	232	134	128	434	336	695	670	394	282	214	423	193
N from EOM for each application (kg N ha-1)		**Scenarios of EOM applications**													
Constraint on N (kg N.ha^{-1})		**<170**	**<250**	**<170**	**<250**	**<170**	**<250**	**<170[1]**	**<170[1]**	**<250**	**<250**	**<250**	**<250**	**<250**	**<250**	**<250**	**<250**
Nb of applications		**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**
succ1-R*WWB		170	222	170	250		184	106	102	250	250	250	250	250	224	170	250	153
succ2-M*WWB		170	223	170	250		185	107	102	250	250	250	250	250	225	171	250	153
succ3-R*WMW		170	246	170	250	170	203	118	112	250	250	250	250	250	247	188	250	169
succ4-R*WWRWB		170	250	170	250	170	250	165	158	250	250	250	250	250	250	250	250	237
succ5-M*WW		170	177	170	223	146		85	81	250	212	250	250	248	178	135	250	121
succ6-M*WPWW		170	250	170	250	170	236	136	130	250	250	250	250	250	250	218	250	196
succ7-R*WPWW		170	250	170	250	170	232	134	128	250	250	250	250	250	250	214	250	193
Nb of applications		**2**		**2**	**2**	**2**				**2**	**2**	**2**	**2**	**2**	**2**	**2**	**2**	**2**
succ1-R*WWB*		111		140		92				172	133	250	250	156			168	
succ2-M*WWB*		112		141		92				173	134	250	250	157			168	
succ3-R*WM*W		123		155		102				190	147	250	250	172			185	
succ4-R*WWR*WB			172	170	218	143				250	207	250	250	242	174	132	250	
succ5-M*WW*				112						137		219	211				133	
succ6-M*WPWW*		149			180	118				221	171	250	250	200	143		215	
succ7-R*WPWW*		144			177	116				217	168	250	250	197	141		211	
Nb of applications		**3**		**3**						**3**		**3**	**3**				**3**	
succ1-R*WW*B*												184	177					
succ2-M*WW*B*												184	178					
succ3-R*W*M*W												203	195					
succ4-R*WWR*WB*		115		145						178		250	250				173	
succ5-M*WW*																		
succ6-M*WPW*W*												235	227					
succ7-R*W*PWW*												232	223					
Nb of applications												**4**	**4**					
succ4-R*WW*R*WB*												214	206					

[1]First constraint on N not binding, second not tested. C* stands for crop amended**.** RWWRWB, RWWB, RWMW, RWPWW, MWW, MWWB and MWPWW are the crop successions where R stands for rapeseed, W for winter wheat, B for spring barley, M for maize, P for dry pea or fodder crops. Among EOM Humival stands for the dried pig slurry, SSd for dried Sewage Sludge, PL stands for Poultry Litter, SSl for limed Sewage Sludge, PLc for Poultry Litter composted, HMs for Horse Manure with straw, HMw for Horse Manure with wooden shavings, GWb for Green Waste compost "BYS", GWa for Green Waste compost "Sepur", CM for Cattle Manure, HMc for Horse Manure composted, GWS for co-compost of green waste and sewage sludge, BIO for Biowaste compost, Fertylis for the Green Waste and bone meal compost.

Table 2S. EOM characteristics. (Noirot-Cosson et al., 20XXa)

	EOM type	Mineral N content (mg N kg^{-1} DM)	OC content (g C kg^{-1} DM)	Organic C/N ratio	P_2O_5 (g P kg^{-1} DM)	C_r (%)	k_l (d^{-1})	k_r (d^{-1})
Fertilising	Dried Pig Slurry ("Humival")	696	407	7.8	47.2	40	0.376	0.00159
	Dried Sewage Sludge (SSd)	2897	397	5.6	51.6	17	0.376	0.00209
	Limed Sewage Sludge (SSl)	171	263	5.9	84.8	29	0.376	0.00209
	Dried Poultry Litter (PL)	2108	344	8.8	43.6	0	0.376	0.00209
	Composted Poultry Litter (PLc)	661	239	9.5	55.0	13	0.114	0.00159
Amending	Biowastes Compost (BIO)	1247	208	8.6	11.0	92	0.114	0.000971
	Green Wastes compost "Sepur" (GWa)	23	334	23.1	5.2	45	0.114	0.000971
	Green Waste compost "BYS" (GWb)	231	268	10.6	6.5	87	0.114	0.000971
	Green Waste and Sewage Sludge compost 2009 (GWS)	4018	266	12.1	29.7	87	0.114	0.000971
	Green waste and bone meal ("Fertylis")	1438	257	16.7	19.7	79	0.114	0.000971
	Horse manure composted (HMc)	624	194	12.8	12.8	86	0.114	0.000971
	Horse manure with straw (HMs)	110	406	22.8	9.3	14	0.114	0.00159
	Horse manure with wood shavings (HMw)	71	428	29.3	10.2	21	0.114	0.00159
	Cattle manure (CM)	508	400	13.6	19.2	54	0.114	0.000971

DM stands for dry matter, C_r for the proportion of resistant carbon of the EOM, k_r and k_l for the decomposition rates of the EOM resistant and labile carbon respectively, C_r, k_r and k_l are models parameters defined in Noirot-Cosson et al. 20XXa).

8. References

Alvarez, R., 2005. A review of nitrogen fertilizer and conservation tillage effects on soil organic carbon storage. Soil Use Manag. 21, 38–52.

Askri, A., 2015. Valorisation des digestats de méthanisation : effets sur les cycles biogéochimiques du carbone et de l'azote. PhD thesis. AgroParisTech, Paris, France.

Barak, P., Molina, J.A.E., Hadas, A., Clapp, C.E., 1990. Optimization of an ecological model with the Marquardt algorithm. Ecol. Model. 51, 251–263.

Barrault, B., 2012. Atlas des installations franciliennes de traitement 2010 (Rapport de synthèse). Observatoire Régional des Déchets de l'Ile de France, Paris.

Bastida, F., Kandeler, E., Moreno, J.L., Ros, M., García, C., Hernández, T., 2008. Application of fresh and composted organic wastes modifies structure, size and activity of soil microbial community under semiarid climate. Appl. Soil Ecol. 40, 318–329.

Biomasse Normandie, Trivalor, 2000. Le compostage en Ile de France, Etat des lieux. Perspectives de développement de la filière. Débouchés. Observatoire Régional des Déchets de l'Ile de France, Paris.

Bruun, S., Christensen, B.T., Hansen, E.M., Magid, J., Jensen, L.S., 2003. Calibration and validation of the soil organic matter dynamics of the Daisy model with data from the Askov long-term experiments. Soil Biol. Biochem. 35, 67–76.

Castelán-Ortega, O.A., Fawcett, R.H., Arriaga-Jordán, C., Herrero, M., 2003. A Decision Support System for smallholder campesino maize–cattle production systems of the Toluca Valley in Central Mexico. Part I—Integrating biological and socio-economic models into a holistic system. Agric. Syst. 75, 1–21.

Chalhoub, M., Garnier, P., Coquet, Y., Mary, B., Lafolie, F., Houot, S., 2013. Increased nitrogen availability in soil after repeated compost applications: Use of the PASTIS model to separate short and long-term effects. Soil Biol. Biochem. 65, 144–157.

Ciais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., Chhabra, A., DeFries, R., Galloway, J., Heimann, M., Jones, C., Le Quéré, C., Myneni, R.B., Piao, S., Thornton, P., 2013. Carbon and Other Biogeochemical Cycles. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Ciais, P., Wattenbach, M., Vuichard, N., Smith, P., Piao, S.L., Don, A., Luyssaert, S., Janssens, I.A., Bondeau, A., Dechow, R., Leip, A., Smith, P., Beer, C., Van Der Werf, G.R., Gervois, S., Van Oost, K., Tomelleri, E., Freibauer, A., Schulze, E.D., Carboeurope Synthesis Team, 2010. The European carbon balance. Part 2: croplands. Glob. Change Biol. 16, 1409–1428.

Crahet, M., 1992. Soil Map of the Versailles Plain. Scale 1:50 000 (Internal report). Institut National Agronomique Paris-Grignon.

Davies, G., Lennartsson, M., 2005. Organic vegetable production. Crowood Press Ldt.

de Wit, C.T., van Keulen, H., Seligman, N.G., Spharim, I., 1988. Application of interactive multiple goal programming techniques for analysis and planning of regional agricultural development. Agric. Syst. 26, 211–230.

Dhaouadi, A.K., 2014. Insertion des Produits Résiduaires Organiques dans les systèmes de culture : Cas des systèmes céréaliers de la Plaine de Versailles et du Plateau des Alluets. Paris, AgroParisTech.

Diacono, M., Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility. A review. Agron. Sustain. Dev. 30, 401–422.

Dogliotti, S., van Ittersum, M.K., Rossing, W.A.H., 2005. A method for exploring sustainable development options at farm scale: a case study for vegetable farms in South Uruguay. Agric. Syst. 86, 29–51.

Doublet, J., Francou, C., Poitrenaud, M., Houot, S., 2011. Influence of bulking agents on organic matter evolution during sewage sludge composting; consequences on compost organic matter stability and N availability. Bioresour. Technol. 102, 1298–1307.

Dragosits, U., Sutton, M.A., Place, C.J., Bayley, A.A., 1998. Modelling the spatial distribution of agricultural ammonia emissions in the UK. Environ. Pollut., Nitrogen, the Confer-N-s First International Nitrogen Conference 1998 102, 195–203.

Duretz, S., Drouet, J.L., Durand, P., Hutchings, N.J., Theobald, M.R., Salmon-Monviola, J., Dragosits, U., Maury, O., Sutton, M.A., Cellier, P., 2011. NitroScape: A model to integrate nitrogen transfers and transformations in rural landscapes. Environ. Pollut., Assessment of Nitrogen Fluxes to Air and Water from Site Scale to Continental Scale 159, 3162–3170.

Edmeades, D.C., 2003. The long-term effects of manures and fertilisers on soil productivity and quality: a review. Nutr. Cycl. Agroecosystems 66, 165–180.

European Commission, 2006. Proposal for a Directive of the European Parliament and of the Council establishing a framework for the protection of soil and amending Directive 2004/35/EC.

European Commission, 2010. Communication from the commission to the council and the European parliament.

Falloon, P., Smith, P., Szabó, J., Pásztor, L., 2002. Comparison of approaches for estimating carbon sequestration at the regional scale. Soil Use Manag. 18, 164–174.

Francou, C., 2003. Stabilisation de la matière organique au cours du compostage de déchets urbains : influence de la nature des déchets et du procédé de compostage - recherche d'indicateurs pertinents. PhD thesis. INAPG (AgroParisTech), Paris, France.

Francou, C., Linères, M., Derenne, S., Villio-Poitrenaud, M.L., Houot, S., 2008. Influence of green waste, biowaste and paper–cardboard initial ratios on organic matter transformations during composting. Bioresour. Technol. 99, 8926–8934.

Franko, U., Crocker, G.J., Grace, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter, D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using the CANDY model. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 109–120.

Gabrielle, B., Da-Silveira, J., Houot, S., Francou, C., 2004. Simulating Urban Waste Compost Effects on Carbon and Nitrogen Dynamics Using a Biochemical Index. J. Environ. Qual. 33, 2333.

Gabrielle, B., Da-Silveira, J., Houot, S., Michelin, J., 2005. Field-scale modelling of carbon and nitrogen dynamics in soils amended with urban waste composts. Agric. Ecosyst. Environ. 110, 289–299.

Gabrielle, B., Gagnaire, N., Massad, R.S., Dufossé, K., Bessou, C., 2014. Environmental assessment of biofuel pathways in Ile de France based on ecosystem modeling. Bioresour. Technol. 152, 511–518.

Gabrielle, B., Laville, P., Duval, O., Nicoullaud, B., Germon, J.C., Henault, C., 2006. Process-based modeling of nitrous oxide emissions from wheat-cropped soils at the subregional scale. Glob. Biogeochem. Cycles 20, 1–13.

Gabrielle, B., Menasseri, S., Houot, S., 1995. Analysis and Field Evaluation of the Ceres Models Water Balance Component. Soil Sci. Soc. Am. J. 59, 1403–1412.

Gabrielle, B., Roche, R., Angas, P., Cantero-Martinez, C., Cosentino, L., Mantineo, M., Langensiepen, M., Henault, C., Laville, P., Nicoullaud, B., Gosse, G., 2002. A priori parameterisation of the CERES soil-crop models and tests against several European data sets. Agronomie 22, 119–132.

Génermont, S., 2014. Valeur fertilisante azotée et pertes azotées influençant la valeur fertilisante-Volatilisation. In: Valorisation Des Matières Fertilisantes D'origine Résiduaire Sur Les Sols à Usage Agricole Ou Forestier. Impacts Agronomiques, Environnementaux, Socio-économiques. Houot et Al., 2014. Expertise Scientifique Collective. INRA-CNRS-Irstea. Synthèse 113pp (http://institut.inra.fr/Missions/Eclairer-Les-decisions/Expertises/Toutes-Les-actualites/Expertise-Mafor-Effluents-Boues-et-Dechets-Organiques#).

Guo, L., Falloon, P., Coleman, K., Zhou, B., Li, Y., Lin, E., Zhang, F., 2007. Application of the RothC model to the results of long-term experiments on typical upland soils in northern China. Soil Use Manag. 23, 63–70.

Hadas, A., Molina, J. a. E., 1993. Simulation of nitrogen assimilation by heterotrophic soil microbial biomass. Physiol. Plant. 89, 664–668.

Hamiache, J., Bel, L., Vaudour, E., Gilliot, J.M., 2012. Spatial stochastic modeling of topsoil organic carbon content over a cultivated peri-urban region, using soil properties, soil types and a digital elevation model. Digit. Soil Assess. Beyond 161–164.

Houot, S., Pons, M.-N., Pradel, M., Caillaud, M.-A., Savini, I., Tibi, A., 2014. Valorisation des matières fertilisantes d'origine résiduaire sur les sols à usage agricole ou forestier. Impacts agronomiques, environnementaux, socio-économiques. Expertise scientifique collective. INRA-CNRS-Irstea. Synthèse 113pp (http://institut.inra.fr/Missions/Eclairer-les-decisions/Expertises/Toutes-les-actualites/Expertise-Mafor-effluents-boues-et-dechets-organiques#).

ITAVI, 2001. Le compostage. Sci. Tech. Avic. 53–59.

Jandl, R., Rodeghiero, M., Martinez, C., Cotrufo, M.F., Bampa, F., van Wesemael, B., Harrison, R.B., Guerrini, I.A., Richter Jr., D. deB, Rustad, L., Lorenz, K., Chabbi, A., Miglietta, F., 2014. Current status, uncertainty and future needs in soil organic carbon monitoring. Sci. Total Environ. 468–469, 376–383.

Kelly, R.H., Parton, W.J., Crocker, G.J., Graced, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter, D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using the century model. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 75–90.

Koga, N., Smith, P., Yeluripati, J.B., Shirato, Y., Kimura, S.D., Nemoto, M., 2011. Estimating net primary production and annual plant carbon inputs, and modelling future changes in soil carbon stocks in arable farmlands of northern Japan. Agric. Ecosyst. Environ. 144, 51–60.

Ladha, J.K., Reddy, C.K., Padre, A.T., van Kessel, C., 2011. Role of nitrogen fertilization in sustaining organic matter in cultivated soils. J. Environ. Qual. 40, 1756–1766.

Lal, R., Follett, F., Stewart, B.A., Kimble, J.M., 2004. Soil carbon sequestration to mitigate climate change and advance food security. Soil Sci. 304, 1623–1627.

Laville, P., Michelin, J., Houot, S., Gueudet, J.-C., Rampon, J.N., Labat, C., Vaudour, E., 2014. Soil N2O Emissions from Recovered Organic Waste Application in Versailles Plain (France): A Laboratory Approach. Waste Biomass Valorization 5, 515–527.

Leclerc, B., 2014. Document de mise à jour du Guide des Matières Organiques. ITAB, Paris.

Leclerc, B., Bouvard, F., Gazeau, G., 2012. Compostage: les principes. Sols Vivants Bio.

Makowski, D., Hendrix, E.M.., van Ittersum, M.K., Rossing, W.A.., 2000. A framework to study nearly optimal solutions of linear programming models developed for agricultural land use exploration. Ecol. Model. 131, 65–77.

Marmo, L., Feix, I., Bourmeau, E., Amlinger, F., Bannick, C.G., De Neve, S., Favoino, E., Gendebien, A., Gibert, J., Givelet, M., Leifert, I., Morris, R., Rodriguez Cruz, A., Ruck, F., Siebert, S., Tittarelli, F., 2004. Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection, vol. 3, taskegroup 4 Exogenous Organic Matter.

Mathieu, P., 2008. Synthèse de la campagne d'analyse de composts été 2007. CESAR.

Matson, P.A., Parton, W.J., Power, A.G., Swift, M.J., 1997. Agricultural Intensification and Ecosystem Properties. Science 277, 504–509.

Mauboussin, J., 2010. Intérêt du compostage dans la valorisation du fumier de cheval sur la plaine de Versailles et le plateau des Alluets. INRA Versailles-Grignon, Grignon.

Molina, J. a. E., Clapp, C.E., Shaffer, M.J., Chichester, F.W., Larson, W.E., 1983. NCSOIL, A Model of Nitrogen and Carbon Transformations in Soil: Description, Calibration, and Behavior. Soil Sci. Soc. Am. J. 47, 85–91.

Mondini, C., Coleman, K., Whitmore, A.P., 2012. Spatially explicit modelling of changes in soil organic C in agricultural soils in Italy, 2001-2100: Potential for compost amendment. Agric. Ecosyst. Environ. 153, 24–32.

Nemecek, T., Kägi, T., 2007. Life cycle inventories of U.S. agricultural production system (No. 15b). Agrosope Reckenholz-Tänikon Research Station ART, Zürich and Dübendorf.

Noirot-Cosson, P.E., Vaudour, E., Aubry, C., Gilliot, J.M., Gabrielle, B., Houot, S., 20XXa. Regional-scale scenarios of organic amendment use on cropped soils to increase soil carbon stocks and substitute synthetic fertilisation: spatial and temporal simulations with the NCSOIL/CERES-EGC crop model. Agric. Ecosyst. Environ. in prep.

Noirot-Cosson, P.E., Vaudour, E., Gilliot, J.M., Gabrielle, B., Houot, S., 20XXb. Parameterisation of the NCSOIL model to simulate short term C and N mineralisation of exogenous organic matters in different soils. Waste Manag. in prep.

Ogle, S.M., Breidt, F.J., Paustian, K., 2005. Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. Biogeochemistry 72, 87–121.

Pacini, C., Wossink, A., Giesen, G., Huirne, R., 2004. Ecological-economic modelling to support multi-objective policy making: a farming systems approach implemented for Tuscany. Agric. Ecosyst. Environ. 102, 349–364.

Peltre, C., Christensen, B.T., Dragon, S., Icard, C., Katterer, T., Houot, S., 2012. RothC simulation of carbon accumulation in soil after repeated application of widely different organic amendments. Soil Biol. Biochem. 52, 49–60.

Petit, C., Aubry, C., 2014. Collecte de grandes cultures biologiques en Île-de-France. Économie Rurale 339-340, 33–47.

Rousk, J., Brookes, P.C., Bååth, E., 2009. Contrasting Soil pH Effects on Fungal and Bacterial Growth Suggest Functional Redundancy in Carbon Mineralization. Appl. Environ. Microbiol. 75, 1589–1596.

Saffih-Hdadi, K., Mary, B., 2008. Modeling consequences of straw residues export on soil organic carbon. Soil Biol. Biochem. 40, 594–607.

Smith, P., Smith, J.U., Powlson, D.S., McGill, W.B., Arah, J.R.M., Chertov, O.G., Coleman, K., Franko, U., Frolking, S., Jenkinson, D.S., Jensen, L.S., Kelly, R.H., Klein-Gunnewiek, H., Komarov, A.S., Li, C., Molina, J.A.E., Mueller, T., Parton, W.J., Thornley, J.H.M., Whitmore, A.P., 1997. A comparison of the performance of nine soil organic matter models using

datasets from seven long-term experiments. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 153–225.

Ten Berge, H.F.M., Van Ittersum, M.K., Rossing, W.A.H., Van de Ven, G.W.J., Schans, J., 2000. Farming options for The Netherlands explored by multi-objective modelling. Eur. J. Agron. 13, 263–277.

Valé, M., 2006. Quantification et prédiction de la minéralisation nette de l'azote du sol in situ, sous divers pédoclimats et systèmes de culture français [WWW Document]. URL http://ethesis.inp-toulouse.fr/archive/00000301/ (accessed 8.19.15).

Val-Moraes, S.P., Marcondes, J., Alves, L.M.C., Lemos, E.G.M., 2011. Impact of sewage sludge on the soil bacterial communities by DNA microarray analysis. World J. Microbiol. Biotechnol. 27, 1997–2003.

Vaudour, E., Bel, L., Gilliot, J.M., Coquet, Y., Hadjar, D., Cambier, P., Michelin, J., Houot, S., 2013. Potential of SPOT Multispectral Satellite Images for Mapping Topsoil Organic Carbon Content over Peri-Urban Croplands. Soil Sci. Soc. Am. J. 77, 2122–2139.

Vaudour, E., Gilliot, J.M., Bel, L., Bréchet, L., Hamiache, J., Hadjar, D., Lemonnier, Y., 2014. Uncertainty of soil reflectance retrieval from SPOT and RapidEye multispectral satellite images using a per-pixel bootstrapped empirical line atmospheric correction over an agricultural region. Int. J. Appl. Earth Obs. Geoinformation 26, 217–234.

Vaudour, E., Noirot-Cosson, P.E., Membrive, O., 2015. Early-season mapping of crops and cultural operations using very high spatial resolution Pléiades images. Int. J. Appl. Earth Obs. Geoinformation 42, 128–141.

Yokozawa, M., Shirato, Y., Sakamoto, T., Yonemura, S., Nakai, M., Ohkura, T., 2010. Use of the RothC model to estimate the carbon sequestration potential of organic matter application in Japanese arable soils. Soil Sci. Plant Nutr. 56, 168–176.

Zhang, M., Heaney, D., Henriquez, B., Solberg, E., Bittner, E., 2006. A Four-Year Study on Influence of Biosolids/MSW Cocompost Application in Less Productive Soils In Alberta: Nutrient Dynamics. Compost Sci. Util. 14, 68–80.

Discussion générale et conclusion

Ce travail de recherche visait à approfondir les connaissances sur l'impact de l'utilisation des Produits Résiduaires Organiques (PRO) sur les cycles du carbone (C) et de l'azote (N) dans les agrosystèmes du territoire de la Plaine de Versailles, et explorer les potentialités de l'optimisation de l'insertion de ces PRO dans les systèmes de cultures à un tel niveau d'organisation spatiale, afin d'améliorer les bénéfices que permettent cette pratique tels que l'augmentation des stocks de C dans les sols ou la substitution des engrais minéraux azotés mais aussi phosphatés tout en veillant à limiter les impacts environnementaux. Ce territoire de 221 km² est caractérisé par une agriculture périurbaine (9907 ha de surface agricole utile). Il bénéficie potentiellement d'une manne assez importante de PRO variés d'origine animale et urbaine (fumiers de cheval, de bovins, fientes de volailles, et compost de déchets verts, boues d'épurations…) pouvant être utilisée dans les nombreux contextes de sol et de systèmes de cultures qui le caractérisent. Pour cette étude, le modèle de culture CERES-EGC a été utilisé. Il simule la croissance de nombreuses cultures, la dynamique de C et N et leurs flux dans les compartiments sol-plante-atmosphère, ainsi que les flux d'eau et de nitrates, de chaleur, tenant compte des effets du climat et des pratiques culturales. Ce modèle mécaniste a été utilisé afin de prendre en compte de nombreux facteurs influençant les cycles biogéochimiques du C et du N et tendre vers des prédictions réalistes. Le paramétrage du module NCSOIL simulant la dynamique de la Matière Organique (MO) au sein de CERES-EGC a été réalisé pour les principaux types de sol et les PRO du territoire sur la base du suivi des dynamiques de minéralisation du C et du N des sols et des PRO en conditions contrôlées de laboratoire. Des caractéristiques analytiques et des résultats de fractions biochimiques ont également été utilisés pour le paramétrage NCSOIL des PRO. Les types de sol du territoire n'ont pas impacté significativement le paramétrage des PRO pouvant ainsi être utilisé quel que soit le sol auquel les PRO sont apportés. Les paramètres NCSOIL des sols et des PRO ont permis de correctement simuler avec CERES-EGC la dynamique de la MO au champ des 13 ans d'applications de PRO dans la succession simplifiée maïs-blé de l'expérimentation QualiAgro, située au cœur du territoire d'étude. Les 18 PRO du territoire ont été classés selon 4 types : (1) composts stables, (2) composts moins matures restant plus réactifs et les fumiers stables, (3) des fumiers très réactifs correspondant à des fumiers de chevaux, (4) des PRO très réactifs tels que des boues et des fientes pouvant plutôt être utilisés comme fertilisants. Les cartes de successions culturales, de sols et de stocks de carbone ont été croisées pour établir les différents ilots culturaux du territoire dans lesquels se fait l'insertion des PRO dans les systèmes de cultures et pour lesquels ont été simulés les dynamiques de C et du N. De nombreux scenarios de 20 ans d'apports de PRO ont été élaborés, se voulant techniquement réalistes, et dont les quantités d'apports ont été limitées sur la base du phosphore (P) et du N pour limiter a priori des fuites dans l'environnement, tenant également compte de la réglementation concernant les épandages et les flux d'ETM associés. Ces scenarios ont été simulés dans les différents contextes de sol, de MO et de successions culturales du territoire permettant ainsi déjà d'appréhender les principaux facteurs d'influence du devenir du C et N de la MO

du sol enrichie par les PRO. Le type de sol s'est révélé un facteur déterminant pour le stockage de carbone et la lixiviation et les successions culturales pour les substitutions d'azote de synthèse grâce aux apports de PRO. Les contraintes P et N des scenarios ont autorisé des apports plus importants des PRO amendants que fertilisants augmentant ainsi le potentiel des composts à augmenter la MO des sols déjà supérieurs à celui des autres PRO de par la stabilité de leur MO. Ces apports plus importants ont permis des économies en engrais azotés de synthèse grâce aux PRO amendants aussi importantes qu'avec les PRO fertilisants, en raison notamment de la prise en compte de l'augmentation des stocks de MO des sols dans le calcul des économies d'engrais potentielles. Les économies en engrais azotés rapportées aux quantités de N appliquées par les PRO sont supérieures avec les PRO fertilisants en particulier lorsqu'apporté sur maïs, mais peuvent atteindre des valeurs élevés avec des PRO amendants peu stables lorsqu'apportés sur colza. Les simulations des effets de ces scenarios d'apports de PRO variés dans les différents contextes du territoire ont été utilisées pour optimiser la distribution des PRO du territoire sur les surfaces cultivées de la Plaine de Versailles, avec pour objectif de maximiser les bénéfices de l'utilisation des PRO au niveau du territoire, en termes de stockage de carbone dans le sol ou de substitution des engrais azotés de synthèse, ou de minimiser les lixiviations de nitrates. Au niveau régional, l'application préférentielle des PRO amendants les plus stables sur les sols les plus riches en argile et calcaire a permis le stockage de carbone maximum. L'application préférentielle des PRO fertilisants sur les successions à maïs et des PRO amendants peu stables sur les successions à colza a permis les plus grandes économies au niveau régional d'engrais azotés de synthèse.

Généricité du paramétrage de la MO des PRO et reproductibilité de la méthode

Que ce soit lors du calibrage du modèle CERES-EGC avec des données de champ et de long-terme, ou lors de la paramétrisation des PRO du territoire à partir de mesures de laboratoire, un effort de généricité a toujours été recherché en vue de rendre la méthode plus simple et transposable à d'autres situations ainsi que les paramètres trouvés réutilisables ou facilement estimables. Dans le chapitre 3, nous avons vu que les cinétiques expérimentales de minéralisation du C et du N des PRO diffèrent très peu avec le type de sol utilisé pour ces expériences, et que les paramètres décrivant les PRO au sein du module NCSOIL (taille de compartiments, rapports C/N, vitesses de décomposition) ne sont pas sensibles au type de sol dans lequel les PRO sont apportés indiquant que ces paramètres sont réutilisables dans d'autres contextes pédologiques pour des types de PRO identiques. De plus, des paramètres ou indicateurs facilement mesurables tels que l'indice de stabilité de la MO (ISMO ou I_{ROC}; Lashermes et al., 2009) pour l'estimation de la fraction récalcitrante de la MO des PRO et le rapport C/N organique (CN_{EOM}) affecté aux 2 fractions labile et récalcitrante des PRO sont adéquats pour les 3 composts et le fumier de bovin épandus pendant 13 ans au champ dans l'expérimentation QualiAgro. Toutefois ces indicateurs ne sont appropriés que pour des PRO très stables de types

composts plutôt matures, ou très labiles et riches en N comme les boues et les fientes. En revanche des PRO peu stables mais plutôt riches en C comme les fumiers ont besoin d'une autre paramétrisation. Un nouvel indicateur basé comme l'ISMO, sur une combinaison des fractions biochimiques et le C/N des PRO a été calculé pour ces derniers PRO mais nécessiterait d'être consolidé sur un échantillon plus fourni de PRO différents. Enfin, pour les paramètres « constantes de décomposition » de la MO des PRO, des classifications basées sur ces paramètres dynamiques ajustés et des caractéristiques analytiques des PRO ont été réalisées pour pouvoir déterminer des classes de PRO ayant les mêmes comportements dynamiques. D'autres recherches précédentes ont voulu caractériser la dynamique de la MO de groupes de PRO pour prédire plus facilement la dynamique d'un nouveau PRO (Parnaudeau et al., 2004; Morvan et al., 2006; Lashermes et al., 2010) sans aller toujours jusqu'à leur paramétrisation dans un modèle. Si la dynamique du C semble pouvoir être simulée assez facilement, cela est plus délicat pour le N. Lashermes et al. (2010) a tenté une approche typologique basée sur la minéralisation du N reliée aux fractions biochimiques des PRO sans que cette approche n'ait été testée au champ. On constate que des mêmes comportements dynamiques de minéralisation C et N peuvent être observés à partir de distributions différentes de la MO des PRO entre fractions labile/récalcitrant ainsi que des paramètres dynamiques différents, ce qui complexifie la potentialité de généricité du paramétrage du modèle.

Paramétrisation et transfert d'échelle

Dans le chapitre 2, l'ensemble des PRO disponibles au niveau du territoire d'étude ont été caractérisés en conditions contrôlées de laboratoire, résultats qui ont servi à paramétrer le module MO du modèle CERES-EGC pour tester ensuite l'insertion des PRO dans les systèmes de cultures du territoire. Cette démarche a démontré son intérêt et sa validité sur le site de longue durée QualiAgro où 4 PRO sont apportés régulièrement depuis 15 ans. On peut se demander si cela reste valable pour un plus large panel de PRO. Le modèle NCSOIL a toujours été utilisé pour simuler le devenir de la MO des PRO dans le sol dans des conditions de laboratoire (Corbeels et al., 1999; Nicolardot et al., 2001; Burger and Venterea, 2008; Antil et al., 2011) et très peu pour des simulations au champ s'intéressant au suivi de stocks de MO et d'évolution de N minéral dans les sols. Les quelques simulations faites ne concernaient que quelques années (Gabrielle et al., 2005). D'ailleurs, la simulation par CERES-EGC de l'augmentation de la MO lors d'apports de PRO se fait surtout par une accumulation de la fraction récalcitrante de la MO du PRO, alors que les études ne montrent pas de changement de la qualité de la MO du sol après apports répétés de PRO (Paetsch et al., submitted). Le transfert de paramétrage réalisé sur la base d'incubations en conditions contrôlées de laboratoire a également été réalisé pour la MO du sol. Ce transfert s'est avéré plus délicat. En effet, si les données d'observation de longue durée permettent de caler les paramètres dynamiques de la MO du sol, nous ne dispositions de ces données que pour 1 type de sol du territoire (Luvisol sur argiles à meulières, haplic luvisol (retic) de l'essai

Qualiagro) mais pas pour les autres sols de la plaine de Versailles. Ainsi, une méthode de « *spin up run* » a été adoptée pour le paramétrage de la MO de ces autres sols, méthode classiquement employée pour des simulations de long-terme notamment sur l'évolution du CO du sol (Coleman et al., 1997; Thornton and Rosenbloom, 2005; Mondini et al., 2012; Exbrayat et al., 2014; Fang et al., 2015). Elle est facile et reproductible mais nécessiterait de plus amples validations. Le chapitre 4 nous a néanmoins montré que quelles que soient les variables étudiées parmi le stockage de carbone (CS) et la lixiviation de nitrates (NL) surtout, pour les économies en engrais de synthèse (NS) dans une moindre mesure, et notamment lors de l'apport de PRO, la MO du sol, de par son niveau, avait bien moins d'influence que le type de sol, caractérisé par sa profondeur, son régime hydrique ainsi que sa texture et sa teneur en calcaire dans l'horizon de labour, paramètres qui conditionnent effectivement le stockage de C et la lixiviation du N. Finalement pour les prédictions du devenir de la MO dans le sol lors de l'apport de PRO, il semblerait que plus d'attention devrait être portée sur la quantification de ces caractéristiques de sol.

Spatialisation des stocks de carbone organique des sols initiaux

La première étape de cette étude a été la spatialisation des stocks de CO des sols c'est-à-dire de définir spatialement les paramètres de l'équation des stocks de CO. Celle-ci est loin d'être facile comme mentionné dans le chapitre 1 de cette thèse et de nombreuses erreurs et biais sont possibles. Pour ce faire un certain nombre de simplifications ont été utilisées d'abord pour effectivement réussir à spatialiser les facteurs de calculs des stocks, mais également pour simplifier l'étape de simulations territoriales. La densité de terre fine a été utilisée intégrant ainsi la pierrosité lorsque celle-ci reste limitée. La pierrosité n'a donc pas totalement été prise en compte, son estimation et en particulier de sa grande variabilité spatiale étant délicate. La densité a été moyennée par type de sol car son estimation ponctuelle est peu précise et temporellement variable, et sa variabilité importe peu pour la variabilité des stocks (Goidts et al., 2009). Seule la teneur en CO des sols a été mesurée et spatialisée avec une RMSE moyenne de 3.92 $g.kg^{-1}$ (Hamiache et al., 2012) comme conseillé par Goidts et al. (2009) concernant l'importance des paramètres de l'équation des stocks. Néanmoins celle-ci a finalement été simplifiée pour faciliter les simulations territoriales et leur interprétation. Si l'estimation et la spatialisation précise des stocks de CO peut être sensible à de nombreux biais générant probablement d'importantes incertitudes sur les stocks totaux d'un territoire, on a néanmoins vu au cours de cette étude que le stockage de C grâce à l'apport de PRO dépendait plus d'autres caractéristiques de sol et des scenarios d'utilisation de PRO que du stock initial en C des sols. Ainsi, l'erreur sur l'estimation du stock initial de C n'impliquerait donc que peu d'erreur dans l'estimation du stockage de C, ceci étant valable dans des conditions éloignées de la saturation en CO des sols. Ceci justifie a posteriori nos choix de simplification de spatialisations des stocks initiaux.

Cependant, la paramétrisation du CO des sols en tenant compte de leurs teneurs en carbonates et argiles a été faite sur la base de résultats d'incubations au laboratoire, mais pour une seule teneur en CO par type de sol. Pour un même type de sol nous avons fait l'hypothèse que les caractéristiques du pool de MO active restaient inchangées quelle que soit la teneur en CO du sol en question. L'extrapolation des simulations sur les différents sols et avec différentes teneurs en CO nécessiterait quelques validations. Des recherches plus approfondies seraient intéressantes pour mieux évaluer les effets relatifs des types de sols et des différentes teneurs en CO du sol. D'autres artefacts sont induits par les incubations au laboratoire tels que la désagrégation du sol limitant le rôle de la structure du sol sur la dynamique de la MO (Vogel et al., 2015).

Calculs des économies en azote de synthèse

Les applications de PRO génèrent d'importants apports de N pouvant être utilisés par les cultures et pouvant se substituer à tout ou partie de la fertilisation minérale. Le calcul de la quantité de N venant du PRO qui sera effectivement utilisée par les cultures des années suivantes est très complexe à cause du caractère progressif de la minéralisation de ce N, de l'accumulation dans le sol de CO et des multiples facteurs qui influent sur ces dynamiques. Les effets directs (disponibilité du N dans l'année de l'apport des PRO) sont mieux connus que les effets à plus long terme ou encore les effets liés à des apports répétés. Le COMIFER propose des coefficient équivalent engrais (Comifer, 2013) permettant d'estimer cette quantité pour un très grand nombre de PRO et pour un grand nombre de cultures amendées. Néanmoins non seulement, cela ne concerne que l'année de l'apport mais en plus cela ne tient pas compte des nombreux effets possibles du sol. Le calcul de l'économie en engrais de synthèse qui a été proposé prend en compte l'effet sol et la fourniture en N par le PRO plusieurs années après apports ainsi qu'en cas d'apport répétés dans le temps, prenant ainsi en compte l'augmentation des stocks de MO dans le sol. Si la méthode de calcul est extrapolable, elle reste complexe et les régressions obtenues sont spécifiques des sols et des cultures du territoire et ne sont donc pas directement réutilisables. En dépit de ces limites, nous avons considéré qu'un calcul tel que celui-ci devait être entrepris étant donné l'enjeu considérable des économies en N de synthèse.

Exploration du bilan des effets de l'application des PRO sur le territoire

En dépit des difficultés liées aux simulations, celles-ci se sont avérées réalistes en comparaison avec d'autres études spatialisées sur les amendements organiques et leurs impacts sur la MO du sol. Nos simulations ont révélé des potentiels très divers selon les types de PRO et leur contexte d'épandage. En plus des caractéristiques intrinsèques des PRO, les effets ont été beaucoup influencés par les quantités potentielles épandues, elle-même contraintes à l'amont par la définition des scenarios basés sur les quantités de N et de Phosphore (P) pour limiter a priori des pollutions relatives aux N et P. En

général, les composts (de déchets verts en particulier) et les fumiers (surtout de cheval), peuvent être apportés en grandes quantités, alors que les PRO fertilisants sont limités par leur richesses en N et P.
Les grandes tendances sont que les PRO de type compost dont la MO est stable permettent le stockage de C le plus important et le plus efficace et des économies en engrais de synthèse moyennes. Les PRO fertilisants génèrent très peu de stockage de C mais des économies en engrais de synthèses plutôt fortes et surtout très efficaces (fort pourcentage du N apporté substitué aux engrais). Les fumiers, surtout de chevaux, en dépit de grandes quantités épandables, permettent un stockage de C plutôt faible car peu efficace (faible pourcentage de C apporté retrouvé dans les sols), et des économies en N moyennes. L'apport de PRO génère en général peu de lixiviation de nitrate supplémentaire car la fertilisation N a été raisonnée, mais peut en générer beaucoup lors de l'apport de PRO amendants. En effet ces PRO peuvent être apportés en plus grandes quantités car ils sont peu contraints par leur teneur en P et la minéralisation de leur N n'est pas forcément synchrone des besoins des plantes.
Les contextes de sol et de succession de cultures se sont révélés très importants concernant les impacts sur le cycle de C et N lors de l'application de PRO. D'ailleurs, les différents sols génèrent autant de variabilité sur le stockage de C que les différents grands types PRO en cas d'un apport dans la succession, et ils génèrent plus de variabilité sur les lixiviations de nitrates. Néanmoins, les caractéristiques des PRO modulent également les résultats, de par les quantités variables qui peuvent être apportées en raison de leurs teneurs différentes en N, P et CO. Les économies en engrais de synthèse se sont avérées plus fortement influencées par les successions culturales, les successions à colza pouvant mieux bénéficier de l'apport de N des PRO amendants et les successions à maïs de celui des PRO fertilisants. Ces résultats sont aussi liés aux hypothèses faites pour la construction des scenarios et pour le calcul de la substitution d'engrais azoté de synthèse. Par exemple dans nos scenarios, les PRO fertilisants sont apportés à la fin de l'hiver ; ils perdraient de leur efficacité à la substitution en engrais si apportés à la fin de l'été comme les amendants. De plus, il a été considéré que l'azote labile des PRO amendants qui se minéralise rapidement en automne pouvait être totalement prélevé par le colza, alors qu'il serait inutilisable pour des cultures de printemps, en dépit du prélèvement et re-minéralisation potentiel par les cultures intermédiaires.

Autres impacts de l'utilisation des PRO

L'utilisation des PRO par l'agriculture génère de nombreux impacts et tous n'ont pas été étudiés comme par exemple la stimulation de l'activité biologique dans les sols ainsi que des impacts environnementaux tels que flux de contaminants. Ainsi, l'augmentation de la MO a été choisie car elle constitue non seulement une compensation des émissions de gaz à effet de serre (GES), mais elle constitue aussi un indicateur clé car elle implique la stimulation de nombreux services écosystémiques des sols. Ensuite, les économies en engrais de synthèse générés par l'utilisation des PRO ont également été choisies de par l'intérêt important pour les agriculteurs et les émissions de GES

indirectes qu'elles peuvent éviter. Parmi les diverses émissions azotées, la lixiviation de NO_3, volatilisation de NH_3 ont été étudiées parce que constituant les deux principales pertes de N diminuant l'efficacité de la fertilisation, les émissions de N_2O car gaz à effet de serre très puissant ainsi que le N_2 car associé aux émissions de N_2O par dénitrification. Si les lixiviations de nitrate voire les émissions d'ammoniac ont fait l'objet de mesures dans le cadre de l'expérimentation QualiAgro et donc d'une certaine calibration, les autres émissions n'ont été que simulées. Néanmoins, le modèle a déjà montré qu'il permettait d'estimer correctement les émissions moyennes de N_2O (Gabrielle et al., 2006). En dehors des cycles C et N, les PRO peuvent avoir de nombreux autres impacts. Comme présenté dans Obriot et al. (in prep.)leurs applications peuvent impacter la qualité du sol, sa fertilité, son activité biologique, sa biodiversité, ses propriétés physiques, son état sanitaire et la qualité des récoltes. Sans prendre en compte de notions d'échangeabilité/disponibilité, les apports de P et d'Eléments Traces Métalliques (ETM) ont été intégrés dans la constitution des scénarios soit en limitant les apports aux exports de P par les plantes, soit en respectant la réglementation sur les ETM dans les produits et les flux apportés. Si le potentiel de substitution des engrais azotés de synthèse a été calculé, il serait intéressant de compléter cela par le calcul de ce même potentiel pour les éléments P et potassium.

A propos de l'organisation de la distribution des PRO

L'optimisation de la distribution des PRO à l'échelon territorial a d'abord montré de sérieux potentiels d'amélioration du stockage de C ou des économies en N de synthèse. En poursuivant un objectif plutôt qu'un autre on peut doubler le stockage de C ou les économies de N. Il s'agit pour cela de sélectionner certains PRO disponibles sur le territoire et de bien choisir où les épandre et en quelles quantités. Cette dernière étape de la thèse correspond bien à une exploration des potentiels maxima prenant en compte des paramètres environnementaux et agronomiques essentiellement. Cependant il est vrai qu'un certain nombre d'autres éléments pourraient rendre difficile l'accession à ces potentiels maxima (cf plus loin). Lorsque plusieurs objectifs sont poursuivis simultanément, des compromis doivent être étudiés. Néanmoins, cette étude a déjà intégré un certain nombre d'impacts. La recherche des impacts existants ainsi que des impacts à prendre en compte dans l'évaluation de la pratique d'apports organiques doit encore progresser, et elle est un préalable à la recherche de compromis.

Originalité de l'étude

L'utilisation de la programmation linéaire a récemment été employée pour l'agriculture, bien souvent pour tenter de trouver des systèmes de production plus rentables, ou permettant une meilleure préservation de la MO des sols, incluant alors parfois des itinéraires techniques tel que l'apport de PRO (Ten Berge et al., 2000 ; Dogliotti et al., 2005; Groot et al., 2012). Ces études ont toujours concerné le niveau d'organisation spatiale de l'exploitation. L'utilisation de tels outils pour

l'organisation de l'usage de la MO et pour en améliorer ses bénéfices à l'échelon territorial est inédite. L'étude des impacts de l'apport de PRO d'un si large panel, dans le cadre de scénarios variés et réalistes, intégrant autant de facteurs environnementaux est en elle-même originale. De plus, si des études portent simultanément sur les impacts sur le C et le N, aucune n'intègre en plus le P alors qu'il constitue un enjeu majeur à cause de son rôle important dans l'eutrophisation. La modélisation des économies en engrais azotés de synthèse est elle aussi originale dans la mesure où elle est adaptée aux contextes pédo-climatiques de la région, étant basée sur des quantités de N effectivement minéralisées, et prenant en compte des effets à long-terme de l'application de PRO. Enfin, l'étude est originale par sa volonté d'être reproductible en particulier en s'efforçant de rendre possible un paramétrage simplifié des PRO et donc des simulations de l'impact de leur application avec CERES-EGC même dans d'autres contextes.

Limites à l'étude prospective

Pour cette exploration, des scénarios d'apports de PRO ont été imaginés. Ceux-ci se sont d'abord voulus réalistes et ont été basés sur plusieurs expertises agronomiques ainsi que de nombreuses enquêtes auprès des agriculteurs (Dhaouadi, 2014). Par exemple ils ont pris en compte : les itinéraires techniques opérés sur le territoire, les dates d'apport de PRO généralement choisies ainsi que d'autres dates possibles compte tenu des calendriers culturaux. De plus, l'imputation du surplus de N disponible lors de l'apport de PRO a été réalisé à des dates spécifiques tenant compte des pics de besoins de plantes et des exigences en terme de teneur en N des grains (cas du blé pour la panification par exemple). Ils ont également pris en compte le fait qu'un agriculteur n'effectuerait de passage dans son champ que pour des quantités d'engrais supérieures à un minimum. Ces scénarios ont également pris en compte les dispositions de la réglementation liés à la directive nitrate: obligation d'interculture avant les cultures de printemps, des périodes et des quantités d'épandages restreintes. A également été pris en compte le risque de surplus de P anticipant d'ailleurs sur la réglementation.

Malgré toutes ces dispositions et pour renforcer cette étude prospective, un certain nombre d'éléments supplémentaires devraient être pris en compte et nécessitent d'être cités. La pratique d'épandage de PRO rencontre parfois des réticences de la part du grand public ou des acteurs locaux, auxquels les agriculteurs ne peuvent rester indifférents. En effet à cause de potentielles nuisances olfactives, de peur de pollutions ou d'impacts imprévus ou de cahier des charges de production, le public et les agriculteurs s'opposent parfois à cette pratique au point de la faire interdire dans certaines communes. De plus, la pratique d'épandage fait partie des pratiques agricoles dont les agriculteurs sont les décideurs au sein de leur exploitation. En effet, si l'exploration des impacts des PRO s'est d'abord faite au niveau de l'îlot (chapitre 4) puis au niveau du territoire (chapitre 4 et 5), elle a sauté l'échelon « exploitation » qui est pourtant un échelon clé, car c'est à cet échelon que les agriculteurs pensent leur système de production. Ce saut a été réalisé ici pour explorer les potentialités d'une organisation à

large emprise spatiale de l'utilisation des PRO. La prise en compte de cet échelon intermédiaire et les considérations socio-économiques associées avant d'aboutir à une étude territoriale constituent une prochaine étape.

De plus, parmi les variables étudiées au niveau du territoire, plusieurs l'étaient dans le but d'estimer les impacts environnementaux à ce même niveau, par des distributions variables des PRO. Or cette distribution implique le transport des matières et différentes distributions impliquent également des émissions de GES associées au transport. Ainsi, pour une étude plus complète sur les potentiels de compensation d'émission de GES, le transport devrait être intégré.

Un autre élément manque peut-être à cette étude qui se veut prospective : l'effet du changement climatique. En effet, dernièrement de nombreuses recherches étudiant l'évolution passée et/ou future du CO des sols prennent en compte le changement climatique dont l'effet pourrait s'avérer substantiel. Néanmoins, prenant en compte cet effet, ces dernières n'étudient que très peu de produits et de scenarios différents. Nous nous sommes au contraire focalisés sur ces derniers aspects, démarche nécessaire pour appréhender une organisation territoriale réaliste de la MO.

Perspectives

Les objectifs principaux de cette étude étaient (i) de mieux connaître l'impact de l'utilisation des PRO lors de leur apport au champ, (ii) d'explorer le potentiel de maximisation des bénéfices de cet usage au niveau d'organisation spatiale d'un territoire.

On a vu que le paramétrage de la MO du sol était délicat et qu'un paramétrage par le biais de mesures labo parallèlement à l'utilisation de la méthode du « spin up run » était nécessaire avant d'étudier les effets des apports de PRO. Des validations sont encore nécessaires pour attester de la qualité des résultats lors de l'emploi de cette méthode. De même, le paramétrage de la MO des PRO déterminé grâce aux mesures en laboratoire s'est montré adéquat lorsqu'utilisé pour des simulations d'amendements organiques au champ, mais la validité de ce transfert de paramétrage n'a été vérifiée que pour 4 PRO, et une validation similaire serait nécessaire pour d'autres types de PRO et dans d'autres conditions de sol et de cultures. Ainsi, si cette étude basée sur de la modélisation s'appuie sur des mesures d'impacts réels de l'apport de PRO au champ, de nombreux résultats qui en ressortent proviennent de l'extrapolation de mesures labo (paramètres MO des sols et des PRO, ainsi que paramètres hydriques des sols), et nécessiteraient d'être validées par des données de champ de long terme. Des efforts restent à faire pour faciliter le paramétrage des PRO via l'utilisation d'indicateurs facilement estimables, ou de paramètres ou de méthodes de paramétrages spécifiques à des groupes de PRO. La variabilité spatiale de certains paramètres liés au sol tels que la texture, les paramètres hydriques ou liés aux systèmes de cultures tels que les successions de cultures au sein des îlots ou les itinéraires techniques utilisés nécessiterait également d'être mieux prise en compte pour des estimations plus précises des impacts de l'utilisation des PRO au niveau du territoire.

Les études des impacts de l'application de PRO au champ doivent encore être poursuivies en diversifiant les effets étudiés pour que leurs prédictions soient précisées et intégrées dans une évaluation multi-critères. Pourraient être étudiés, les effets positifs sur la stimulation de l'activité biologique du sol, sur l'amélioration de la structure du sol impliquant une meilleure résistance à l'érosion ou au tassement, une meilleure rétention d'eau, mais également les risques en termes de contaminations en éléments traces métalliques, en polluants organiques, en pathogènes, en gènes de résistance aux antibiotiques. L'optimisation territoriale de l'utilisation des PRO a été explorée en poursuivant certains objectifs relatifs aux impacts des apports de PRO sur le C et le N. Néanmoins cette optimisation pose la question de la multiplicité fréquente des objectifs poursuivis à travers l'utilisation des PRO. Il faudrait définir une méthode d'évaluation les conciliant dans cette démarche l'optimisation. La concertation et la collaboration avec différents acteurs et différentes disciplines scientifiques permettraient de mieux définir les objectifs de l'utilisation des PRO et de leur importance relative. Les critères utilisés pour la définition des scénarios d'insertion des PRO dans les systèmes de cultures puis pour l'optimisation territoriale de leur distribution, sont principalement environnementaux et agronomiques. Malgré la petite emprise territoriale considérée, la prise en compte du transport, notamment pour le bilan des émissions de CO_2, semble importante. Un certain nombre de critères économiques et sociaux devraient être intégrés également. D'autres scenarios mélangeant les types de PRO apportés sur une culture ou dans la succession sont à étudier. Enfin les interactions entre apports de PRO et d'autres modifications dans les systèmes de cultures, telles que l'insertion de légumineuses devraient également être étudiées car ces différentes innovations sont souvent envisagées de façon concomitante par les agriculteurs alors qu'elles pourraient avoir un fort impact sur le bilan environnemental de l'agriculture en particulier sur les cycles C et N.

References

Antil, R.S., Bar-Tal, A., Fine, P., Hadas, A., 2011. Predicting Nitrogen and Carbon Mineralization of Composted Manure and Sewage Sludge in Soil. Compost Sci. Util. 19, 33–43.

Burger, M., Venterea, R.T., 2008. Nitrogen Immobilization and Mineralization Kinetics of Cattle, Hog, and Turkey Manure Applied to Soil. Soil Sci. Soc. Am. J. 72, 1570.

Coleman, K., Jenkinson, D.S., Crocker, G.J., Grace, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter, D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. Geoderma, Evaluation and Comparison of Soil Organic Matter Models 81, 29–44.

Comifer, 2013. Calcul de la fertilisation azotée Guide méthodologique pour l'établissement des prescriptions locales. Comifer.

Corbeels, M., Hofman, G., Cleemput, O.V., 1999. Simulation of net N immobilisation and mineralisation in substrate-amended soils by the NCSOIL computer model. Biol. Fertil. Soils 28, 422–430.

Dogliotti, S., van Ittersum, M.K., Rossing, W.A.H., 2005. A method for exploring sustainable development options at farm scale: a case study for vegetable farms in South Uruguay. Agric. Syst. 86, 29–51.

Exbrayat, J.-F., Pitman, A.J., Abramowitz, G., 2014. Response of microbial decomposition to spin-up explains CMIP5 soil carbon range until 2100. Geosci. Model Dev. 7, 2683–2692.

Fang, Y., Liu, C., Leung, L.R., 2015. Accelerating the spin-up of the coupled carbon and nitrogen cycle model in CLM4. Geosci. Model Dev. 8, 781–789.

Gabrielle, B., Da-Silveira, J., Houot, S., Michelin, J., 2005. Field-scale modelling of carbon and nitrogen dynamics in soils amended with urban waste composts. Agric. Ecosyst. Environ. 110, 289–299.

Gabrielle, B., Laville, P., Henault, C., Nicoullaud, B., Germon, J.C., 2006. Simulation of Nitrous Oxide Emissions from Wheat-cropped Soils using CERES. Nutr. Cycl. Agroecosystems 74, 133–146.

Goidts, E., Van Wesemael, B., Crucifix, M., 2009. Magnitude and sources of uncertainties in soil organic carbon (SOC) stock assessments at various scales. Eur. J. Soil Sci. 60, 723–739.

Groot, J.C.J., Oomen, G.J.M., Rossing, W.A.H., 2012. Multi-objective optimization and design of farming systems. Agric. Syst. 110, 63–77.

Hamiache, J., Bel, L., Vaudour, E., Gilliot, J.M., 2012. Spatial stochastic modeling of topsoil organic carbon content over a cultivated peri-urban region, using soil properties, soil types and a digital elevation model. Digit. Soil Assess. Beyond 161–164.

Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuries, L., Chaussod, R., Guillotin, M.L., Lineres, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S., 2009. Indicator of potential residual carbon in soils after exogenous organic matter application. Eur. J. Soil Sci. 60, 297–310.

Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuriès, L., Chaussod, R., Guillotin, M.L., Linères, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S., 2010. Typology of exogenous organic matters based on chemical and biochemical composition to predict potential nitrogen mineralization. Bioresour. Technol. 101, 157–164.

Mondini, C., Coleman, K., Whitmore, A.P., 2012. Spatially explicit modelling of changes in soil organic C in agricultural soils in Italy, 2001-2100: Potential for compost amendment. Agric. Ecosyst. Environ. 153, 24–32.

Morvan, T., Nicolardot, B., Péan, L., 2006. Biochemical composition and kinetics of C and N mineralization of animal wastes: a typological approach. Biol. Fertil. Soils 42, 513–522.

Nicolardot, B., Recous, S., Mary, B., 2001. Simulation of C and N mineralisation during crop residue decomposition: A simple dynamic model based on the C:N ratio of the residues. Plant Soil 228, 83–103.

Obriot, F., Stauffer, M., Goubard, Y., Cheviron, N., Peres, G., Eden, M., Revallier, A., in prep. Multi-criteria indices to evaluate the effects of repeated organic amendment applications on soil and crop quality. Agric. Ecosyst. Environ.

Parnaudeau, V., Nicolardot, B., Pagès, J., 2004. Relevance of Organic Matter Fractions as Predictors of Wastewater Sludge Mineralization in Soil. J. Environ. Qual. 33, 1885–1894.

Ten Berge, H.F.M., Van Ittersum, M.K., Rossing, W.A.H., Van de Ven, G.W.J., Schans, J., 2000. Farming options for The Netherlands explored by multi-objective modelling. Eur. J. Agron. 13, 263–277.

Thornton, P.E., Rosenbloom, N.A., 2005. Ecosystem model spin-up: Estimating steady state conditions in a coupled terrestrial carbon and nitrogen cycle model. Ecol. Model. 189, 25–48.

Vogel, L.E., Makowski, D., Garnier, P., Vieublé-Gonod, L., Coquet, Y., Raynaud, X., Nunan, N., Chenu, C., Falconer, R., Pot, V., 2015. Modeling the effect of soil meso- and macropores topology on the biodegradation of a soluble carbon substrate. Adv. Water Resour. 83, 123–136.

Title : Optimising Exogenous Organic Matter use in cropping systems of a francilian region: evolution of soil organic carbon stocks and substitution of mineral fertiliser

Keywords : Exogenous Organic Matter, Compost, cropping system, soil carbon storage, mineral N saving, modelling, regional optimisation

Abstract :

The use of Exogenous Organic Matter (EOM) in agriculture could be an efficient way to substitute mineral fertilisation and increase soil organic matter (SOM) enhancing soil fertility and storing carbon (C). It could also cause nitrogen (N) pollutions such as nitrate leaching and gas emissions. Better understanding of C and N fate after EOM applications on cropped soils would allow improving these benefits while limiting environmental impacts.

This thesis aims at: (i) predicting EOM impacts when applied on cropped soils, (ii) studying the effects of various scenarios of EOM applications in terms of C storage, synthetic N saving and N pollutions in the context of the Plain of Versailles region (221 km²) and taking into account soil diversity, crop successions and soil organic C contents, (ii) studying the potential for improving these benefits at the regional scale with an optimal distribution of EOM.

The CERES-EGC crop model was used to simulate the effects of repeated applications of EOM over 13 years on both soil C and N dynamics in the soil-crop-water-air system of the long-term field experiment QualiAgro located within the region. The sub-model NCSOIL was parameterised from C and N mineralisation kinetics of EOM measured in laboratory conditions. When transposing the parameters into the CERES-EGC model, C storage at the field scale was well simulated, together with crop N uptake and yields, as well as soil mineral N contents.

The kinetics of C and N mineralisation of the 18 EOM available in the region were used along with EOM biochemical fractionations for parameterising the NCSOIL model. The soil type did not significantly change EOM parameters. Four groups of EOM were distinguished based on their C and N dynamics: (i) stable composts, (2) more reactive and less mature composts and stable manures, (3) manures with reactive OM corresponding to horse manures and (4) very reactive EOM as sludges, litters that should be used as fertilisers.

Numerous scenarios of EOM applications, constrained on the phosphorus and N quantities they bring (and limiting the input in trace elements), were simulated for 20 years in all regional contexts of soil, crop successions and soil organic C contents. The soil type was the main factor controlling C storage and N leaching while it was crop successions for N saving. Some composts allowed C storage up to 1.1 t C ha^{-1} yr^{-1} reaching 63% of C applied. N saving of 74 kg N ha^{-1} yr^{-1} were possible with a dried sewage sludge and a compost. N substitution could reach more than 90% of N applied with EOM, these high percentages being related with the indirect effect of EOM on soil OM and the hypothesis made for N substitution

An optimisation model was developed to select EOM application scenario for each crop plot (soil x crop succession x initial soil OC content x area) accounting for EOM availability in the region with the objectives of maximising C storage or synthetic N saving or minimising N leaching at the regional scale. Applying preferentially the most stable EOM on soils with the highest potential for C storage i.e. with the highest calcareous and clay contents, up to 0.47 t C ha^{-1} yr^{-1} could be stored. Applying preferentially fertilising EOM on crop succession with maize and amending EOM on succession with rapeseed, up to 53 kg N ha^{-1} yr^{-1} could be saved.

Titre Optimisation de l'insertion des Produits Résiduaires Organiques dans les systèmes de cultures d'un territoire francilien: évolution des stocks de carbone organique des sols et substitution des engrais minéraux

Mots clés : Produits Résiduaires Organique, compost, système de culture, stockage carbone sol, économies d'engrais minéraux, modélisation, optimisation territoriale

Résumé :

L'utilisation des Produits Résiduaires Organiques (PRO) par l'agriculture peut permettre de substituer une partie des engrais minéraux et augmenter les niveaux de matière organique des sols. Elle peut également générer des pollutions azotées via la lixiviation de nitrate ou des émissions gazeuses. Mieux connaitre le devenir du carbone (C) et de l'azote (N) après applications de PRO sur les sols agricoles contribue à améliorer les bénéfices de cette pratique et à en limiter les impacts environnementaux.

Cette thèse a pour objectif: (i) de prédire les dynamiques du C et du N en cas d'applications de PRO sur les sols agricoles, (ii) d'étudier les effets de scenarios d'apports de PRO sur les stocks de carbone dans les sols, la substitution des engrais azotés et les pollutions azotées dans le contexte du territoire de la Plaine de Versailles, en prenant en compte la diversité des sols, de leurs teneurs en C et des successions culturales, (iii) d'améliorer ces bénéfices à l'échelon du territoire via une distribution optimale des PRO.

Le modèle CERES-EGC a été utilisé pour simuler les effets de 13 ans d'apports de PRO sur les dynamiques de C et de N dans le système sol-plante-atmosphère de l'expérimentation QualiAgro, située au cœur du territoire d'étude. Le sous-modèle NCSOIL a été paramétré à partir de cinétiques de minéralisation de C et N mesurées en conditions contrôlées de laboratoire. Le transfert de ces paramètres dans CERES-EGC a permis de simuler correctement les évolution des stocks de carbone dans les sols, les rendements et les prélèvements de N par les cultures ainsi que l'évolution des stocks de N minéral dans les sols.

Les dynamiques de minéralisation de C et N des 18 PRO disponibles sur le territoire ont été déterminées. NCSOIL a été paramétré à partir des caractéristiques analytiques et des résultats de fractions biochimiques des PRO. Les types de sol du territoire n'ont pas impacté significativement le paramétrage. Les PRO ont été classés selon 4 types : (1) composts stables, (2) composts moins matures restant plus réactifs et les fumiers stables, (3) des fumiers très réactifs correspondant à des fumiers de chevaux, (4) des PRO très réactifs tels que des boues et des fientes pouvant plutôt être utilisés comme fertilisants.

De nombreux scenarios d'apport de PRO, contraints par les flux de phosphore et de N apportés, ont été simulés sur 20 ans dans tous les contextes de sols, successions culturales et teneurs en C organique des sols du territoire. Le type de sol a impacté le plus le stockage de C et les lixiviations de nitrate tandis que les successions culturales étaient prépondérantes sur la substitution des engrais. Des composts ont permis de stocker 1.1 t C ha^{-1} an^{-1} allant jusqu'63% du C apporté. Des substitutions de 74 kg N ha^{-1} an^{-1} ont été atteintes avec une boue mais aussi un compost en prenant en compte la substitution des engrais liée aux augmentations de matière organique dans les sols. La substitution des engrais a pu atteindre plus de 90% du N apporté par les PRO, ceci étant lié aux arrières-effets sur le stockage de matière organique dans les sols et dépendant des hypothèses de calcul de la substitution des engrais azotés.

Un modèle d'optimisation a été développé pour sélectionner les scenarios d'apports de PRO pour chaque ilot du territoire (sol x succession de culture x teneur en C organique initiale x surface) tenant compte de la disponibilité des PRO et avec pour objectif de maximiser le stockage de C ou les substitutions en engrais azotés ou de minimiser la lixiviation de nitrate au niveau du territoire. En apportant préférentiellement les PRO les plus stables sur les sols à fort potentiel de stockage de C (fortes teneurs en argile et calcaire), jusqu'à 0.47 t C ha^{-1} an^{-1} pourrait être stocké. En appliquant préférentiellement les PRO fertilisants sur les successions à maïs, et les PRO amendants sur les successions à colza jusqu'à 53 kg N ha^{-1} an^{-1} de N de synthèse pourrait être économisés.