



Insertion of energy cover crops in cropping systems in France : multi-scale assessment of potential production and water-nitrogen-carbon impacts.

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Insertion of energy cover crops in cropping systems in France: multi-scale assessment of potential production and water–nitrogen–carbon impacts

Insertion de cultures intermédiaires énergétiques dans les systèmes de cultures en France : évaluation multi-échelles du potentiel de production et des impacts eau–azote–carbone

Thèse de doctorat de l'université Paris-Saclay

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Thèse soutenue à Toulouse, le 03 juillet 2023, par

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Titre : Insertion de cultures intermédiaires énergétiques dans les systèmes de cultures en France : évaluation multi-échelles du potentiel de production et des impacts eau–azote–carbone.

Mots clés : Biogaz ; Culture énergétique ; Stock de carbone organique du sol ; Recyclage des nutriments ; Modélisation

Résumé : Les cultures intermédiaires, implantées entre deux cultures principales, pourraient fournir une quantité considérable de biomasse pour la méthanisation, en particulier dans les exploitations sans élevage. Jusqu'à présent, elles n'étaient ni fertilisées ni récoltées et leur rôle était de fournir des services écosystémiques (cultures intermédiaires multi-services, CIMS). Avec le développement de la méthanisation, leur conduite change pour produire plus de biomasse (i.e. fertilisation, irrigation, récolte). On parle alors de cultures intermédiaires à vocation énergétique (CIVEs). Il existe un double questionnement concernant les CIVEs : (i) sur la quantité de biomasse et donc d'énergie qu'elles pourraient produire à l'échelle nationale, (ii) sur les impacts associés à leur développement et au retour au sol de digestats.

Pour y répondre, nous avons mis en place une démarche couplant expérimentation au champ et modélisation puis déployé une chaîne de modélisation à large échelle sur le long terme (couplage STICS, ALFAM2, SYS-Metha). Les CIVEs peuvent être un gisement important d'énergie en France : de 17 à 115 TWh. Elles sont bien un levier d'atténuation du changement climatique, malgré leur fertilisation et l'augmentation des émissions de N_2O associée, grâce à la substitution de gaz fossile et au stockage de C dans les sols. Cependant, il est probable que la production de biomasse et la fertilisation supplémentaires se fassent au détriment de la production alimentaire, de la qualité de l'air et de la disponibilité en eau. Elles réduisent tout de même la pollution au nitrate mais dans une moindre mesure que certaines espèces de CIMS.

Title : Insertion of energy cover crops in cropping systems in France: multi-scale assesement of potential production and water–nitrogen–carbon impacts.

Keywords : Biogas; Energy crop; Soil organic carbon stock; Nutrient cycling; Modeling

Abstract : Cover crops, grown between two main crops, could provide a considerable amount of biomass for anaerobic digestion (AD), especially on farms without livestock. Until now, they were neither fertilized nor harvested; their function was to provide ecosystem services (multiservice cover crops, MSCC). With the development of AD, their management changes to produce more biomass (i.e. fertilization, irrigation, harvesting). These crops are referred to as energy cover crops (ECC). There are two questions concerning ECC: (i) on the quantity of biomass thus energy they could produce on a national scale, (ii) on the impacts associated with their development and the return of digestates to the soil.

To answer these questions, we have set up an approach combining field experiment and modeling, then deployed a large-scale modeling chain over the long term (coupling STICS, ALFAM2, SYS-Metha). ECCs can be a major source of energy in France: from 17 to 115 TWh. They are indeed a lever for climate change mitigation, despite their fertilization and the associated increase in N_2O emissions, due to the substitution of fossil gas and the storage of C in the soil. However, the additional biomass production and fertilization are likely to be at the expense of food production, air quality and water availability. They still reduce nitrate pollution but to a lesser extent than some MSCC species.

Financement



Accueil



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Avant-propos

« Cette posture d'expert engagé au service d'une cause ne peut dispenser les chercheurs d'adopter clairement une posture de partie prenante défendant une cause (la santé des sols) et des valeurs plutôt qu'une posture « d'expert au-dessus de la mêlée » afin de favoriser une gestion démocratique plutôt que technocratique des problèmes. »

Recommandation du comité consultatif commun d'éthique INRAE-Cirad-Ifremer-IRD dans son avis de juillet 2022 pour les domaines de recherche liés aux pratiques agricoles et à la qualité des sols

Dans l'esprit de cette recommandation du comité d'éthique du groupement INRAE-Cirad-Ifremer-IRD, j'ai adopté tout au long de mon travail de thèse, une posture de recherche défendant certaines causes écologiques et sociétales. Cette posture m'a influencée dans mes choix méthodologiques, mes hypothèses de travail, ma façon d'interpréter les résultats, ma discussion. Pour donner un exemple concret, je ne discuterai pas de la concurrence d'usage des terres entre les CIVEs et les cultures dérobées à destination de l'alimentation animale parce que je considère, pour des raisons éthiques, que nous devrions considérablement réduire (voire cesser) l'élevage. Pour donner un autre exemple, je considère que le changement climatique est un problème majeur pour l'humanité. Pour faire face à ce problème, je pense que nous devons avant tout diminuer drastiquement notre consommation d'énergie et adapter en conséquence nos sociétés et nos modes de vie. Dans un second temps, je pense qu'il est nécessaire de remplacer l'usage des énergies fossiles par des énergies bas carbone. Cependant, je considère également que nos émissions de gaz à effet de serre sont loin d'être la seule menace. Aucune énergie n'étant « propre », il faut arbitrer entre les différents impacts environnementaux de chaque type de production d'énergie. D'où le choix de ce sujet de thèse qui, je l'espère, permettra un jour d'alimenter un débat éclairé entre citoyen.ne.s.

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List of publications and oral communications

- Launay C, Houot S, Frédéric S, Girault R, Levavasseur F, Marsac S, Constantin J (2022) Incorporating energy cover crops for biogas production into agricultural systems: benefits and environmental impacts. A review. *Agron Sustain Dev* 42:57. <https://doi.org/10.1007/S13593-022-00790-8>
- Launay C, Hermet L, Frédéric S, Houot S, Constantin J. Growing energy cover crops in double cropping: which ecosystem services or environmental impacts associated? XVII. Congress of the European Society for Agronomy, Aug 2022, Potsdam, Germany. (hal-03937907)
- Launay C, Hermet L, Houot S, Frédéric S, Constantin J. Quels services écosystémiques rendus par les CIVEs ? Résultats d'expérimentations et de simulations. JRI Biogaz et Méthanisation, Mar 2022, Lyon, France. (hal-03937948)
- Launay C, Crépeau M, Girault R, Levavasseur F, Houot S. New crop fertilization strategies after introduction of anaerobic digesters in a territory and their consequences on carbon and nitrogen dynamics in soils: case study of the Versailles plain. International Crop Modelling Symposium (iCROP2020) - STICS 2020 conference side-event, Feb 2020, Montpellier, France. 34 p. (hal-02951967)
- Launay C, Crépeau M, Girault R, Levavasseur F, Houot S. Projet MéthaPolSol, impacts de l'introduction de méthaniseurs dans un territoire sur les stratégies de fertilisation des cultures et leurs conséquences sur les dynamiques du carbone et de l'azote dans les sols : cas de la plaine de Versailles. Journées Recherche et Industrie biogaz méthanisation - JRI 2020, Association Technique Energie Environnement (ATEE), Sep 2020, Toulouse, France. (hal-02951996)

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List of abbreviations

AD: anaerobic digestion

C: carbon

CC: catch crop

DM: dry matter

ECC: energy cover crop

EOM: exogenous organic matter

FM: fresh matter

GHG: greenhouse gases

HHV: higher heat value

LHV: lower heat value

MSCC: multi-services cover crop

N: nitrogen

RD: raw digestate

SLD: solid and liquid digestate

SMNC: soil mineral nitrogen content

SOC: soil organic carbon

SOM: soil organic matter

SON: soil organic nitrogen

SWC: soil water content

General introduction

1 Issue and research strategy

This thesis is at the crossroads of two major societal issues: the energy transition and the ecological transition. The involvement of GRDF in this thesis shows the interest of the French energy sector for the potential of biogas production from energy cover crops (EECs) grown in annual cropping systems during fallow period, in particular in areas without livestock. Agricultural anaerobic digestion (AD) is the degradation of organic matter by microorganisms under controlled conditions and in the absence of oxygen. This degradation produces gas, a mix of CO_2 and CH_4 , which can be purified before being injected into the gas network. The remaining organic matter is called digestate and is spread on agricultural soils as organic fertilizer. AD experienced a revival in France in the early 2000s. Breeders were first interested in this process to treat their livestock effluents in co-digestion with organic matter from food industries or municipalities (e.g. green waste) (Béline et al. 2012). However, livestock effluents are not very methanogenic. Energy crops and energy cover crops produce more biogas, which is why they have started to be used as co-substrate (Béline et al. 2012). They are also the main resource available to field crop farms. In order to achieve a phase-out of fossil gas by 2050, the AD of ECCs emerges as central in the prospective scenarios. The first prospective study by the French Agency for the Environment and Energy Control (ADEME) in 2018 predicted that one third of the potential renewable gas production would be provided by AD (140 TWh higher heat value (HHV)) and that one third of this gas produced by AD would be supplied by ECCs (51 TWh_{HHV}; ADEME 2018). The actors of the French gas network estimated that out of these 140 TWh_{HHV}, 130 would be injectable in the network; the rest would be used in co-generation to produce electricity and heat (GRDF et al. 2022). In a second study, ADEME (2021) estimated that AD would provide 35 to 63% of the gas supply (95-128 TWh_{HHV}) and ECCs in particular would provide 50% of the gas produced from anaerobic digestion (51-61 TWh_{HHV}). Besides that, the number of press articles published in recent years against AD and ECCs as well as the growing interest of environmental associations for this technology (WWF France 2020, 2022) reflects the environmental concerns of a part of the citizens. This conflict reminds us that to get through the current climate and ecological crisis, it is not enough to replace fossil fuels with renewable energies. Criteria other than avoided CO_2 emissions must be taken into

account, such as emissions of other greenhouse gases (N_2O and CH_4), impacts on biodiversity, air and water pollution, crop and food production, etc. Faced with this complex problem, the objective of this thesis is to assess whether and to what extent energy cover crops can meet both environmental and energy challenges without compromising food production.

The objectives of the thesis are i) to consolidate scientific knowledge on the impacts of cropping systems integrating energy cover crops and digestate fertilization on the water, nitrogen and carbon cycles, ii) to analyze the drawbacks of increased production of cover crops related to potential competition with food production or the increased use of inputs (i.e. nitrogen fertilizers and irrigation), iii) to propose a new estimation of ECC potential biomass production and associated energy production using a high resolution approach and taking into account the diversity of climate, soil and cropping systems in France. These objectives are broken down into several more specific research questions that will be investigated at three different scales:

- Do cropping systems with ECCs and digestate inputs mitigate climate change? (i.e. do they store carbon in soils, reduce direct and indirect N_2O and induced emissions, substitute enough fossil gas to offset their emissions?)
- Does the introduction of fertilized ECCs in a cropping system increase nitrogen inputs and nitrogen losses (i.e. nitrate leaching, N_2O emissions, NH_3 volatilization)?
- What is the water consumption of ECCs? What is its impact on the water cycle?
- How much do ECCs increase agricultural biomass production? Is it at the expense of biomass for food? Could they produce biomass everywhere in France ?
- Does considering both the fluxes at the plot scale and for the biogas plant modify the conclusions on the previous questions?

Despite the interest and need for a more complete evaluation of these cropping systems with ECCs, we will not address issues related to the impact on soil structure, on meso- and macro-fauna, on weed management, on other nutrients cycling, and on the use of agrochemicals within the rotation. We will focus on nitrogen, carbon and water impacts at short and long term over a large range of agropedoclimatic situations.

A review of the literature about the knowledge and gap of knowledge on ECCs and their possible impacts is needed as a first step to refine the research questions and to make work hypothesis for the next chapters of the thesis. The state of the art on the benefits and environmental impacts of incorporating ECCs for biogas production into agricultural systems is presented in **Chapter 1**. Without going into detail now, it shows that few studies investigated this topic but seems to indicate that ECCs could meet their environmental goals, mainly based on one hand on the known impacts of non-exported cover crops and on the other hand on digestate mainly from livestock origin. However, given the few number of studies on the combined effects of exported cover crops and their return as digestate, there are still large uncertainties on their impacts in various pedoclimatic conditions and cropping systems that the thesis will attempt to fill.

After the Chapter 1, the thesis is divided into four chapters corresponding to four different spatial scales at which the impacts were studied: Chapter 2 begins at the plot scale, Chapter 3 goes down to the laboratory scale, Chapter 4 moves up to the scale of a small region and Chapter 5 to the scale of the whole France. The methods used and the objectives/scientific questions are therefore complementary in each chapter (Figure 1).

Chapter 2 presents a two-year field experiment in which ECCs are compared to other fallow managements: bare soil and non exported cover crops. The experiment is coupled with the STICS crop model (Brisson et al. 2009). This experimental set-up meets different objectives:

- Determine the impact of ECCs and digestate return on the water and nitrogen cycles at the field scale and over the short term.
- Analyze the impact of ECCs on the following crop yield and stress.
- Compare these effects to those of non-exported cover crops terminated in autumn or in spring and to a bare soil during fallow period.
- Calibrate the soil crop model STICS on some ECCs species.

Chapter 3 is complementary to Chapter 2 as it is based on the same field experiment. Part of the biomass harvested in the experiment is digested in a pilot reactor and the C and N mineralization of the produced digestate and initial plant biomass is followed in laboratory

incubations. The main objective of this chapter is therefore complementary to those of the previous chapter:

- Determine the impact of the AD of ECC on the carbon cycle.

In **Chapter 4**, we use the STICS model to represent the cropping systems of a small agricultural region for 30 years. Coupled with an AD model SYS-Metha (Bareha et al. 2021b), it allows us to simulate different scenarios for the implementation of AD plants in the territory, including a scenario for the development of ECCs. With this study scale, we take into account the constraints on the cropping systems and practices of the farmers in the area, the available organic resources, the total volume of digestate produced and the areas available for land application of this digestate. The objectives of this chapter are quite different from the plot scale:

- Analyze the impacts of ECCs on N and C exchanges with a biogas plant.
- Compare the consequences of the development of different AD development scenarios on the N inputs and outputs from the region, and C storage in the region.

Finally, in **Chapter 5**, we set up a simulation chain reusing results or methodological points from previous chapters. We use it to simulate different scenarios of development of cover crops for AD or not at the scale of the whole France on grids of a few km² for 30 years. This scale allows fulfilling the following objectives:

- Quantify the potential biomass production of ECCs in France and the associated environmental impacts.
- Analyze the long-term impacts of ECCs insertion in a wide range of soil and climatic conditions and crop successions in cropping systems without livestock.

We will conclude in **Chapter 6** with a general discussion in which we will synthesize and bring coherence to the highlights of the results.

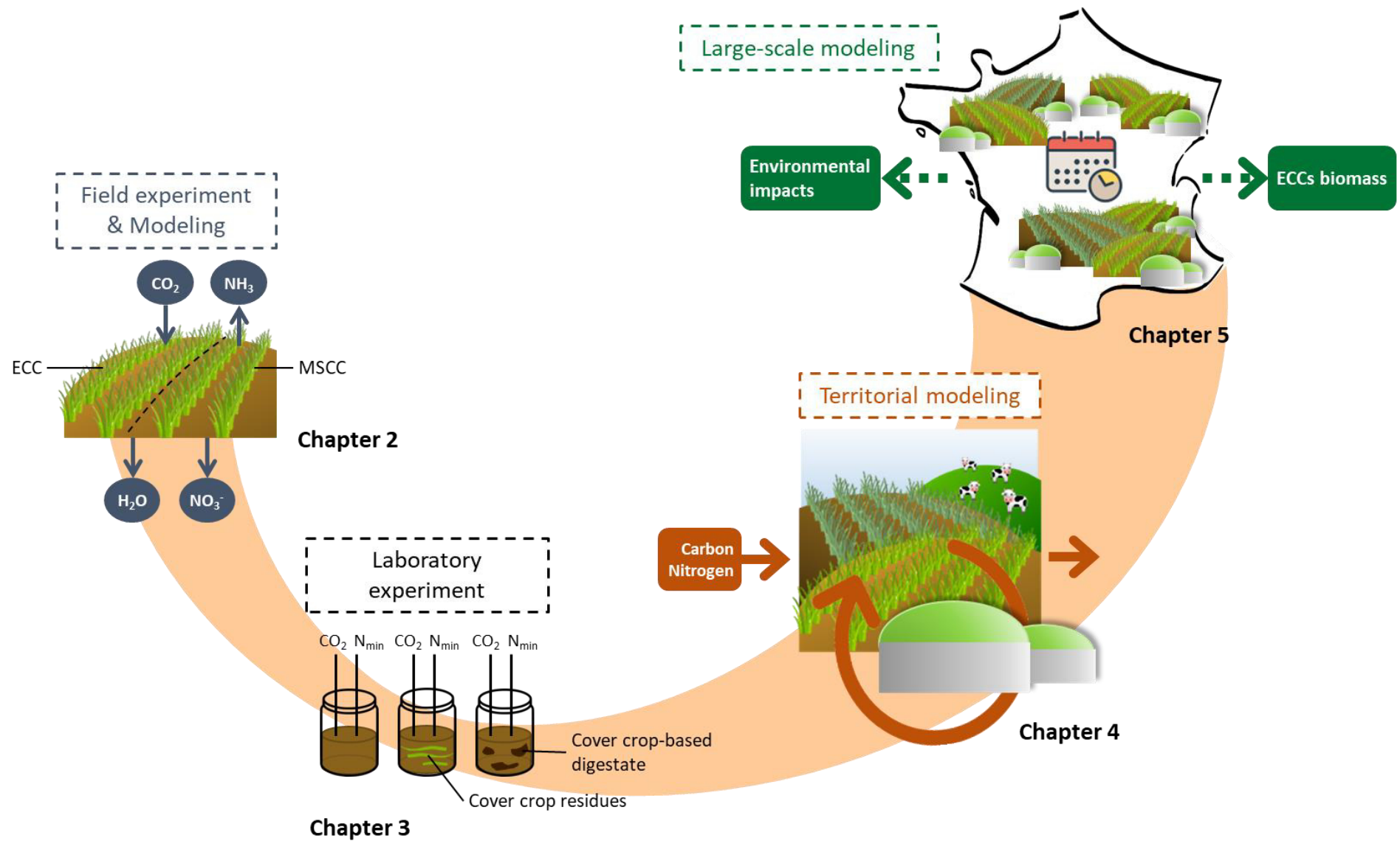


Figure 1. Schematic representation of the approach of the thesis with the division into chapters.

2 PhD environment

This PhD was funded by Gaz Réseau de France (GRDF) through the Cifre program of the National Association for Research and Technology (ANRT). GRDF is the main gas distribution system operator in France. Through its public service contract with the Government, GRDF is committed to support research and development on the ecological and energy transitions.

The PhD is fully integrated into the BioCIGES project (2020-2023) funded by ADEME, which has been connected to the PhD of Nicolas Malet (UMR ISPA), also partly on ECCs and defended on December 9, 2022. The objective of the project is to provide an environmental assessment (GHG balance) and economic evaluation of two ways to valorize the biomass of cover crops: i) direct incorporation into the soil to promote C storage or ii) valorization into biogas to promote the substitution of fossil fuels. My thesis deals with the task of deepening the knowledge on the impacts of ECCs with return of digestates to the soil on the N, C and water cycles, and on the task of estimating the potential source of cover crops biomass on a national scale. Nicolas Malet's work involved the estimation of other substrate sources for AD and the construction of a GHG balance sheet to compare the two ways of biomass valorization: with and without AD. The ECOPUB research unit of INRAE carries out the economic evaluation of these two valorization ways.

Since the beginning of the thesis, we have been in contact with the RECITAL project¹ (2019-2023) led by Arvalis, with which we have been able to share our experiences. Agricultural cooperatives, chambers of agriculture, associations promoting AD, and energy companies were associated to this project. The aim of the project was to provide regionalized technical-economic recommendations to optimize the overall profitability of crop systems with ECCs. Among other things, the participants examined the technical management of ECCs in different soil and climate contexts in order to control production hazards. To meet these objectives, the project relied on experiments carried out on Arvalis experimental fields and on farmer plots, as well as on workshops for the co-construction of typical situations and scenarios by region. Thanks to the link with this project, I benefited from the knowledge of technical experts in

¹ <https://aile.asso.fr/projet-rd/recital/>

ECCs, which was useful for me to have a vision of the reality and to build my experimentation and my scenarios of ECCs insertion.

A part of the work presented in this thesis is a continuation of the MéthaPolSol project funded by ADEME and completed in 2019. The objective of the MéthaPolSol project was to study the impact of certain practices associated with AD on nitrogen and carbon flows within the system combining the digester and the cropping systems. This work was carried out in two different pedoclimatic and agricultural contexts: i) the Coglais, a livestock farming area located in Ille-et-Vilaine and ii) the Versailles plain, a cereal farming area located in the Yvelines. At the end of the project, we determined that the case study of the Versailles plain deserved to be further explored. The research question being close to the research question of the PhD, this was the opportunity to introduce this intermediate scale in the PhD. Besides, the methodology used in this study was reused and adapted in Chapter 5 for the simulation of scenarios on a France-wide scale.

Chapter 1 – Incorporating energy cover crops for biogas production into agricultural systems: benefits and environmental impacts. A review.

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Abstract: Some European countries are exploring the idea of replacing dedicated crops with energy cover crops for biogas production. Indeed, energy cover crops can generate consequential biomass without competing with food crops for land use. However, the potential benefits and impacts of this choice are not fully understood. Here, we review what is known about the consequences of energy cover crop usage by examining management regimes and digestate use, including impacts on the environment and cropping system performance. First, compared to cover crops, energy cover crops are intensively managed to produce more biomass (< 5 t DM.ha⁻¹ vs. up to 16 t DM.ha⁻¹). Second, nitrogen is conserved during anaerobic digestion and is more readily available to crops in digestate than in cover crops residues. However, ammonia is lost via volatilization, which could reduce nitrogen use efficiency, depending on the storage conditions and application method. Third, 43–80% of the crops' initial carbon is transformed into biogas. That said, levels of soil carbon storage may nonetheless resemble those obtained with cover crops left behind because carbon is stabilized during anaerobic digestion and the energy cover crops' roots and stubble are left behind in the soil. Fourth, energy cover crops can act as multiservice cover crops, reducing nitrate leaching, improving soil microbial activity, and enhancing soil physical properties during the fallow period. Fifth, energy cover crop usage can have certain disservices, such as soil compaction, the need for additional inputs (e.g., irrigation, fertilization, pesticides), reduced groundwater recharge, and reduced following crop yield. In summary, expanding the usage of energy cover crops for biogas production does not seem to be an environmental threat. However, care must be taken to avoid the intensification of irrigation and lengthening growing periods to boost biomass, which could reduce food production.

Keywords: energy cover crop, catch crop, ecosystem services, digestate, carbon storage, leaching, volatilization, drainage, microbial activity

1.1 Introduction

After rapidly developing in the late 2000s, anaerobic digestion-based biogas production in Europe has stagnated since 2017 because of new measures and regulations in the key countries (e.g., Germany, the United Kingdom), which have reduced economic incentives and limited the use of cash crops as source materials (EurObserv'ER 2020). Cash crops can be included in the digester ration, as "dedicated crops" or "energy crops", given that they do not surpass a certain threshold, which varies among countries (e.g., 2021: 15% in France vs. 44% in Germany; Thrän et al. 2020). The use of food crops to produce energy (e.g., biogas, biofuels) is a subject of debate because land use competition between food and energy crops must be avoided. Consequently, Europe has introduced sustainability criteria to apply when producing biomass for energy. Described in the European Union Renewable Energy Directive 2018/2001, these criteria specify that energy crops should not be grown in areas with high biodiversity nor in soils containing high levels of stored carbon. Moreover, the use of biofuels and biogas should prevent a certain proportion of greenhouse gas emissions defined by the directive. Biogas production will need to meet the above criteria to be categorized as a renewable energy source (EurObserv'ER 2020). The sustainability of anaerobic digestion is also a key concern (WWF France 2020). It is recommended that (i) all stakeholders be mobilized when a new project is launched (ii) agroecological practices be implemented at the farm level; and (iii) further steps be taken to sustainably manage biomass, promote a positive carbon balance, reduce greenhouse gas emissions, and preserve biodiversity (WWF France 2020).

France is the European country in which primary biogas production has shown the most growth (+11% between 2018 and 2019) (EurObserv'ER 2020). Initially employed to treat livestock farming and food processing waste, anaerobic digestion is now being promoted to produce 6 to 8% of the gas consumption by 2028 (Ministère de la transition écologique et solidaire 2020). To meet high demands for biomass, France is employing cover crops, particularly in grain-growing areas without appreciable livestock farming. According to a futures study, 30% of the country's gas needs should be met by anaerobic digestion by 2050. Energy cover crops could provide one-third of the necessary biomass and serve as the main source of agricultural biomass (ADEME 2018b). Energy cover crops are seeded and harvested between two cash crops (i.e., in systems using double cropping or growing three crops in two years). By definition, energy cover crops are not cash crops and do not compete with food crops because they

develop over a period that is usually too short to grow food crops. However, this period is already being increasingly used to produce supplementary fodder (Binder et al. 2020; Andersen et al. 2020) or even food crops due to climate change (Meza et al. 2008; Sandler et al. 2015). Cover crops without immediate monetary return are already in use because they provide many ecosystem services; their deployment is also mandatory in zones susceptible to nitrate leaching, where they limit groundwater pollution during the long fallow period (European Union Nitrate Directive 91/676/EEC). In addition to reducing nitrate leaching (Constantin et al. 2010), cover crops also protect soils from erosion (Blanco-Canqui et al. 2015), provide nitrogen to the subsequent crop (Tonitto et al. 2006), increase carbon storage in soils (Poeplau and Don 2015), provide habitat and resources for wildlife and microorganisms (Ellis and Barbercheck 2015; Finney et al. 2017; Wilcoxon et al. 2018; Carmona et al. 2021), and, under certain conditions, can limit diseases and weeds (Blanco-Canqui et al. 2015). For this reason, they are also referred to as multiservice cover crops (Couëdel et al. 2019).

Countries other than France are also interested in exploiting energy cover crops, such as the Biogasdoneright™ initiative in Italy (Dale et al. 2016) and the Syn-Energy research project in Austria (Szerencsits 2014). However, there are some key concerns. Certain levels of biomass production are required for the process to be economically sustainable; energy cover crops need more intensive management than do conventional cover crops; and the use of inputs such as water or fertilizer is generally recommended (Marsac et al. 2019). Few studies have looked at the environmental impacts of energy cover crops, and questions remain with regards to the sustainability of anaerobic digestion, if we apply the criteria of the European Union or WWF France. For example, more information must be gathered about the relationship with land use, greenhouse gas emissions, soil organic matter storage, and biodiversity. Stated more succinctly, would energy cover crops retain their status as multiservice cover crops if utilized to produce biogas? We sought to answer this question by reviewing what is known about energy cover crops and, more specifically, by attempting to formulate predictions based on current knowledge about cover crops and digestate-based fertilizers. After providing a detailed description of energy cover crops and how they are affected by anaerobic digestion, we reviewed the effects of growing energy cover crops and using their digestates as fertilizers, examining how both interact to influence different fluxes and processes. Finally, we examined the potential impacts of energy cover crop use on cropping systems (Figure 2).



Figure 2. Energy cover crop experiment in Auzeville-Tolosane, France: (a) digestate obtained from energy cover crops is spread before the summer energy cover crop, sorghum, is seeded (in July) and (b) the sorghum prior to harvest (in September). Photographs by the authors.

1.2 Material and Methods

A literature review was carried out using the Web of Science and Google Scholar databases (accessed between January 2020 and September 2021) complemented with scientific reports that the authors were aware of. To find publications in the scope of our study, we combined different groups of keywords: "ecosystem services", "sustainability", "soil quality", "ecological footprint", "environmental impact", "environmental assessment" for environmental impacts and benefits; "water balance", "drainage", "soil water content", "water deficit", "water stress" for water-related impacts; "nitrogen balance", "nutrient cycling", "nitrous oxide emissions", "nutrient limitation", "volatilization", "leaching" for nitrogen-related impacts; "carbon balance", "carbon storage", "carbon sequestration", "soil organic carbon" for organic matter-related impacts; "energy crop", "green manure", "catch crop", "cover crop", "double crop*" for cover crops; "fertilization" or "digestate" for the use of digestate; "anaerobic digestion", "biogas

production” to include anaerobic digestion. To complete and broaden the search, we also checked the references in the collected papers. We started the investigation with reviews and meta-analysis of the benefits and drawbacks of cover crops on the one hand and digestates (all origins) on the other. We then restricted the reading as much as possible to papers dealing with cover crops used for biogas production or digestate derived, at least in part, from plant biomass.

1.3 Differences between multiservice cover crops and energy cover crops

1.3.1 Multiservice cover crops

As noted in several literature reviews, cover crops furnish numerous additional environmental benefits, which has given rise to the term multiservice cover crops. Part of the benefits is provided through soil cover and part is provided through the return of residues to the soil. Cover crops improve water quality in several ways. They reduce drainage, thus limiting pesticide contamination of groundwater (Giuliano et al. 2021). They mobilize nitrates before the drainage period, helping to preserve groundwater quality by preventing nitrate leaching (Constantin et al. 2010; Tribouillois et al. 2016). Cover crops serve the same function for sulfates (Couëdel et al. 2018b). Furthermore, the roots of cover crops increase soil porosity, both promoting water infiltration and reducing surface runoff (Blanco-Canqui et al. 2015). As a result of the latter, dissolved phosphorus is retained in the soil for the following crop, and water pollution is limited (Daryanto et al. 2018). Cover crops act as physical barriers against water and wind erosion (Blanco-Canqui et al. 2015). They can also help directly mitigate climate change because, in temperate regions, they have higher albedo than do soils; in Europe, the mitigation potential is 3.16 Mt CO₂-eq per year (Carrer et al. 2018). Cover crops suppress weeds by competing for space and resources (i.e., light, water, nutrients) (Schipanski et al. 2014; Blanco-Canqui et al. 2015; Smith et al. 2020). Finally, cover crops provide habitat and food resources for birds, insects, and microorganisms, thus promoting biodiversity (Blanco-Canqui et al. 2015). Upon destruction, cover crops residues continue to provide other services. Residue mineralization supplies nitrogen and sulfur to the following crop (Thorup-Kristensen et al. 2003; Tribouillois et al. 2016; Couëdel et al. 2018a). Leguminous cover crops fix atmospheric dinitrogen, boosting the supply available for the next crop, even if they have a lower “catch

crop” effect on nitrates (Tonitto et al. 2006). When cover crop biomass (roots and shoots) is incorporated into the soil, carbon storage levels can reach 320 kg C/ha per year (Poeplau and Don 2015) or even 560 kg C/ha per year (Jian et al. 2020), further contributing to climate change mitigation. In fact, expanding cover crop use in France could reduce greenhouse gas emissions by 515 kg CO₂-eq/ha per year (Launay et al. 2021) despite a slight increase in nitrous oxide emissions also demonstrated in other studies after the incorporation of cover crops (Blanco-Canqui et al. 2015; Guenet et al. 2020; Abalos et al. 2022). If cover crop roots and shoots are left in place, they can also control pathogens and weeds via allelopathy (Snapp et al. 2005; Matthiessen and Kirkegaard 2006; Blanco-Canqui et al. 2015).

Cover crops can also have deleterious effects (i.e., disservices). For example, they may engage in pre-emptive competition with cash crops for nutrients and water. Their water and nitrogen consumption, as well as cases of nitrogen immobilization after residue incorporation can lead to yield losses (Krueger et al. 2011; Alonso-Ayuso et al. 2014; Alvarez et al. 2017; Meyer 2020). Such effects can be limited by destroying the cover crops early or increasing soil nitrogen mineralization over time by accumulating organic matter (Constantin et al. 2011; Krueger et al. 2011; Alonso-Ayuso et al. 2014; Acharya et al. 2017). Cover crops can also host pathogens, allowing their populations to persist and multiply during the fallow period, which can put subsequent crops at greater risk of diseases (i.e., the “green bridge” phenomenon). Such dynamics have led to yield losses in several cash crop-cover crop (Acharya et al. 2017). To eliminate potential green bridge effects, care should be taken to select a cover crop species that is non-host for the next crop. Finally, cover crop water consumption in the winter can also reduce drainage and, consequently, groundwater recharge and the water supply available for other uses (Meyer et al. 2019).

Depending on the specific combination of services and disservices, cover crops can have positive or negative impacts on the following crop’s yield. For example, green manure effects, pathogen control, and weed control often boost yield (Matthiessen and Kirkegaard 2006; Tonitto et al. 2006; Schipanski et al. 2014; Bergtold et al. 2017). In general, yield tends to be positively influenced, particularly over the long term (Constantin et al. 2011; Blanco-Canqui et al. 2015), and becomes less vulnerable to climatic hazards (Snapp et al. 2005; Bergtold et al. 2017).

The selection and management of cover crop species must take into consideration local climatic conditions, soils, cropping systems, and desired services. For example, if the objective is to prevent nitrate leaching, a fast-growing, non-leguminous species should be chosen, and the cover crop should be seeded early, so it can take up as much nitrogen as possible before the drainage period. To increase soil organic matter content, it is better to select a species with a long establishment period that is capable of producing large quantities of biomass. Sometimes trade-offs are observed. For example, there may be a trade-off in water quality and the supply of blue water under specific soil-climate conditions where certain agricultural practices are used (Obiang Ndong et al. 2021). In such contexts, species mixtures could be useful (Tosti et al. 2014; Tribouillois et al. 2016; White et al. 2017; Couédel et al. 2019). Finally, incorporating cover crops into the soil early on (i.e., more than 15 days before the next crop is seeded) generally limits the risk of deleterious effects (Justes et al. 2012; Acharya et al. 2017).

1.3.2 Energy cover crops

In France, the definition of an energy cover crop is provided in Decree n°2016-929 for the Application of Article L.541-39 of the Environmental Code: it is a crop grown between two cash crops, and its biomass is harvested and anaerobically digested to produce biogas. In western Europe, energy cover crops are planted either during the summer fallow period (June to October) or during the winter fallow period (September to May). They can function as multiservice cover crops.

The fact that energy cover crops are harvested for biogas production rather than being incorporated into the soil is what differentiates them from conventional multiservice cover crops. Indeed, the function of energy cover crops more closely resembles that of cover crops transformed into livestock feed, which are also known as “double crops.” Due to the short growing period, energy cover crops are harvested before they reach maturity. Because methanogenic potential varies little among crop species during vegetative stage, the amount of biomass harvested is the key factor determining levels of biogas production (Graß et al. 2013; Marsac et al. 2019). Aboveground biomass production is higher for species with high growth levels, such as sorghum [*Sorghum bicolor* (L.) Moench], corn [*Zea mays* L.], and sunflower [*Helianthus annuus* L.] in the summer and triticale [*x Triticosecale rimpai* Wittm.], rye [*Secale cereale* L.], barley [*Hordeum vulgare* L.], and oats [*Avena sativa* L.] in the winter. Inputs (e.g., fertilizer, irrigation water, pesticides) can be provided to further boost biomass

accumulation. During the summer fallow period, solar radiation levels are high, but seeding conditions can be challenging, particularly in the dry regions of southern Europe, which may face water scarcity during this time of year. Farmers should favor short-cycle species that are resistant to water stress, such as sunflower and sorghum, and recognize that irrigation must sometimes be employed to ensure cover crop emergence and biomass production. During the winter fallow period, biomass production mostly takes place in spring. Farmers should thus target species with explosive growth during the early spring, such as grasses. Because energy cover crops are good at accumulating biomass, they take up large amounts of nitrogen from the soil and require a moderate supply of fertilizer (40–80 kg N/ha), mainly in the form of digestate; using this approach can ensure sufficient yields without impairing the growth of the following crop (Szerencsits 2014; Marsac et al. 2019).

Much research has been dedicated to management strategies for maximizing biomass production by energy cover crops (Heggenstaller et al. 2008; Graß et al. 2013; Molinuevo-Salces et al. 2013, 2014; Negri et al. 2014; Szerencsits 2014; Igos et al. 2016; Marsac et al. 2019; Wannasek et al. 2019). Best practices for summer energy cover crops include early seeding, the use of drought-resistant species, and a sufficient supply of water at seedling emergence (Marsac et al. 2019). In the case of winter energy cover crops, biomass is largely harvested from April onwards, so a good approach is to delay harvesting as much as possible, without overly shortening the subsequent crop's growing period (Szerencsits 2014; Marsac et al. 2019). To produce sufficient energy cover crop biomass, attention must be paid to seeding and harvesting dates, which could require farmers to redesign crop rotations. It may be enough to employ early varieties of cash crops and harvest them a few days or weeks in advance, in the case of summer energy cover crops; for winter energy cover crops, cash crops can be seeded with a slight delay. To increase the total biomass production of energy cover crops, the crop cycle can be modified either by removing winter cash crops—allowing the addition of winter energy cover crops—or by introducing winter cash crops that can be harvested before July—allowing the addition of summer energy cover crops.

With regards to environmental impacts, energy cover crops should display the same services and disservices as conventional cover crops during the soil cover period. These include reducing drainage; protecting groundwater quality; structuring the soil; mitigating climate change thanks to higher albedo levels and enhanced carbon storage in soils (via belowground

biomass); maintaining biodiversity; and controlling weeds. Questions remain with respect to service intensity. Because energy cover crops are harvested, they do not supply any benefits associated with aboveground biomass incorporation into the soil (i.e., allelopathy, green manure effects, enhanced carbon storage via aboveground biomass). It is also unclear whether the use of energy cover crops could negatively affect food production because of (i) the resulting changes to crop rotations and (ii) preemptive competition for water and nitrogen between crop types.

1.4 From cover crop biomass production to digestate storage

1.4.1 Cover crop biomass and nutrient absorption

Cover crop biomass production can vary markedly depending on species, management regime, soil characteristics, and climatic conditions. In the summer, it is hard to obtain dense and homogenous cover if plants are seeded in dried-out soil and water is scarce, given that cover crops are rarely irrigated. In the fall, the important limiting factors are the total number of growing degree days and levels of global radiation. Furthermore, conventional cover crops are not usually fertilized and are destroyed early on, by either frost or the farmer. In Europe, aboveground biomass rarely exceeds 5 tons of dry matter (DM) per hectare (ARVALIS - Institut du végétal et al. 2011; Justes et al. 2012; Hansen et al. 2021). In general, energy cover crop biomass is higher because of the longer growing season, the species chosen and the use of inputs (fertilizer, irrigation water, pesticides). That said, production remains highly variable for summer and winter energy cover crops (3–15 t DM/ha and 2–16 t DM/ha, respectively), depending on species, variety, pedoclimatic conditions, and management regime (Szerencsits 2014; Marsac et al. 2019). Initial results have shown that cereal-legume mixtures where the proportion of legumes does not exceed 40% did not impact the cereal yield (Marsac et al. 2019). Nitrogen levels in the aboveground biomass of cover crops differ based on species: they range from 13.6 to 52 g N/kg of dry matter for brassicas and grasses (Justes et al. 2009; Bareha et al. 2018; Hansen et al. 2021). They are higher for legumes: between 43 and 84 g N/kg of dry matter (Bareha et al. 2018; Hansen et al. 2021). Across trials conducted in France and Denmark, the total nitrogen absorbed by unfertilized cover crops ranged from 10 to 171 kg N/ha for legumes and 9 to 89 kg N/ha for non-legumes (ARVALIS - Institut du végétal et al. 2011; Hansen et al. 2021). Other studies conducted in temperate regions have found that nitrogen uptake by

legumes can exceed 300 kg N/ha since the plants are not limited by levels of soil nitrogen (Thorup-Kristensen et al. 2003; Tonitto et al. 2006). Brassicas also have a high nitrogen uptake capacity, up to 300 kg N/ha (Constantin et al. 2015); furthermore, their nitrogen acquisition rates per growing degree day are higher than those of legumes when nitrogen is not limiting (Tribouillois et al. 2015). The C:N ratios of cover crops generally vary from 9 to 40 (Justes et al. 2012; Hansen et al. 2021). They are lower for legumes and/or cover crops that have experienced a short growing period or conditions of high nitrogen availability (Justes et al. 2012). There were ranges of values for other nutrients: 2–8.2 g/kg of dry matter for phosphorus; 15–52.8 g/kg of dry matter for potassium; 0.9–4 g/kg of dry matter for magnesium; and 1–9 g/kg of dry matter for sulfur (MERC tool, French National Reference Database; Chambre Régionale d'Agriculture Nouvelle-Aquitaine 2020; Hansen et al. 2021). The carbon content of plant dry matter can be quite consistent (40–50%; Bertrand et al. 2019). Root:shoot ratios of carbon and nitrogen may differ greatly among cover crop species. Constantin et al. (2011) found that the root:shoot ratio for carbon was 20% for mustard [*Sinapis alba* L.] versus 72% for ryegrass [*Lolium multiflorum* Lam.]; for nitrogen, it was 6% for mustard versus 37% for ryegrass. Bispecific mixtures, especially when including legumes, tend to have intermediate values between those obtained by both species in sole crop. Tribouillois et al. (2016) and Couëdel et al. (2018b) observed nitrogen uptake values intermediate or equal to the best species for cereal-legume and crucifer-legume mixtures. The same observation was made by Couëdel et al. (2018a) for sulfur uptake. C:N and C:S ratios are always halfway between both species (Tribouillois et al. 2016; Couëdel et al. 2018b, a). A recent meta-analysis showed that it was very rare for a mixture to perform better than the best of the species alone on various criteria including biomass production and nitrogen uptake (Florence and McGuire 2020).

1.4.2 Ensiling energy cover crops

After being harvesting, fresh energy cover crop biomass is preserved as silage until it is anaerobically digested. Feedstock quality greatly affects the success of the storage process: the feedstock should have high dry matter content, high accessible carbohydrate content, and low buffering capacity (Teixeira Franco et al. 2016). Farmers should aim for dry matter content of 25–30% to limit fermentation, loss of matter and energy, and leachate formation (Teixeira Franco et al. 2016). However, this threshold is difficult to achieve when energy cover crops have a short growing period. The harvest date can be delayed to allow further declines in moisture

content, but increased lodging risks may result because of poor weather conditions in the spring and fall (for winter and summer energy cover crops, respectively) (Marsac et al. 2019). That said, it remains unclear whether dry matter content affects biochemical methane potential even if it affects fermentation during ensiling (Teixeira Franco et al. 2016). In addition, silage juices could be recovered to cofeed the digester. It is known that biochemical methane potential can be affected by air exposure within the silo. To avoid any potential losses, it is important to use good management practices, such as rapid silo closure and high levels of biomass compaction (Teixeira Franco et al. 2016).

1.4.3 Fate of carbon during anaerobic digestion

During anaerobic digestion, between 20 and 95% of the substrate's carbon content is transformed into biogas (CH_4 and CO_2), depending on substrate type (Möller and Müller 2012). The percentage is higher for plants that have undergone little to no transformation: 64% or 80% for a 100% corn substrate (Thomsen et al. 2013; Béghin-Tanneau et al. 2019) versus 46% for corn previously digested by animals (assuming that ruminal degradation of original carbon content is 70%) (Thomsen et al. 2013). For energy cover crops, the amount of carbon converted into biogas during anaerobic digestion represents 43–74% of initial carbon content, compared to 36–41% in the case of livestock manure (Bareha et al. 2018). These figures are maxima given that the degree of degradation depends on material residence time and substrate preparation (Bareha 2018). During anaerobic digestion, microorganisms preferentially degrade the labile fraction of the organic substrates—avoiding recalcitrant molecules (e.g., lignin)—and produce stabilized metabolites (Coban et al. 2015; Möller 2015). Past research has found differences in the degree of degradation of the different organic matter fractions in energy cover crops: it is between 17 and 30% for the most recalcitrant fraction (lignin); between 32 and 72% for the intermediate fraction (cellulose + hemicellulose); and between 10 and 75% for the soluble fraction (Bareha et al. 2019).

1.4.4 Fate of nutrients during anaerobic digestion

During anaerobic digestion, the nitrogen in crop residues is largely conserved in the digestate (Möller and Müller 2012), which contrasts with the fate of carbon. Moreover, the nitrogen can change form: depending on the proportion of mineral nitrogen in the substrate, further mineralization of organic nitrogen can occur (Bareha et al. 2018). In fresh crop residues with

N-NH₄⁺:total nitrogen ratios of around zero, an average of 57% of the organic nitrogen is mineralized during anaerobic digestion; this percentage is closer to 33% if residues are transformed beforehand (e.g., via ensiling or animal consumption; Bareha et al. 2018). Consequently, the digestate has a higher N-NH₄⁺ content than the substrate of origin, a difference that is further accentuated for crop residue digestate versus livestock manure digestate (Möller and Müller 2012). The percentage of N-NH₄⁺ varies greatly, from 4 to 82% of total nitrogen, depending on the substrate (Möller and Müller 2012; Nkoa 2014; Bareha et al. 2018; Guilayn 2018). Even for fresh cover crop residues, the final digestate displays marked variability: from 1.2 g N-NH₄⁺/kg of fresh matter for barley (31% of total nitrogen) to 6.0 g N-NH₄⁺/kg of fresh matter for vetch (71% of total nitrogen) (Bareha et al. 2018). This range of values results from differences in substrate organic nitrogen content and in organic nitrogen biodegradability. Other nutrients are similarly retained during anaerobic digestion. Levels of phosphorus range from 0.2 to 31.5 g/kg of dry matter, and levels of potassium range from 0.6 to 95 g/kg of dry matter (Möller and Müller 2012; Nkoa 2014; Guilayn 2018). Although phosphates, sulfates, and micronutrients (e.g., Fe, Mg, Ca) are mineralized, they are not necessarily more available to plants for several reasons: (i) they precipitate as phosphates, sulfide, carbonate, and hydroxides due to increases in pH; (ii) they experience sorption in the digestate's solid phase; and (iii) they undergo complexation with other compounds in solution (Möller and Müller 2012). Sulfur might be an issue for anaerobic digestion because at high doses it reduces the efficiency of the digestion and produces a corrosive gas, H₂S (Yang et al. 2016). The production of H₂S can be predicted by the C:S ratio of the substrates. Peu et al. (2012) found that above a C:S of 40, the amount of H₂S produced is treated efficiently. However, the C:S of cover crops varies between 63 and 319 for a large number of species (Peu et al. 2012; Couëdel et al. 2018a; Hansen et al. 2021), above the C:S of pig manure at 44-51. Consequently, the use of cover crops produces H₂S but in acceptable quantities to be treated efficiently by the equipment already in place on the installations.

1.4.5 Digestate storage

Fermentation continues during digestate storage, and part of the carbon in the digestate is transformed into methane and carbon dioxide. Approximately 8% of the carbon in the raw digestate is transformed, with figures of 15% for the liquid phases and 34% for the solid phases (Bareha et al. 2021). These emissions are significant since they represent 1.43 to 10.36% of

methane production for a given biogas unit (Liebetrau et al. 2010). If the digestate is not covered, this biogas is lost and contributes to greenhouse gas emissions (Balsari et al. 2013). Digestate degradation during the storage period also results in the release of nitrous oxide and ammonia, which can be limited by covering the digestate (Möller 2015; Holly et al. 2017). According to different studies, nitrogen loss ranges from 9% for uncovered raw cattle manure digestate to 6% for liquid digestate (Holly et al. 2017) and to 30% of nitrogen for uncovered raw pig slurry digestate (Sommer 1997). Based on these figures, a recent review by Walling and Vaneeckhaute (2020) found that emissions from livestock manure digestate under storage conditions ranged from less than 0.01 to 0.13 kg CO₂-eq/kg N per day. For the moment, we have not identified any figures from the digestion of cover crops.

1.5 How the anaerobic digestion of energy cover crops affects nitrogen fluxes

1.5.1 Nitrogen availability for crops

Upon cover crop destruction, soil levels of mineral nitrogen are often 50% lower than those associated with bare soil because cover crops absorb nitrogen, a phenomenon that is particularly pronounced in dry climates (Tribouillois et al. 2016a; Alvarez et al. 2017; Meyer 2020). The level of mineral nitrogen available to the following crop is strongly correlated with cover crop termination date and winter drainage intensity. Later destruction dates result in greater differences relative to what is seen on bare soil because the absorption period is longer and growth is faster in the spring. Lower levels of drainage have the same effect because mineral nitrogen remains in the bare soil while it is absorbed by the cover crop. When levels of soil organic matter and cover crop mineralization are insufficient in the early spring, the low quantities of mineral nitrogen may induce preemptive competition, resulting in nitrogen stress when the next crop begins growing (Thorup-Kristensen et al. 2003; Marcillo and Miguez 2017). However, cover crops may furnish nitrogen to the subsequent crop via green manure effects. Green manure effects result because (i) the growing cover crop takes up mineral nitrogen that would otherwise leach away during winter and (ii) mineralization releases this nitrogen after the cover crops are incorporated into the soil, making it available to the next crop. The degree of these effects depends on the cover crop's nitrogen uptake efficiency; residue mineralization dynamics; and leaching risks during the drainage period. When legumes are used as cover

crops (by themselves or in combination with non-legumes), the fixation of atmospheric nitrogen can boost green manure effects. Furthermore, the C:N ratio can shape the level and timing of the nitrogen available for the following crop (Jensen et al. 2005; Justes et al. 2009). Residues with a ratio of less than 13 resulted in immediate net positive nitrogen mineralization; in contrast, residues with a ratio of more than 26 resulted in net nitrogen immobilization over the five-month incubation period. For residues with intermediate ratios, temporary immobilization occurred during the first few weeks but was then followed by net mineralization (Justes et al. 2009). It is difficult to reliably determine the range of nitrogen made available to subsequent crops by cover crops because of all the aforementioned factors. Justes et al. (2012) tried to establish potential orders of magnitude based on the research to date. They found available nitrogen ranges of -20 to $+10\%$, -10 to $+30\%$, and $+10$ to $+50\%$ of absorbed N from grasses, crucifers, and legumes, respectively.

When digestate is applied to fields, the nitrogen made available to crops is the sum of the mineral nitrogen present in the digestate and the relative amount of organic nitrogen that mineralizes in the months following application. This figure corresponds almost entirely to the amount of N-NH_4^+ , since only 10–20% of the organic nitrogen is mineralized within six months (based on studies of manure digestates) (Möller and Müller 2012). Since organic nitrogen is mineralized during anaerobic digestion, nitrogen in energy cover crop digestate should be more readily available than nitrogen in cover crop residues (Möller and Müller 2012; Bareha et al. 2018). For example, it has been found that a mean energy cover crop digestate input of 30 t/ha should provide 37–179 kg N-NH_4^+ /ha (Bareha et al. 2018). Moreover, digestate application could be timed to better correspond to crop demands for nitrogen.

Energy cover crops are particularly likely to provoke preemptive competition since they are harvested late and are likely to deplete the mineral nitrogen in the soil as they grow. On the other hand, avoiding the incorporation of cover crops with high C:N ratios can limit or prevent nitrogen immobilization. Producing an energy cover crop yield of 4.5 t dry matter/ha requires the uptake of 60–100 kg of nitrogen, which could cause nitrogen stress for the next crop. By providing a source of nitrogen, such as digestate fertilizer, it may be possible to both meet the needs of cover crops and reduce the risk of nitrogen stress for the subsequent cash crop (Szerencsits 2014). While the latter sometimes occurs after winter energy cover crops (Szerencsits 2014; Marsac et al. 2019), it is less common after summer energy cover crops.

Indeed, the initial nitrogen requirements of winter cash crops are often low enough that nitrogen uptake by cover crops does not have a lasting impact. For example, Szerencsits (2014) did not observe any nitrogen stress after the use of a summer energy cover crop. However, in simulations comparing crop cycles with and without energy cover crops, it was necessary to fertilize winter wheat at higher levels when the cash crop followed an energy cover crop (+30–80 kg N/ha) to maintain yields at control levels (as defined in Launay et al. 2020). Thus, the risk of nitrogen stress could be lowered by providing mineral or organic fertilizer to the cash crop. Another solution could be utilizing an energy cover crop mixture containing legumes (Valkama et al. 2015).

In conclusion, anaerobic digestion can help promote the green manure effects of energy cover crops as long as the digestate contains levels of available nitrogen that can compensate for nitrogen losses during storage and digestate application. At the very least, removing residues from fields can prevent the nitrogen immobilization that can result from energy cover crops with high C:N ratios (Brozyna et al. 2013). Moreover, digestate application can be optimally timed to better respond to the nitrogen needs of cash crops (Möller and Stinner 2009).

1.5.2 Ammonia volatilization

Table 1. Effect of anaerobic digestion on the intensity of ammonia volatilization after application of digestate as fertilizer across field studies. +: significant increase in volatilization associated with digestate use; -: significant decrease in volatilization associated with digestate use; =: no change in volatilization associated with digestate use. ¹Laboratory experiment. ²Calcium ammonium nitrate, calcium nitrate, ammonium nitrate, urea.

Digestate substrate	Control	Application	Crop/soil applied to	Duration (days)	Effect on volatilization	Emission factor	Reference
Animal slurry and/or food waste	Animal slurry	Surface	Wheat; grassland; bare soil	2-5	+	10-40% of total N or N-NH ₄ ⁺ applied	Sommer et al. 2006 ¹ ; Möller and Stinner 2009; Nyord et al. 2012; Nicholson et al. 2017
Animal slurry with or without food waste	Animal slurry	Surface; Injected	Grassland; bare soil	4-19	=	30-42% of N-NH ₄ ⁺ applied on surface; 10% of N-NH ₄ ⁺ injected	Wulf et al. 2002; Chantigny et al. 2004
Pig slurry	Pig slurry	Surface	Bare soil; Grassland	8-10	-	10-18% of total N applied; 23% of N-NH ₄ ⁺ applied	Chantigny et al. (2007, 2009)
Energy crops with or without pig slurry	Animal slurry	Surface	Wheat; grassland; bare soil; corn	3	+	10-13% of N-NH ₄ ⁺ applied	Ni et al. (2012)
Energy crops or pig slurry or sewage sludge	Mineral fertilizer ²	Surface; incorporated; injected	Grassland; wheat/corn	2-8	+ or = if injected	10-14% of total N applied on surface; 6-29% of N-NH ₄ ⁺ applied on surface; 12% of N-NH ₄ ⁺ injected	Chantigny et al. 2007; Quakernack et al. 2012; Wolf et al. 2014; Zilio et al. 2021

When nitrogen fertilizer is applied, the degree of ammonia volatilization depends on fertilizer characteristics (pH, NH_4^+ content, dry matter content), soil characteristics (pH), the application method (surface vs. injection), and climatic conditions (temperature, wind speed, rainfall) (Ni et al. 2012; Möller 2015). Higher pH and NH_4^+ content both enhance volatilization (Möller 2015). Temperature and wind speed are positively correlated with the degree of volatilization because they promote ammonia's transition from its liquid phase to its gaseous phase; the occurrence of rainfall immediately after application strongly reduces volatilization (Ni et al. 2012). Since volatilization occurs at the interface with the atmosphere, injecting the fertilizer limits its exposure, reducing volatilization (Webb et al. 2010; Maris et al. 2021). Similarly, low levels of dry matter content allow the fertilizer to better infiltrate the soil, also reducing volatilization. Consequently, the identity of the crop in place when fertilizer application occurs can have an indirect effect on volatilization by slowing down infiltration; for example, the presence of corn will result in less volatilization than the presence of grasslands or wheat because the soil is largely bare under corn (Ni et al. 2012; Quakernack et al. 2012). The same thing for the presence of crop residues (Maris et al. 2021). Studies have found that volatilization is generally higher for digestate than for slurry (Table 1, Table 22). Increases in pH and NH_4^+ levels during anaerobic digestion should have a marked effect on volatilization. However, some work has observed a decrease in volatilization for digestate versus untreated slurry when the former is more fluid than the latter (Chantigny et al. 2004, 2007, 2009). Very little research has considered anaerobic digestion in areas with arable crops but no livestock, which means that there have been few examinations of digestate serving as a substitute for mineral fertilizer. In their meta-analysis, Pan et al. (2016) found that, on average, mineral fertilizers lose 18% of applied nitrogen to volatilization, recognizing that urea releases the most emissions and that substituting in non-urea-based fertilizers can reduce volatilization by 75%. Chantigny et al. (2007) found that digestate emitted three times more ammonia than did mineral fertilizer, while Wolf et al. (2014) and Quakernack et al. (2012) found much larger differences since emissions from the mineral fertilizer control occurred at levels deemed to be negligible. On the contrary, Zilio et al. (2021) found that with good spreading practices, i.e. injecting the digestate directly, no difference was visible with urea.

Most studies to date have looked at digestates produced from monofermented slurry. Digestates resulting from the codigestion of crop residues are more viscous than are pure

slurry digestates (Plöchl et al., 2009 in Quakernack et al., 2012). Thus, infiltration-mediated reductions in volatilization are probably less pronounced for crop residue digestates. The few studies examining digestates from monofermented or cofermented crops have found that 6 to 29% of the N-NH_4^+ supplied is released via volatilization (Ni et al. 2012; Quakernack et al. 2012; Wolf et al. 2014), which falls within the value range for digestates of all origins (6–42% of the N-NH_4^+ or total nitrogen supplied). In the small number of studies where the digestate was injected, ammonia emissions drop to 4–12% of the N-NH_4^+ supplied (Wulf et al. 2002; Zilio et al. 2021; Maris et al. 2021). In addition, it is important to recognize that the use of nitrogen fertilizers will increase if energy cover crops are added to rotations, a move that will enhance volatilization overall. If the release of ammonia is not limited, there could be negative impacts on air quality and greenhouse gas emissions.

1.5.3 Nitrate leaching

Nitrate leaching occurs during the drainage period. The best way to minimize leaching is to keep soil mineral nitrogen as low as possible before and during this time. One way to achieve this outcome is by planting a fall cover crop, which will take up mineral nitrogen from the soil. On bare soils, such an approach efficiently limits nitrate leaching, mainly by reducing soil mineral nitrogen but also by decreasing drainage (Justes et al. 2012; Tribouillois et al. 2016a; Meyer et al. 2019). The degree of efficacy is species dependent: on average, non-legumes versus legumes reduce leaching by 70% and 40%, respectively (Tonitto et al. 2006; Tribouillois et al. 2016a).

Table 2. Effect of anaerobic digestion on the intensity of nitrate leaching across field studies. +: significant increase in leaching or leaching risk associated with digestate use; -: significant decrease in leaching or leaching risk associated with digestate use; =: no difference in leaching or leaching risk associated with digestate use. ¹Ammonium sulfate, ammonium nitrate, potassium nitrate. ²Pot experiment.

Digestate substrate	Control	Application rates (using equivalents)	Agricultural practices	Effect on leaching	Reference
Animal manure and/or energy crops or food waste	Mineral fertilizer ¹	Total N or N use efficiency for one study	Applied in spring or fall to a grassland or in spring to corn	= when same N use efficiency and – when same total N applied	Matsunaka et al. 2006; Chantigny et al. 2008; Walsh et al. 2012a ² ; Svoboda et al. 2013; Tsachidou et al. 2019
Animal manure and/or energy crop and/or food waste	Animal manure	Total N or others	Applied in spring or in fall to grassland or different rotations	= for 9 studies out of 12	Jäkel and Mau 1999 in Svoboda et al. 2013; Pötsch 2004 in Möller 2015; Brenner and Clemens 2005 in Svoboda et al. 2013; Börjesson and Berglund 2007; Chantigny et al. 2008; Möller and Stinner 2009; Goberna et al. 2011; Walsh et al. 2012a; Sieling et al. 2013; Svoboda et al. 2013; Nicholson et al. 2017; Sigurnjak et al. 2017
Animal manure and/or crop residues + energy cover crops	Cover crops and crop residues left behind	Total N or others	Applied in spring; 3 out of 4 years to 1 out of 3 years	= for the highest frequency of application and – for others	Möller and Stinner 2009; Gunnarsson et al. 2011; Brozyna et al. 2013

Fertilizer use may increase leaching risks if application takes place in the fall just before the drainage period and/or if too much is employed. The European Union's Nitrate Directive (91/676/EEC) prohibits applying mineral nitrogen fertilizers, as well as some organic nitrogen fertilizers (with C:N ratios < 8), including digestates, to winter crops seeded in the fall. It similarly prohibits the use of all fertilizer types on bare soil from early summer to February, before spring crops are planted, if no cover crops are seeded. Medium- and long-term leaching risks can also be increased by higher levels of soil organic matter, which can arise from repeated cover crop use or the application of organic amendments. In such cases, levels of mineralized nitrogen will climb (Constantin et al. 2011) if fertilizer quantities are not concomitantly reduced (Schröder et al. 2007; Constantin et al. 2012; Girault et al. 2019). In the short term, using digestate as fertilizer does not increase leaching risks if fertilizer levels are calculated based on the nitrogen use efficiency for the digestate and if the digestate is applied when nitrate leaching risks are low, such as after the planting of a crop with high nitrogen uptake (Matsunaka et al. 2006) (Table 2, Table S223).

During the winter fallow period, leaching risks can be reduced by growing energy cover crops without fertilizer; the effect is similar to that obtained with cover crops. When Riau et al. (2021) tested the efficacy of three energy cover crop species, they found that black oat reduced leaching more than did ryegrass or forage rapeseed because the former had faster, more uniform development. When black oat was grown without fertilizer as an energy cover crop and harvested in the spring, it was more effective than the same species terminated early and left in the field as a cover crop (Möller and Stinner 2009; Gunnarsson et al. 2011). Indeed, when residues undergo mineralization in the late fall and early spring, it creates the opportunity for the nitrogen absorbed by the cover crop to leach (Tribouillois et al. 2016a). In contrast, the digestate created from the harvested energy cover crops is applied at a time when leaching risks are lower (Möller and Stinner 2009; Gunnarsson et al. 2011). Applying fertilizer to energy cover crops does not appear to diminish their ability to reduce nitrate leaching. First, Heggenstaller et al. (2008) observed lower levels of leaching in systems with energy cover crops versus in systems with bare soil despite the higher levels of nitrogen fertilizer usage across the crop succession. Second, modeling research showed that, when identical species were used, spring-fertilized, harvested energy cover crops reduced leaching more than unfertilized cover crops that were destroyed a couple of weeks early (Szerencsits 2014; Malone et al. 2018).

Malone et al. (2018) found that fertilized, harvested rye reduced leaching by 18% compared to unfertilized, unharvested rye and by 54% compared to what was seen on bare soil. Similarly, Szerencsits (2014) found that, in multiyear experiments, fertilized winter energy cover crops reduced leaching by 20% compared to the same species when destroyed 15 days earlier and by 25% compared to what was seen on bare soil. This result could be explained by greater biomass production leading to a larger reduction in drainage (Szerencsits 2014) or the decrease in nitrogen mineralization due to residue removal (Malone et al. 2018). Summer energy cover crops also seem to be effective in reducing leaching during the following winter compared to what is seen on bare soil in summer (Szerencsits 2014; Girault et al. 2019).

Clearly, cover crops have a demonstrated ability to reduce leaching. Initial studies of energy cover crops suggest that they display this function, which is sometimes even enhanced. Managing energy cover crops in specific ways can affect nitrate leaching dynamics: (i) leaching can be reduced by producing more biomass and avoiding asynchrony between residue mineralization and nitrogen uptake by the following crop, and (ii) leaching may be increased in the medium to long term if nitrogen fertilizer is used. It is important to underscore that long-term research in this area remains scarce, and it is necessary to further explore the effects of crop cycle management when rotations include energy cover crops.

1.5.4 Nitrous oxide emissions

Nitrous oxide (N_2O) is emitted mainly during denitrification, i.e. the transformation of NO_3^- into N_2 , as an intermediate product under anaerobic conditions. A small portion of N_2O is also emitted as a co-product during nitrification, i.e. the transformation of NH_4^+ into NO_2^- then into NO_3^- under aerobic conditions (Hénault et al. 2012). Both reactions are influenced by the availability of their substrate (NH_4^+ for nitrification and NO_3^- for denitrification) (Hénault et al. 2012; Nicholson et al. 2017) and organic carbon can boost the activity of denitrifying bacteria if it is easily mobilized/degradable (Möller and Stinner 2009). Beyond that, N_2O emissions are mostly influenced by climatic conditions: temperature and moisture (Petersen 1999; Hénault et al. 2012). Soil moisture above a certain threshold promotes denitrification by creating anoxic conditions (Möller and Stinner 2009).

Table 3. Effect of anaerobic digestion on the intensity of nitrous oxide emissions across field studies. +: significant increase in N₂O emissions associated with digestate use; -: significant decrease in emissions associated with digestate use; =: no difference in emissions associated with digestate use.

¹Modeling.

Digestate substrate	Control	Application	Effect on N ₂ O emissions	Emissions factor	Reference
Animal slurry with or without food waste and energy crops	Animal slurry	On surface; incorporated; injected	-72% to +126%	0.08-1.9% of total N	Petersen 1999; Wulf et al. 2002; Amon et al. 2006; Clemens et al. 2006; Vallejo et al. 2006; Chantigny et al. 2007; Möller and Stinner 2009; Chantigny et al. 2010; Thomsen et al. 2010; Senbayram et al. 2014; Rodhe et al. 2015; Baral et al. 2017; Herrmann et al. 2017; Nicholson et al. 2017
Crop residues and energy cover crops and grass	Cover crops left behind		-25 to -38%	1% of total N	Möller and Stinner 2009; Szerencsits 2014 ¹

Within crop cycles, there is no clear consensus on the effect of cover crops on N₂O emissions (Blanco-Canqui et al. 2015; Kaye and Quemada 2017; Abdalla et al. 2019; Guenet et al. 2020). During their growth, they reduce the amount of N available to microorganisms and the amount of nitrate leached, thus reducing the risk of direct and indirect N₂O emissions. On the other hand, the decomposition of their residues after their destruction releases N₂O which tends to offset the previous effect (Viard et al. 2013; Blanco-Canqui et al. 2015; Guenet et al. 2020; Abalos et al. 2022). The magnitude of N₂O emissions depends on the C:N ratio of residues, their rate of decomposition and their incorporation or not into soil (Guenet et al. 2020; Abalos et al. 2022). For example, several studies have found an overall increase in N₂O emissions with the insertion of legume cover crops due to their low C:N (Tribouillois et al. 2018; Abdalla et al. 2019; Guenet et al. 2020).

Since digestates are richer in mineral nitrogen but poorer in labile carbon than their substrates of origin, their use as fertilizers could have contrasting impacts on nitrous oxide emissions. No consistent pattern has been seen in past research comparing the effects of digestates with their substrates of origin (Table 3, Table 24). When the soil is rich in carbon, either because it is covered by grassland or because of its crop history, labile carbon is no longer limiting denitrification, and digestate use is no longer advantageous (Vallejo et al. 2006; Pelster et al. 2012; Corré and Conijn 2016). Under dry conditions, nitrous oxide emissions largely arise from nitrification, whose rate outstrips that of denitrification. In this case, the supply of NH₄⁺ determines the level of nitrous oxide emissions (Möller and Stinner 2009). Reviewing available studies, we found an average field emissions factor for digestates of 0.52% (0.08–1.9%) of the total nitrogen applied. This figure is lower than the reference emissions factor provided by the IPCC (1%). The digestate application method influences the emissions factor. For example, injection reduces volatilization and increases denitrification (Wulf et al. 2002; Thomsen et al. 2010).

To date, only two studies have compared the effects of using crop residue digestates to leaving cover crop residues in the field; one was a field study, and the other was a modeling study (Möller and Stinner 2009; Szerencsits 2014). They reached the same conclusion: compared to terminating and incorporating cover crops into the soil, removing energy cover crop biomass to later return it as digestate seems to reduce nitrous oxide emissions. This difference can be

explained by the lower levels of labile carbon in the digestate versus in the incorporated cover crop (Möller and Stinner 2009).

1.5.5 Synthesis of the nitrogen balance

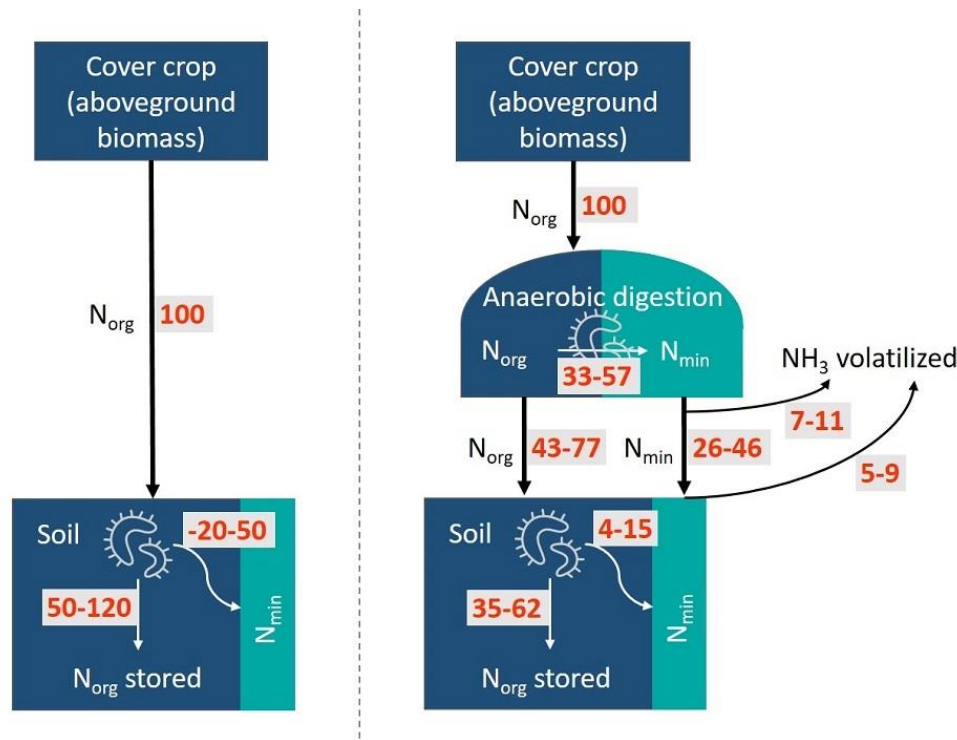


Figure 3. Schematic representation of the nitrogen balance after the incorporation of aboveground cover crop biomass into the soil (on the left) or the application of digestate obtained from anaerobically digested aboveground cover crop biomass (on the right). In both scenarios, initial cover crop nitrogen content is the same (100 units) and the mineralization period is short (5–6 months). Each number is a percentage of the original value. The cover crops' nitrogen mineralization figures were obtained from Justes et al. (2012); the nitrogen mineralization figures associated with anaerobic digestion were obtained from Bareha et al. (2018); and the digestate mineralization figures were obtained from Möller and Müller (2012). In both scenarios, we consider that ammonia volatilization took place during digestate storage and application. For the storage period, we considered that, on average, 20% of the $N-NH_4^+$ volatilized (Sommer 1997; Holly et al. 2017). For the application period, we assumed the same average level of volatilization (20%), based on figures for crop residues digestates (Table 1). The nitrogen balance represented here focuses on the mineralization of the nitrogen in cover crop residues in the soil. It does not represent nitrous oxide emissions or nitrate leaching. N_{org} : organic nitrogen; N_{min} : mineral nitrogen.

To summarize, we compared the nitrogen balance of a cropping system using multiservice cover crops and a cropping system using digestate produced via the anaerobic digestion of

energy cover crops (Figure 3). When the systems start with the same initial amount of nitrogen, the nitrogen potentially available to the plants across the crop rotation depends on the amount of available nitrogen added and lost (i.e., nitrogen is preserved during anaerobic digestion). Nitrogen mineralization during anaerobic digestion and the transformation of green manures into a controllable fertilizer increased nitrogen availability. With regards to the nitrogen lost, we saw no increase in nitrous oxide emissions, an increase in volatilization after digestate creation, and a potential decrease in leaching. Overall, digestate use seemed to slightly improve nitrogen balance, but the issue should be explored further as few studies are available for energy cover crops.

1.6 How the anaerobic digestion of energy cover crops affects carbon dynamics

By fixing atmospheric carbon dioxide, cover crops can increase the amount of carbon stored in the soil (Blanco-Canqui et al. 2015; Poeplau and Don 2015; Kaye and Quemada 2017; Tribouillois et al. 2018; Jian et al. 2020). Incorporating the above- and belowground biomass of cover crops could result in the storage of 320 kg C/ha per year, based on a meta-analysis by Poeplau and Don (2015), or even 560 kg C/ha per year, according to a meta-analysis by Jian et al. (2020). In France, the proportion in the cover crops of potential remaining carbon after application to soil is 28% on average, similar in magnitude to that of grain straw (Justes et al. 2012). The main factors driving carbon storage levels are the frequency of cover crop inclusion and cover crop biomass (Launay et al. 2021a). Species identity also has an effect, given differences in C:N ratios. The meta-analysis by Jian et al. (2020) showed that cover crop C:N ratios tend to negatively correlate with the amount of carbon stored in the soil. Residues with high C:N ratios are hardly stabilized due to the lower carbon use efficiency of decomposers (Sinsabaugh et al. 2016) arising from stoichiometric constraints in organic matter decomposition; microorganisms C:N ratios vary generally between 6 and 11 (Bertrand et al. 2019). Furthermore, it is important to look at the distribution of carbon in aboveground versus belowground biomass. Indeed, the belowground sources of organic carbon (i.e., roots and rhizodeposition) contribute more to soil carbon levels than do aboveground sources (Chenu et al. 2019). Due to their physical and chemical nature and incorporation depths, belowground carbon sources are more effectively stabilized by adsorption or physical protection (Chenu et al. 2019).

Anaerobic digestion increases substrate stability (i.e., Stumpe et al. 2012; Wentzel et al. 2015; Coban et al. 2015; Möller 2015). The organic carbon remaining in digestate is at least 50% more stable than it is in the initial substrate (Chen et al. 2012; Thomsen et al. 2013; Béghin-Tanneau et al. 2019). Consequently, carbon sequestration is equivalent in the initial crop biomass and its various byproducts (animal digested and/or biogas plant digested) and corresponds to 12–14% of the carbon present at the start, according to Thomsen et al. (2013). Other work has found that the digestion of corn results in a sequestration level of 23% of the carbon initially present in corn; in contrast, direct incorporation of corn residues does not result in carbon sequestration but rather in the release of 4% of the initial carbon (Béghin-Tanneau et al. 2019). This result primarily arises because the fresh biomass triggered a significant priming effect on soil organic matter mineralization. A temporary inhibition of certain microbial activities after digestate application has been observed, which contrasts with the boost in response to slurry (Abubaker et al. 2015). Chen et al. (2012) had already demonstrated that digestate use had less of a priming effect on soil organic matter degradation than did crop residue incorporation.

Marsac et al. (2019) identified some factors affecting how energy cover crops can influence carbon storage. Using above- and belowground biomass data in AMG model, they observed that harvested energy cover crops could result in as much carbon storage as incorporated cover crops grown over shorter periods without fertilizer (and that thus produced less biomass). Indeed, energy cover crops grown over longer periods with fertilizer would leave behind, post harvest, quantities of stubble (1–2 t dry matter/ha depending on cutting height) and roots (~20% of total biomass) equivalent to quantities of cover crop residues. Choosing a cover crop species with a high root:shoot ratio, such as grass (Constantin et al. 2011), could (i) enhance carbon returns via belowground biomass and (ii) provide carbon more effectively stabilized than aboveground biomass (Chenu et al. 2019). Marsac et al. (2019) found that, if the resulting digestate was applied as fertilizer, the levels of stored carbon would exceed those associated with incorporated cover crops. Subsequently, Szerencsits (2014) assessed the humic balance using the above findings in conjunction with the method described in Kolbe (2007). It was found that applying the digestate derived from the aboveground biomass more efficiently stored carbon than leaving the biomass in place as residues.

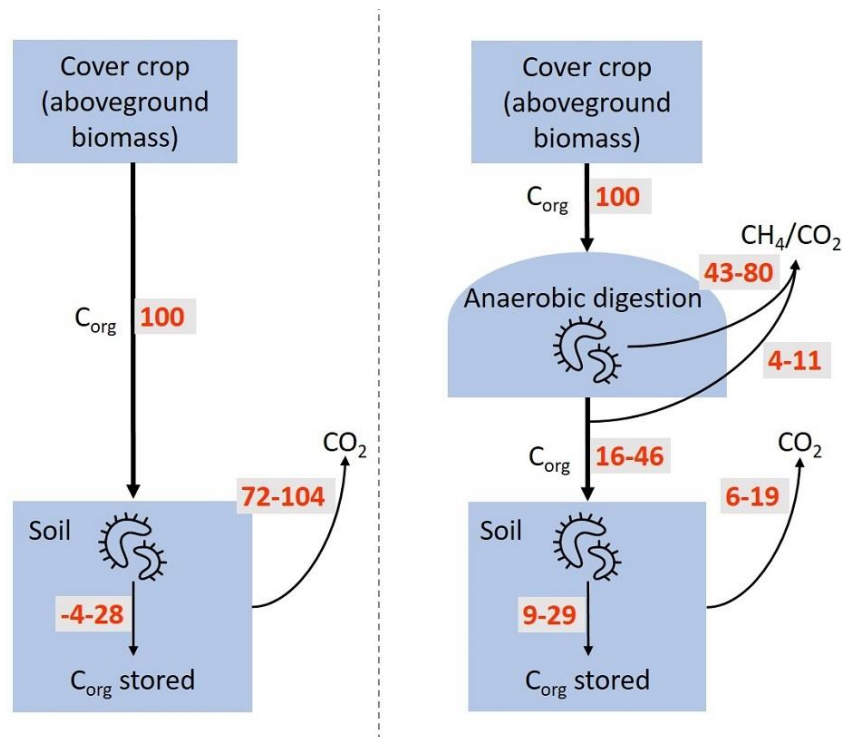


Figure 4. Schematic representation of the carbon balance after the incorporation of aboveground cover crop biomass into the soil (on the left) or the application of digestate obtained from anaerobically digested aboveground cover crop biomass (on the right). In both scenarios, initial cover crop carbon content is the same (100 units). Each number is a percentage of the original value. The figures for the carbon mineralization of cover crop residues are based on the decomposition of corn (i.e., serving as a summer energy cover crop) in Thomsen et al. (2013) and Béghin-Tanneau et al. (2019) and on the decomposition of winter cover crops in Justes et al. (2012). The figures for the cover crop decomposition via anaerobic digestion were taken from Thomsen et al. (2013), Bareha et al. (2018), and Béghin-Tanneau et al. (2019). We assumed that, on average, 20% of the digestate's carbon was lost during storage (Bareha et al. 2021a). The figures for the decomposition of cover crop digestate were taken from Thomsen et al. (2013) and Béghin-Tanneau et al. (2019). C_{org} : organic carbon.

In conclusion, our initial results suggest that the use of energy cover crops can have rather positive impacts on carbon storage, when the results are compared to those for cover crops incorporated into the soil (Figure 4). Although some carbon is lost during anaerobic digestion, net levels of soil organic carbon are seemingly unaltered because (i) biomass production increases, increasing the amount of carbon returned below ground and (ii) the remaining carbon is stabilized during anaerobic digestion. However, this assessment is based on a handful of studies. Some results are still being discussed and investigated, such as the extent of carbon stabilization during anaerobic digestion compared to carbon losses in biogas or the amount

of above-ground and root biomass left in the field by energy cover crops compared to traditional cover crops.

1.7 How the anaerobic digestion of energy cover crops affects soil biological activity

1.7.1 Microbial activity

During their growth, cover crops increase microbial abundance and activity via their inputs of carbon from root exudates and root turnover (Elfstrand et al. 2007; Blanco-Canqui et al. 2015; Finney et al. 2017). A recent meta-analysis from Muhammad et al. (2021) found that cover crops significantly increase microbial biomass compared to a bare soil from 24 to 51% depending on the indicator. On average, they increase the abundance of bacteria by 15% and the abundance of fungi by 19%, thus increasing the fungi/bacteria ratio. In general, non-legumes increase the abundance of microorganisms slightly more than legumes due to higher C substrate supply through higher biomass production. In addition, bacteria and fungi respond differently to these two groups of species. Non-legumes favor fungi because they are specialized in the decomposition of high C:N residues, whereas bacteria specialized in low C:N residues are favored by legumes. Among fungi, arbuscular mycorrhizal fungi (AMF) are particularly important for crop production because they improve nutrient uptake and provide resistance to drought and soil pathogens (Soti et al. 2016). In addition, they can have a stabilizing effect on the soil by entangling soil particles with their mycelium or by sticking them together with glomalin, which is a glycoprotein produced by AMF that acts as a glue (García-González et al. 2018). Because they live in symbiosis with their host plants, fallow periods are particularly detrimental to mycorrhizal fungi (Soti et al. 2016). On average, cover crops increase AMF abundance, AMF root colonization and AMF spore density by 26%, 13%, and 47%. Legumes had slightly less effect than non-legumes because increased N returns may be deleterious to AMF root colonization (Muhammad et al. 2021). Schipanski et al. (2014) calculated that after a winter cover crop, the roots of the following crop were colonized at 100% of their potential by AMF against 85% if the soil was left bare during winter. Cover crops residue management has an impact on microbial community abundance and structure. Exporting residues as well as leaving them on the surface reduces the abundance of bacteria (+10% relative to bare soil) compared to incorporating them (+25% relative to bare soil). The

abundance of fungi is not impacted, in all cases it is increased. But on AMF in particular, exporting residues improves root colonization less (+5%) than incorporating residues (+50%) and seems to have a little less effect than residues left on the surface (+10%) (Muhammad et al. 2021). Finally, cover crops also tend to increase the size of earthworm populations, resulting in increased water infiltration and soil aggregate stability (Blanco-Canqui et al. 2015).

In the short term, digestate use tends to increase soil microbial activity, compared to the use of mineral fertilizers or no fertilizers, although the boost is less than that provided by undigested substrates (based on measurements of induced respiration; Fuchs et al. 2008; Abubaker et al. 2015; Möller 2015; Gómez-Brandón et al. 2016; Risberg et al. 2017). This climb in microbial activity is not due to the digestate adding microorganisms to the soil since such microorganisms do not persist in the soil (Fuchs et al. 2008; Stumpe et al. 2012; Coelho et al. 2020). Nor is it correlated with the quantity of carbon supplied (Abubaker et al. 2015). Instead, it is associated with the quality of carbon supplied (Stumpe et al. 2012; Wentzel and Joergensen 2016). DNA analysis and the quantification of taxon-specific growth rates have revealed that a shift may occur in microbial communities due to the lack of readily degradable organic matter (Chen et al. 2012; Abubaker et al. 2013). Fast-growing microorganisms (r-strategists) that preferentially degrade labile organic matter disappear; they are replaced by slow-growing microorganisms (K-strategists) that more efficiently degrade recalcitrant organic matter. This change induces a modification in the ratio of fungi to bacteria (Chen et al. 2012). Differences between treatments tend to fade a few months or years into digestate use (Walsh et al. 2012b; Abubaker et al. 2013; Möller 2015; Gómez-Brandón et al. 2016; Sadet-Bourgeteau et al. 2018). Consequently, a single dose of exogenous organic matter has a temporary effect on microbial communities, depending on dose size. In the case of repeated applications, the effects on microbial communities can be long lasting and associated with changes in soil chemical characteristics such as pH, cation exchange capacity, and soil organic carbon (Sadet-Bourgeteau et al. 2018). Several studies have shown that soil type also has a significant impact: clay soils are more resilient than sandy soils (Walsh et al. 2012b; Abubaker et al. 2013; Wentzel et al. 2015). If a cover crop is in place when fertilizer is applied, microbial population size is not directly affected by the fertilizer's physicochemical characteristics, but is rather indirectly affected by the characteristics' impact on plant growth (Terhoeven-Urselmans et al. 2009; Walsh et al. 2012b; Abubaker et al. 2013).

1.7.2 Earthworms

Rollett et al. (2020) observed a positive correlation between the amount of organic matter supplied and the increase in earthworm population size. Sizmur et al. (2017) has shown that it is the quantity of energy provided by the organic matter that matters most; it is therefore organic matter quality that strongly affects earthworms. Digestate is a source of food for earthworms, particularly anecic earthworms. In the short term, digestate use increases earthworm abundance (Clements et al. 2012) and biomass, as seen in field and microcosm studies (Ernst et al. 2008; Koblenz et al. 2015; Sizmur et al. 2017). Endogeic earthworms are not able to directly consume organic matter from digestate (Ernst et al. 2008), but they can still benefit from the input of energy by consuming the waste generated by anecic earthworms (Koblenz et al. 2015). In some cases, short-term mortality has resulted from the high quantity of ammonium introduced by larger doses of digestate or slurry ($> 170 \text{ kg N.ha}^{-1}$) (Johansen et al. 2015; Tigini et al. 2016; Renaud et al. 2017; Rollett et al. 2020). Sizmur et al. (2017) showed that, when equivalent levels of carbon were used, straw increased the biomass of an anecic earthworm, *Lumbricus terrestris*, significantly more than did plant digestate because of the higher energy input. Similarly, in a field study, Frøseth et al. (2014) observed that the immediate incorporation of green manure increased the size of the earthworm population compared to the use of plant digestate. In the long term, such differences seem to disappear (Johansen et al. 2015; Koblenz et al. 2015; Rollett et al. 2020). However, there are no long-term studies on the impacts of directly incorporating cover crop biomass into the soil versus returning later in the form of digestate.

1.8 The impact of energy cover crops and their digestate on water dynamics

Cover crops can have complicated effects on groundwater recharge. First, by covering the soil, they can increase transpiration and reduce evaporation (Qi and Helmers 2010; Nielsen et al. 2015; Tribouillois et al. 2016). Second, they can increase water infiltration and reduce runoff (Snapp et al. 2005; Blanco-Canqui et al. 2015; Yu et al. 2016). A recent meta-analysis by Meyer et al. (2019) found that, in most studies, cover crops decreased drainage, although the results were highly variable (-110 to $+40 \text{ mm}$). Depending on climatic conditions, this reduction in drainage may represent a small or a large percentage of annual water drainage, which could

have major implications for water recharge in dry regions. Cover crop biomass seems to be one of the main determinant factors, with seeding date close behind (Meyer et al. 2020; Tribouillois et al. (2018). Tribouillois et al. (2018) observed that increases in cover crop biomass were strongly correlated with increases in evapotranspiration and decreases in drainage. However, at a certain threshold of biomass (< 2.5 t dry matter/ha) or leaf area index values, evapotranspiration showed no further increases (Meyer et al. 2020). Based on this work, advancing seeding by one month can result in a threefold difference in the degree of drainage reduction; the termination date does not affect drainage but does affect soil water levels for the next crop. Based on these findings, the large quantities of biomass produced by energy cover crops should not significantly reduce drainage, compared to what is seen for multiservice cover crops. However, the seeding date should be chosen so as to trade off between biomass production and groundwater recharge. In any case, the broader-scale use of cover crops (whether multiservice or energy) could create challenges for groundwater recharge, an issue should be assessed.

The above increase in evapotranspiration could result in a water deficit for the following summer crop. The depletion of water reserves in surface has often been seen in association with multiservice cover crops or energy cover crops terminated/harvested in the spring (Krueger et al. 2011; Alonso-Ayuso et al. 2014; Blanco-Canqui et al. 2015; Marsac et al. 2019; Meyer et al. 2020). During the first months after cover crop seeding, water profiles are generally the same for fields with cover crops and fields with bare soil because of heavy rainfall (Alonso-Ayuso et al. 2014). However, in the spring, the profiles begin to differ as the cover crop grows, notably if rainfall levels do not compensate for evapotranspiration levels (Alonso-Ayuso et al. 2014; Meyer et al. 2020). The factor with the greatest impact is termination date (Krueger et al. 2011; Alonso-Ayuso et al. 2014; Meyer 2020), followed by crop species, and amount of precipitation (Meyer 2020). Thus, the next crop is likely to face water stress if termination takes place later; if the cover crop produces large quantities of biomass; if climatic conditions are dry; and if water storage capacity is low. Apart from this latter situation, water stress appears to be minimal in the temperate zone, even if termination occurs at a later date, because the soil (particularly the first centimeters) has time to recharge before the next crop is established (Szerencsits 2014; Blanco-Canqui et al. 2015; Marsac et al. 2019; Meyer 2020). These findings suggest there is a risk of water stress for the subsequent crop when cultivating energy cover

crops under specific circumstances (i.e., late destruction and high levels of biomass production), an issue that should be studied further. Mean quantities of digestate ($30 \text{ m}^3 \cdot \text{ha}^{-1}$) contain less than a millimeter of water, which does not at all compensate for water depletion by cover crops. With regards to summer energy cover crops, it is theoretically possible for them to reduce soil temperatures, thus reducing evaporation and leading to greater water reserves than what is seen on bare soil (Blanco-Canqui et al. 2015). While such a result was observed for a multiservice cover crop at one site during a dry year (Blanco-Canqui et al. 2015), it was not observed for energy cover crops at an experimental site in southwestern France (Marsac et al. 2019). More studies are needed on this topic.

1.9 The impact of energy cover crops and their digestate on soil physical properties

Because they provide cover during periods when the soil would usually be left bare, cover crops reduce wind and water erosion. They are particularly effective at protecting sensitive soils, such as sandy soils (Snapp et al. 2005). Using a meta-analysis, Daryanto et al. (2018) found that, on average, cover crops reduce the amount of soil lost by 75%, compared to situations in which the soil is left bare over the winter. In their study, Du et al. (2022) even found an average reduction of 90% at different points of the globe. The determinant factors were the degree of cover and cover duration (Snapp et al. 2005; Blanco-Canqui et al. 2015). By reducing erosion, cover crops also reduce the loss of dissolved nutrients (e.g., phosphorus and nitrate) via runoff (Blanco-Canqui et al. 2015). In addition to directly protecting the soil from the disturbance caused by rain and wind, cover crops improve soil structural stability (Blanco-Canqui et al. 2015; Daryanto et al. 2018). Cover crops rapidly increase the stability of aggregates (< 3 years) by protecting them against the impacts of raindrops; by providing root-mediated carbon inputs; and by boosting microbial activity (Blanco-Canqui et al. 2015). Increasing aggregate stability subsequently increases water retention, carbon storage, macroporosity, and root growth; it decreases the soil's susceptibility to compaction (Blanco-Canqui et al. 2015).

Few studies have looked at the impact of digestate use on the physical properties of soils (Möller 2015). Albuquerque et al. (2012) performed a two-year experiment but found no effect of digestate use on structural stability when compared to other treatments (i.e., no amendment, mineral fertilizer, or cattle manure). Some studies cited in Möller (2015) found a positive effect

of digestate use on bulk density, hydraulic conductivity, water retention capacity, and aggregate stability, compared to what was seen for unamended soil. Béghin-Tanneau (2020) also observed an increase in aggregate stability following digestate application over periods of 12 to 265 days. However, the digestate had a significantly weaker effect than its substrate of origin (corn). Similarly, Sarker et al. (2018) found that while digestate use increased aggregate stability, the effect was less pronounced than that seen for alfalfa residues. These results were attributed to a correlation between the decomposability of the organic residues and both soil microbial activity and aggregate stability.

Consequently, introducing energy cover crops into crop cycles and utilizing the resulting digestate as fertilizer should help reduce erosion and promote aggregate stability. As these services are furnished during the growing period, they will be unaffected by the fact that energy cover crops are harvested rather than being left in place. On the contrary, service quality should be better than that provided by conventional multiservice cover crops because their magnitude is positively correlated with biomass. That said, energy cover crops have one drawback compared to multiservice cover crops: when three crops are cultivated in two years instead of two, field traffic climbs, increasing the risk of soil compaction (Peters et al. 2016; Quennesson and Decaux 2020). Ensiling the energy cover crops and applying the digestate (Duttmann et al. 2014; Lantz and Börjesson 2014) requires the use of heavier machinery, sometimes under sensitive conditions during the early spring or fall. The risk of soil compaction is particularly high on clay soils and can lead to yield losses (Lantz and Börjesson 2014). However, this risk can be reduced by using tank-free spreading systems (Lantz and Börjesson 2014) or controlled traffic farming systems for silage operations (i.e., the equipment always follows the same path) (Duttmann et al. 2014). Moreover, commonly used energy cover crop species are rarely taproot species, which are able to loosen the soil (Chen and Weil 2010; Blanco-Canqui et al. 2015).

1.10 Impacts on cropping systems and farms

1.10.1 Food/feed production

In the previous sections, we noted that energy cover crops can reduce the yield of subsequent crops because of preemptive competition for water and nitrogen and because of increased soil compaction risks. However, the greatest potential deleterious effect of energy cover crops on subsequent crops is associated with the delay in seeding and the use of early varieties

(Szerencsits 2014; Marsac et al. 2019). Szerencsits (2014) observed that the yield of spring crops declined by an average of 10% if seeding was delayed by more than 7 days, and Marsac et al. (2019) observed a 7% loss in yield if the delay attained 10–15 days. When the delay was even longer (one month or more), the next cash crop could not reach maturity before harvest. Thus, the cash crop can no longer feed humans but can be used to feed animals or can undergo anaerobic digestion (Graß et al. 2013; Peters et al. 2016; Quennesson and Decaux 2020). In such systems, the objective is to optimize the production of both crops in tandem, and both crops are harvested before maturity. In such cases, there is no longer a clear distinction between the cash crop and the cover crop. These systems do not align with the intended purpose of energy cover crops, which is to produce biomass for energy purposes without replacing food crops. The widespread use of such systems in areas where food crops are grown could end up reducing overall food production (Kemp and Lyutse 2011; WWF France 2020).

1.10.2 Nitrogen balance at the farm level

A survey program in France contacted farmers with anaerobic digesters and obtained data to calculate the nitrogen balance on their farms. Unfortunately, most were livestock farmers, and only a small number (9 out of 46) had introduced energy cover crops to their crop rotations. None of these nine farms increased their mineral fertilizer consumption following the introduction of anaerobic digestion and energy cover crops. Four of them even reduced their mineral nitrogen fertilizer purchases (ADEME and Solagro 2018). However, this survey does not allow us to isolate the impact of energy cover crops. Anaerobic digestion is often accompanied by other changes in farm practices and, above all, by exchanges of materials with neighboring farms, industries and collectivities. In addition, these farms still have insufficient hindsight on their new production system to observe long-term effects on soil fertility. According to field trials or simulations, introducing a third fertilized crop within a two-year rotation would likely mean an increased need for nitrogen (Heggenstaller et al. 2008; Igos et al. 2016; Berti et al. 2017; Girault et al. 2019). Additional nitrogen would be required to meet the energy cover crop's needs and, possibly, to compensate the following crop for nitrogen lost between the ensiling of the energy cover crop and the application of the resulting digestate; there could also be preemptive competition for nitrogen. The use of synthetic fertilizers could be reduced by codigesting farm-derived biomass with externally derived biomass or by using legumes alone or in mixture as energy cover crops.

1.10.3 Life cycle assessment

A recently released Life Cycle Assessment (LCA) found that an anaerobic digestion scenario with 50% energy cover crops in the feedstock supply performed better than a non-biogas scenario with multiservice cover crops on indicators of energetic resource depletion, climate change and ozone depletion (Esnouf et al. 2021). This study considered the production of heat energy through the combustion of methane injected into the network, the management of livestock effluents and soil fertilization. The finality of the biogas produced had a strong impact on these indicators. In LCAs studying biogas transformed by cogeneration, the poor valorization of heat completely degrades the environmental balance of the anaerobic digestion (Bacenetti et al. 2016; Hijazi et al. 2016). If we look at the greenhouse gas balance in more detail, the studies agree that double cropping increased i) nitrogen fertilizer use and therefore N_2O emissions in the field or CO_2 emissions upstream and ii) field operations and thus CO_2 emissions from fuel combustion (Igos et al. 2016; Berti et al. 2017; Maier et al. 2017; Esnouf et al. 2021). However, the soil C storage and above all the substitution of fossil gas largely compensated these side-effects in the study of Esnouf et al. (2021) where the anaerobic digestion scenario reduced by 75% the greenhouse gas emissions. The indicators of fine particle emissions, environmental acidification and terrestrial eutrophication that were also measured in this study depend to a very large extent on ammonia emissions during storage and spreading of the effluent and digestate. In this case, adopting anaerobic digestion with good storage and spreading practices improved the performance on these indicators compared to the reference scenario. Conversely, not covering the digestate could increase greenhouse gas emissions by 80% (Bacenetti et al. 2016; Esnouf et al. 2021). Finally, still in the same study, for indicators related to electricity consumption and fertilizer consumption, the performance was worse with anaerobic digestion but the introduction of legumes in energy cover crops and the optimization of digestate spreading equipment could reduce these impacts by 10 to 50%. The use of legumes had already been noted to reduce the greenhouse gas balance related to the reduction of the use of synthetic fertilizers and the reduction of N_2O emissions (Stinner 2015). Other LCAs exist in the literature but they rather study energy crops whose impact on land use change strongly influences the performance on the climate change indicator (Bacenetti et al. 2016; Igos et al. 2016; Hijazi et al. 2016). Styles et al. (2015) compared different energy production systems, including one in which corn serves as a summer energy

cover crop or as a simple energy crop. They found that the greenhouse gas balance of the first system was rather neutral compared to the baseline system. In contrast, anaerobically digesting dedicated crops increased emissions, notably because additional land was needed to compensate for the loss in food production.

1.11 Conclusions

This review reveals that the use of energy cover crops and their digestates has several advantages. In addition to allowing the production of renewable energy, the crops can provide several ecosystem services, including improved water quality, climate change mitigation, reduced soil erosion, and increased microbial activity. Thus, to answer the question raised in the introduction, we can still consider them as multiservice cover crops. However, they could also have some disservices, such as reduced groundwater recharge and the need for increased nitrogen inputs. Furthermore, energy cover crops are not always used as intended, leading to competition with food crops. Energy cover crops can compete with food crops for water and nutrients. This competition can be limited if there are sufficient levels of spring precipitation and if fertilizer is used. Additionally, incorporating energy cover crops into rotations induces changes in cropping systems that can lead to certain excesses, where energy cover crop production is favored to the detriment of food crop production. Cropping systems should be designed so as to maximize the non-energy-related services provided by energy cover crops. Alternatively, trade-offs should be identified, such that energy cover crops can be treated more as multiservice cover crops than as cash crops. In this way, farmers would view energy cover crops not only as a new income source, but also as a way to improve their fields over the long term (e.g., via increased soil organic matter, improved soil structural stability, decreased pest pressure). Some research is still needed to expect widespread adoption of energy cover crops by farmers. We can suggest a few leads. At the varietal selection level, improvements are possible to adapt forage species to double cropping. In terms of technical management, the problem of summer cover crops establishment need a solution; seeding under cover could be an opportunity to explore. Finally, at the academic level, we have a great deal of knowledge about cover crops and digestates that allows us to speculate on the impacts of energy cover crops. They remained to be confronted with the field in a wide variety of situations.

Learnings

Cropping systems integrating ECCs have been little studied so far and the few studies are concentrated in Europe. However, by crossing the literature on cover crops and the literature on the use of AD digestate, we were able to formulate hypotheses on what this new practice could achieve:

- ECC with the return of digestate would store as much carbon in the soil as MSCC left in place.
- The insertion of fertilized ECCs is expected to increase overall N consumption and NH_3 volatilization associated. The use of digestate should create additional NH_3 volatilization.
- Exporting biomass from ECCs and returning it in a more stable form would decrease N_2O emissions compared to MSCCs left in place.
- Even if fertilized, the ECCs would reduce nitrate leaching compared to bare soil and compared to MSCCs due to higher biomass production and residue export.
- ECCs should decrease drainage in the same way as MSCCs compared to bare soil but irrigation of ECCs could increase pressure on the water resource.
- The insertion of fertilized ECCs is expected to increase overall N consumption despite
- ECCs are more likely to reduce the yield of the following crop than MSCCs because of their late harvest, but more importantly because of changes in farmers' practices who would sow their crops later with earlier varieties.

In the next chapters, we will confront these assumptions with data obtained in laboratory, field and modeling experiments.

Chapter 2 – Productivity and impacts on water and nitrogen of double cropping systems integrating winter and summer energy cover crops: Results from a field experiment coupled with modeling

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2.1 Introduction

Agroecology promotes the use of cover crops for a more environmentally friendly agriculture (Altieri 2018). From there, two paths emerge: the insertion of cover crops left in place to produce ecosystem services and the insertion of harvested cover crops to produce renewable energy.

Since 1991, the European Nitrate Directive (91/676/EEC) has encouraged and then required farmers to grow cover crops for a minimum of two months during winter fallow in areas classified as nitrate vulnerable zones. The objective is to protect groundwater quality by limiting nitrate leakage from agricultural soils to the water tables. These particular cover crops are called "catch crops" (CC) in reference to their role as nitrate traps. The species used are fast-growing species with a high capacity to absorb nitrogen (e.g. mustard, radish, etc.). Conservation agriculture already advocated the use of cover crops during fallow periods looking for other services (Scopel et al. 2013; Du et al. 2022). The desired services are mainly soil protection against erosion and the green manure effect providing nitrogen to the following crop. Therefore, the sown species differed from catch crop species with a high soil cover rate for the first service and legumes for the second service. Cover crops provide additional ecosystem services: climate change mitigation, disease and weed control, biodiversity improvement, soil structure improvement, etc. (Blanco-Canqui et al. 2015; Abdalla et al. 2019). Thus, they are now called "multi-services cover crops" (MSCC) and not "catch crops" or "green manure".

Today, the use of cover crops as biomass input for anaerobic digestion is growing with the development of renewable gas production. The gas used in the world is almost exclusively from fossil origin (IEA 2021). The production of renewable gas could contribute to the decrease of fossil gas consumption and to mitigate the global warming, in addition to other priority actions such as the decrease of energy use. Anaerobic digestion is the most successful process today to produce renewable gas. Germany is the world leader in the production of biogas by anaerobic digestion with more than 10,000 sites in operation by 2018 (OFATE 2020). In France, the sector is younger but has been growing rapidly until the Covid crisis (GRDF et al. 2022). It has been estimated that 30% of the biomass used in biogas plants could come from energy cover crops (ECC) by 2050 (ADEME 2018). However, the French regulation wants to limit competition between food and energy crops. To this end, the main crops have been defined

as the crops present on June 1 in the field, when energy cover crops are sown between two main crops (Article D543-291). They introduced a limit of 15% of main crops in the ration of biogas plants (Article D543-292). Unlike MSCC, the ECCs are harvested at the end of the fallow and not destroyed and left on the field. They are sown to produce biomass on the first place. Therefore, they are fertilized (most often with digestate from anaerobic digestion but also with mineral fertilizer) and irrigated if the regulations allows it. In addition, ECC can be sown in autumn between a crop harvested in summer and the following crop sown in spring, so-called winter ECC, or the ECC is sown after harvesting of crop in late spring and the following main crop sown in late autumn, so-called summer ECC. Launay et al. (2022) reviewed the current knowledge on the ecosystem services that ECCs could provide. From this review, energy cover crops could perform as well as MSCC in terms of soil carbon storage, reduction of nitrate leaching, reduction of N₂O emissions, reduction of erosion and increase of soil microbial activity during fallow. However, ECCs could decrease the production of the following spring crop, increase the risk of soil compaction, and decrease drainage.

The first objective of the study was to evaluate the impact of ECCs on nitrogen losses (ammonia volatilization, N₂O emissions by nitrification and denitrification, nitrate leaching), groundwater replenishment, and the production of the following crop in comparison to fallows with bare soil or planted with unharvested MSCC. We mobilized an approach combining a field experiment and field modeling to quantify the different fluxes and stocks of interest, considering the N cycle together with water fluxes.

2.2 Material and Methods

2.2.1 Methodological approach

In order to quantify the impact of fallow management on N, water and C fluxes in a crop succession, we used the following methodological approach:

1. A two-year field experiment was conducted to compare the impact of implanting cover crops with three types of management against a soil left bare during intercropping. The monitoring concerned soil water content (SWC), soil nitrogen content (SNC), aboveground biomass produced by cover crops and main crops as well as their N content, and ammonia volatilization following N fertilizer applications.

2. Data collected in the field experiment were used to calibrate the STICS crop model and evaluate the model capacity to predict the experimental results measured in the field experiment. We assumed that if dynamics of SWC and SNC, crop biomass and nitrogen content were simulated accurately, then those of water and nitrogen fluxes would also be simulated accurately. After verifying satisfactory agreement between model predictions and experimental observations, STICS was used to predict evapotranspiration, water drainage, nitrate leaching, N₂O emissions, nitrogen and water stress for crops (processes not measured in the field) for the two experimental years.
3. To integrate the variability of the experimental and simulated results with the interannual variability of climatic conditions of the site, we used the calibrated STICS model to repeat the experiment over a 30-year climate series.

2.2.2 Field experiment

2.2.2.1 Experimental site and design

A field experiment was conducted in southwestern France at Auzeville-Tolosane (43°31'N, 1°30'E) over the period summer 2020–spring 2022 (Figure 5). The weather station at the experimental field provided weather data (precipitation, global radiation, minimum and maximum temperature, Penman potential evapotranspiration) for the last 31 years (1991–2022). The average temperature, precipitation, potential evapotranspiration and radiation per year are described in the Table 25. The climate of this site is temperate with a mean (\pm standard deviation) annual precipitation of 639 ± 120 mm, an annual potential evapotranspiration (Penman equation) of 1031 ± 64 mm, and an average temperature of $15.2 \pm 6.7^\circ\text{C}$. The spring and summer 2021 (124 and 152 mm, respectively) were more humid than spring 2022 (87.5 mm) and summer 2020 (52 mm) (Table 26). The soil was deep (≥ 90 cm), a clay loam type with 34% clay in the tilled horizon (0–30 cm) and a pH of 8.2. The organic matter content of 1.2% of the tilled horizon was slightly high compared to same soils of the area but within the French average.

Two parallel crop successions were set up in order to be able to study various management of summer or winter fallow period (Figure 5). In the “summer” half, the succession was: winter durum wheat (*Triticum turgidum subsp. durum*) – short summer fallow (from June 30, 2020 to

November 6, 2020) – winter barley (*Hordeum vulgare* L.) – short summer fallow (from July 2, 2021 to November 21, 2021) – winter pea (*Pisum sativum* L.). In the “winter” half, the succession was: winter durum wheat – winter long fallow (from June 30, 2020 to May 5, 2021) – sunflower (*Helianthus annuus* L.) – winter long fallow (from September 14, 2021 to May 12, 2022) – sorghum (*Sorghum bicolor* L.). In both cases, the fallow management modalities were: i) bare soil (BS), ii) energy cover crop (ECC) and iii) multi-services cover crops (MSCC). In the “winter” succession, another management option was added with a catch crop (CC). In both parallel trials, all modalities were repeated three times in randomized blocks of 6x20 m plots. In addition, four large plots (24x20 m) were set on the same soil and with similar crop successions to monitor ammonia volatilization after fertilizer application on the energy and multi-services cover crops.

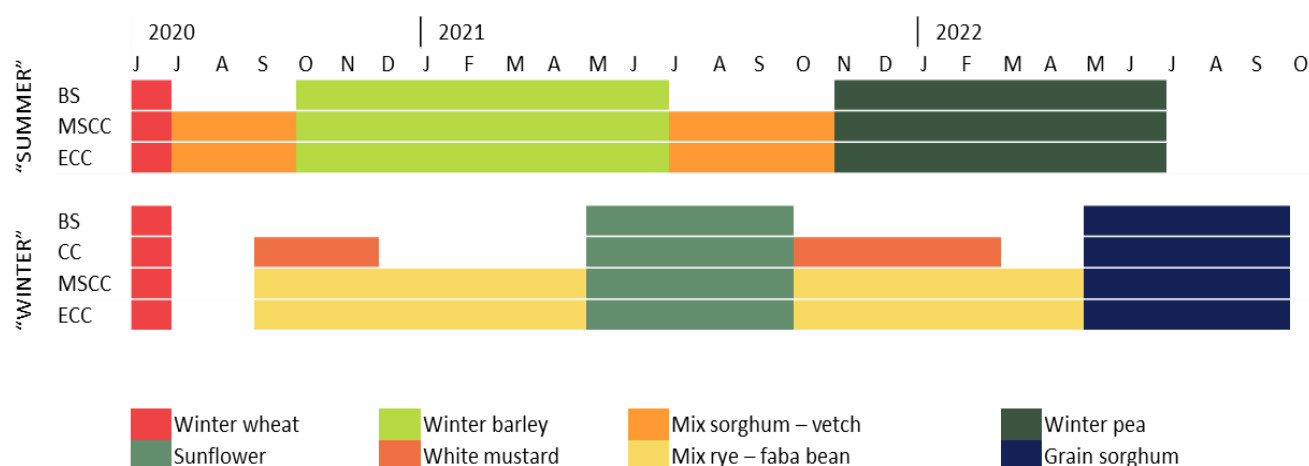


Figure 5. Crop successions of “summer” half and “winter” half of the experiment with the different modalities.

2.2.2.2 Cover crop management

Choice of species

We chose to sow cereal-legume mixtures as cover crops: a mixture of Bengal vetch (*Vicia benghalensis* L.) – forage sorghum (*Sorghum sudanense* × *Sorghum sudanense*) in summer and faba bean (*Vicia faba* L.) – rye (*Secale cereale* L.) in winter. These species were chosen according to their potential biomass production in the region. In winter fallow, the additional catch crop modality was sown with white mustard (*Sinapis alba* L.) classically used in France as a nitrate catch crop.

Table 4. Details of the cover crops management on both parts of the field experiment ("summer" and "winter" parts).

		Sowing	Termination	Fertilization	Irrigation
"Summer" 2020	MSCC	2020/07/10 <i>Sorghum: 125 plt.m⁻²</i> <i>Vetch: 90 plt.m⁻²</i>	2020/09/17 <i>Destroyed</i>	/	2020/07/13 40 mm 2020/07/31 35 mm
	ECC	2020/07/10 <i>Sorghum: 125 plt.m⁻²</i> <i>Vetch: 90 plt.m⁻²</i>	2020/09/17 <i>Harvested</i>	2020/07/10 <i>55 kg N-NH₄⁺.ha⁻¹</i> <i>from digestate</i>	2020/07/13 40 mm 2020/07/31 35 mm
"Summer" 2021	MSCC	2021/07/16 <i>Sorghum: 125 plt.m⁻²</i> <i>Vetch: 90 plt.m⁻²</i>	2021/10/20 <i>Destroyed</i>	/	/
	ECC	2021/07/16 <i>Sorghum: 125 plt.m⁻²</i> <i>Vetch: 90 plt.m⁻²</i>	2021/10/20 <i>Harvested</i>	2021/07/16 <i>63 kg N-NH₄⁺.ha⁻¹</i> <i>from digestate</i>	/
"Winter" 2020-21	CC	2020/09/18 <i>Mustard: 230 plt.m⁻²</i>	2020/11/24 <i>Destroyed</i>	/	/
	MSCC	2020/09/18 <i>Rye: 135 plt.m⁻²</i> <i>Faba bean: 70 plt.m⁻²</i>	2021/05/03 <i>Destroyed</i>	/	/
	ECC	2020/09/18 <i>Rye: 135 plt.m⁻²</i> <i>Faba bean: 70 plt.m⁻²</i>	2021/05/03 <i>Harvested</i>	2021/03/03 <i>67 kg N-NH₄⁺.ha⁻¹</i> <i>from digestate</i>	/
"Winter" 2021-22	CC	2021/10/20 <i>Mustard: 215 plt.m⁻²</i>	2022/03/28 <i>Destroyed</i>	/	/
	MSCC	2021/10/20 <i>Rye: 240 plt.m⁻²</i> <i>Faba bean: 20 plt.m⁻²</i>	2022/05/10 <i>Destroyed</i>	/	/
	ECC	2021/10/20 <i>Rye: 240 plt.m⁻²</i> <i>Faba bean: 20 plt.m⁻²</i>	2022/05/10 <i>Harvested</i>	2022/03/31 <i>75 kg N-NH₄⁺.ha⁻¹</i> <i>from digestate</i>	/

Sowing

The experiment started after the harvest of the previous durum wheat with the soil preparation. Throughout the trial, there was no tillage with soil inversion nor deep tillage (>15 cm). The wheat stubble was crushed and the soil was tilled to a depth of 10 cm. Before each sowing, the soil was tilled to a depth of 10-15 cm and then the cover crops were sown with a combined seeder. The bare soil was tilled several times to keep it bare throughout the fallow period. The

details about the sowing date and density are presented Table 4. The sowing density of winter ECC and MSCC mixture were readjusted in the second year. Indeed, the first faba bean having frozen in January and the rye having been strongly competed by weeds, we changed the faba bean and rye varieties in the second year and increased the density of rye. Because the summer 2020 was dry, the summer cover crops received two rounds of water of 40 and 35 mm 15 days apart. With more rainfall in 2021, the cover crops were not watered.

Fertilization

Table 5. Chemical composition (g.kg^{-1} of fresh weight) of the digestate used in the field experiment and sampled at application dates.

Date of analysis	Dry matter (%)	Total N	$\text{NH}_4^+\text{-N}$	Organic C	C : total N	pH
2020/07/10	4.7	4.59	1.56	15	3.2	8.8
2021/03/03	5.9	4.97	1.92	19	3.8	8.5
2022/03/31	2.7	3.3	2.1	8.2	2.5	7.7

No mineral fertilizers were applied to the cover crops. The ECCs were fertilized with a raw digestate while the other cover crops were not fertilized (Table 4). The digestate came from an agricultural biogas plant located in the neighboring department. Its characteristics, measured on three of the four applications, are described in Table 5. The application rate was 35-40 $\text{m}^3.\text{ha}^{-1}$ to target about 60 $\text{kg N-NH}_4^+.\text{ha}^{-1}$. The digestate was injected directly to a depth of a few centimeters on the sowing day of the summer energy cover crops or applied to the surface with a hose on the winter energy cover crops at the end of winter.

Termination

The mustards were crushed and plowed in at flowering, i.e. two months after sowing the first year and five months after sowing the second year due to the later sowing (Table 4). The ECCs were mowed and exported. The MSCC were crushed and plowed in the day the ECC were harvested.

2.2.2.3 Main crops management

Table 6. Details of the main crops management on both parts of the field experiment ("summer" and "winter" parts).

		Sowing	Harvest	Fertilization
"Summer" 2020	MSCC	2020/11/06 <i>Winter barley: 200 plt.m⁻²</i>	2021/07/01	2021/03/03 <i>50 kg N.ha⁻¹ from ammonium nitrate</i>
	ECC	2020/11/06 <i>Winter barley: 200 plt.m⁻²</i>	2021/07/01	2021/03/03 <i>67 kg N-NH₄⁺.ha⁻¹ from digestate</i>
	BS	2020/11/06 <i>Winter barley: 200 plt.m⁻²</i>	2021/07/01	2021/03/03 <i>50 kg N.ha⁻¹ from ammonium nitrate</i>
"Winter" 2020-21	CC	2021/05/05 <i>Sunflower: 7 plt.m⁻²</i>	2021/09/13	/
	MSCC	2021/05/05 <i>Sunflower: 7 plt.m⁻²</i>	2021/09/13	/
	ECC	2021/05/05 <i>Sunflower: 7 plt.m⁻²</i>	2021/09/13	/
	BS	2021/05/05 <i>Sunflower: 7 plt.m⁻²</i>	2021/09/13	/

The winter barley was sown after the summer cover crops or the bare soil in the "summer" part of the trial (Table 6). It received a nitrogen fertilizer application at the end of winter. The fertilizer was ammonium nitrate in the BS and MSCC modalities and digestate for the ECC modality. The sunflower was sown soon after the ECC harvest and MSCC soil incorporation on the "winter" part. No mineral fertilizer was applied on sunflower because of enough soil nitrogen supply. We tilled the soil over 10 cm before sowing each crop and used a combined seeder.

2.2.2.4 Measurements

Soil water and mineral nitrogen content

Soils were sampled from 0-90 cm in all treatments before sowing and after the destruction/harvest of the cover crops and at the beginning and end of winter in each modality (seven times in total in each half of the trial; Table 27). Soils were sampled from 0-30 cm seven days and about one month after the fertilizer applications in each modality. Soil water,

ammonia and nitrate content were measured in the laboratory. To determine the water content, soils were weighed fresh, dried for 48h at 105°C and reweighed according to standard ISO 17892-1. Ammoniacal N and nitric N determinations were performed by extraction with potassium chloride solution using an automated spectrophotometric method according to standard ISO 14256-2.

Plant measurements

We counted the number of plants emerged per meter in two rows for cover crops and sunflower to calculate an emergence density (Table 27). At harvest of the main crops, three samples of 1 m² per plot for sunflower and five samples of 0.25 m² per plot for barley were used to estimate straw and grain production. The above-ground biomass production of the cover crops was estimated at harvest or destruction using the same methodology as for barley, with a particularity for ECCs for which we separated the stubble samples (below the 10 cm cutting height) from the harvested part (above the cutting height) after the first harvest in 2020 (Table 27). We considered that the weeds present in the cover crops were part of it and therefore measured their biomass with that of the sown species. Dry matter content was determined by weighing fresh matter and then weighing after drying at 80°C for 48 h according to standard ISO 712. N concentration in grain and vegetative parts was determined with the Dumas method according to standard ISO 16634-2 for vegetative parts and barley grains and ISO 16634-1 for sunflower grains.

Ammonia volatilization

Air ammonia concentrations were measured after fertilizer applications for 20 days (Table 27) using the inverse dispersion modeling method applied to a multiple source (Loubet et al. 2018). Measurements were made on all the plots of the ECC modality that received digestate in summer before sowing the cover crop and at the end of winter on the barley, and on the "control" plots of the MSCC modality that received mineral fertilizer at the end of winter on barley and no fertilizer on cover crops. The model (FIDES) relates the concentration measured at a given location and height to the emission source and the background concentration by means of a transfer function calculated from measurements of atmospheric turbulence. It requires the concentration of NH₃ in the air of each plot, as well as background levels, along with measurements of atmospheric turbulence. NH₃ concentrations were measured using ALPHA passive samplers (Adapted Low-cost Passive High Absorption; Tang et al. (2001)). Three

samplers were placed at 50 cm and three samplers at 70 cm above the canopy or bare soil in the 24x20 m plots. The small 6x20 m plots were also used as pseudo-replicates. On these, three sensors were placed at 30 cm above the canopy or bare soil. To measure background concentrations of NH_3 , three additional poles with three samplers each (approximately 3 m high) were installed 400 m from the experimental field in three opposing directions. During the measurement period, wind speed and three-dimensional wind direction were measured with a sonic anemometer (20 Hz frequency) installed at a height of 2 m on the field site. The samplers were theoretically changed 12h, 1d, 2d, 3d, 5d, 15d, 20d after the fertilizer application. Ammonium was extracted from the samplers in the laboratory and determined by conductimetry.

2.2.3 Modeling

2.2.3.1 STICS modeling approach

The Multidisciplinary Simulator for Standard Crops (STICS) model (Beaudoin et al. 2022) was used i) to simulate carbon, nitrogen and water fluxes during the field experiment and assess the fluxes not actually measured in the field and ii) to repeat the experiment over 30 independent climate years (past climate). This model was chosen for its ability to reproduce daily the interactions between climate, soil, plant and technical itinerary as well as the coupling of nitrogen, water and carbon cycles. A previous version of the model (v8.2.2) was validated on more than 1800 field crop production situations in France (Coucheney et al. 2015). The version used (v2802) integrates now the description of dynamic root mortality (Beaudoin et al. 2022) and a new formalism of organic fertilizer mineralization (Levavasasseur et al. 2021) with the calibration of default parameters for digestate (Levavasasseur et al. 2022a).

After a calibration step of the missing cover crops species (sorghum and rye) from the two trial years, we used the technical cropping management of the first experimental year and simulated this crop sequence for 30 different climatic years (from July, 8 to July, 8 of the following year for the “summer” part and from August, 31 to September, 29 of the following year for the “winter” part). Indeed, except for the frost of the faba bean, the first year of experimentation was more consistent with the experimental design than the second experimental year. Therefore, each year of simulation starts with the initial soil status of 2020 and the technical operations were always performed at the same date. The only exception was the irrigation during summer, which was reduced to one water turn of 20 mm or less at sowing

of the cover crops triggered by the model depending on the climatic year. Because the model was not able to simulate our mixtures, the sorghum–vetch mixture of the summer MSCC and ECC modalities became a pure sorghum crop and the rye–faba bean mixture became a pure rye crop.

2.2.3.2 Calibration

White mustard, common cover crop species in France, was already calibrated in the model and used otherwise, but not sorghum and rye. For sorghum, we started from a parameterization previously created for corn. For rye, we started from the triticale parameterization. Some parameters related to growth temperatures, the nitrogen dilution curve, and the conversion of radiation into biomass were issued from bibliographical research (Table 7). The remaining missing parameters related to biomass production potential and sensitivity to water and nitrogen stress were mathematically optimized. The dataset used to optimize the sorghum parameters consisted of field measurements of cover crop biomass and nitrogen content, and the monitoring of soil mineral nitrogen content (SMNC) and soil water content (SWC) during both summer fallows. The same data was used to calibrate the rye during winter fallows. The optimization was performed in several steps: i) parameters related to the biomass production potential on the biomass production of the cover crops, ii) parameters related to the sensitivity to water stress on the biomass production of the cover crops and the evolution of the SWC, iii) parameters related to the sensitivity to nitrogen stress on the biomass production, the amount of nitrogen absorbed and the SMNC. This loop was performed twice on each species to obtain correct fits. The mathematical optimization of the parameters and the comparison between simulated and observed results were performed on R version 4.0.3 using the packages *SticsRFiles* version 0.4.2 and *CroPlotR* version 0.7.1.

Table 7. Values of the parameters used in the sorghum and rye STICS plant files obtained either from the scientific literature (reference in brackets) or by mathematical optimization or not changed from the original file.

Related process	Parameter	Description	Sorghum	Rye
Growth temperatures	tdmin	minimum temperature below which development stops	5.3 (Tribouillois et al. 2016b)	0.5 (Tribouillois et al. 2016b)
	tdmax	maximum temperature above which development stops	40.6 (Tribouillois et al. 2016b)	38.1 (Tribouillois et al. 2016b)
	temin	minimum temperature for development	9.4 (Tribouillois et al. 2016b)	0.5 (Tribouillois et al. 2016b)
	temax	maximal temperature above which plant growth stops	40 (Tribouillois et al. 2016b)	38.1 (Tribouillois et al. 2016b)
	teopt	range of optimal temperature for plant growth	30 (Nguyen and Blum 2004)	25 (Tribouillois et al. 2016b)
	teoptbis		35.6 (Tribouillois et al. 2016b)	30 (Tribouillois et al. 2016b)
	phyllotherme	thermal duration between the apparition of two successive leaves on the main stem		
Conversion of radiation into biomass	extin	extinction coefficient of photosynthetic active radiation in the canopy	0.3 (Clegg et al. 1974)	0.75 (Sieling et al. 2016)
Germination	potgermi	soil water potential under which seed imbibition is impeded	-0.8 (Tribouillois et al. 2016b)	-1.6 (original)
	tgmin	minimum temperature below which emergence is stopped	10 (Nguyen and Blum 2004)	0 (original)
Biomass production potential	efcroijuv	maximum radiation use efficiency during the juvenile phase	7	1.9 (original)
Sensitivity to water stress	efcroiveg	maximum radiation use efficiency during the vegetative stage	9.36	4.60
	psisto	potential of stomatal closing	19.50	15 (original)
	psiturg	water stress threshold below which cellular extension is reduced	13.44	4 (original)
	kmax	maximum crop coefficient for water requirements	1.09	0.5
	rapsenturg	soil water content threshold below which senescence due to water stress occurs	0.762	0.5 (original)
Sensitivity to nitrogen stress	croirac	elongation rate of the root apex	0.147	0.156
	Vmax2	maximum specific N uptake rate with the high affinity transport system	0.0171	0.1
	Kmabs2	affinity constant of N uptake by roots for the low uptake system	32669	40000
	innsen	parameter of the N stress function accelerating the senescence	0.279	0.17 (original)
	INNmin	minimum value of INN possible for the crop	0.304	0.60 (original)
	adil	parameter of the critical dilution curve $[N_{\text{plante}}] = \text{adil} \cdot \text{MS}^{-\text{bdil}}$	3.9 (Lemaire et al. 2008)	5.00
	bdil	parameter of the critical dilution curve $[N_{\text{plante}}] = \text{adil} \cdot \text{MS}^{-\text{bdil}}$	0.39 (Lemaire et al. 2008)	0.0108
	adilmax	parameter of the maximum dilution curve $[N_{\text{plante}}] = \text{adilmax} \cdot \text{MS}^{-\text{bdilmax}}$	4.7 (Barbanti et al. 2006)	7.3 (original)
	bdilmax	parameter of the maximum dilution curve $[N_{\text{plante}}] = \text{adilmax} \cdot \text{MS}^{-\text{bdilmax}}$	0.5 (Barbanti et al. 2006)	0.41 (original)

2.2.3.3 Model outputs

The model outputs were the above-ground and root biomass produced by the cover crops and the main crops, the carbon fixed and nitrogen absorbed by the plant biomasses, the nitrogen nutrition index (calculated as the ratio between the actual nitrogen concentration in the plant to the critical concentration), the water stress (depending on the available soil water content and some thresholds defined by species), the crops transpiration and soil evaporation, the quantity of water drained, the quantity of nitrogen leached, nitrified and denitrified. All of these variables were output either at the end of the fallow and at the end of the fallow-main crop succession and were averaged over the 30 years of independent simulations. The performance of the model on ammonia volatilization following organic fertilizer applications being inconclusive, we did not use this output variable. Instead, we entered into the model the emission factor obtained in the field which we did not vary according to the climatic years.

2.2.4 Statistical analysis

On the field experiment, the effect of modalities on the different variables was tested with two-factor analyses of variance by setting the block to fixed effect. If the modality effect was significant, then two-way comparisons of means were performed with Tukey tests. Since these tests were performed on a small number of points, we considered a significance level of 10% in the analysis. The postulates of homogeneity and normality of the residuals were verified.

Regarding the results of the simulations, the effect of the modalities on the model outputs was tested with two-factor analyses of variance by setting the year as a random effect. As for the tests on the field experiment, the postulates of the ANOVA were checked and if the modality effect was significant, then pairwise comparisons of means were performed. Here, given the large number of points, we considered a significance level of 5%. We tested the influence of climatic variables (sum of precipitation and irrigation, sum of temperature, sum of radiation) over different time windows (whole crop cycle, from sowing to 10 days after, from February 1st to destruction) on the production of ECCs and MSCCs by constructing a correlation matrix with the Pearson test.

2.3 Results

2.3.1 Field experiment

2.3.1.1 Cover crops biomass production

In the "summer" part of the field trial, the total above-ground biomass of the summer mixtures was 6.7 and 9.3 t DM.ha⁻¹ in 2020 compared to 2.6 and 4.4 t DM.ha⁻¹ in 2021 for the MSCC and ECC, respectively (Table 8). The production in 2021 was lower despite almost equivalent water inputs in both years, including rainfall and irrigation, and the same initial SWC in the first soil layer (Table 28). The digestate input to the ECC increased the above-ground biomass production by 40% in 2020 and 70% in 2021, compared to the MSCC (Table 8). The larger difference between MSCC and ECC in 2021 may be related to the initial SMNC in the MSCC modality being 50 kg N.ha⁻¹ lower in 2021 than in 2020 (Table 28). The digestate also significantly increased the total nitrogen absorbed in the ECC mixture. Of the two species in the mixture, sorghum production was higher than vetch production. In addition, vetch suffered from fertilization in the ECC modality unlike sorghum. The vetch accounted for less than 10% of the final biomass of all modalities except for MSCC in 2021, likely due to lower SMN availability (Table 28).

In the "winter" part, the mustard of the CC modality produced between 2.2 and 3.5 t DM.ha⁻¹ before its destruction at flowering in both years of experimentation (Table 8). The biomass of the other cover crops at the time of mustard destruction was lower (between 1.2 and 1.3 t DM.ha⁻¹). At their destruction in spring, the rye-faba bean mixtures in both MSCC and ECC modalities produced more biomass without significant difference between them (7.6 and 8.0 t DM.ha⁻¹ for the MSCC against 7.9 and 8.1 t DM.ha⁻¹ for the ECC in 2020-21 and 2021-22, respectively). The higher SMNC in the ECC modality than in the MSCC modality at the sowing of 2020 did not influence the final production (Table 28). The addition of digestate at the end of winter on the ECC did not increase biomass production even though it slightly increased nitrogen uptake in 2021-22 and more significantly in 2020-21 (Table 8). In 2020-21, mustard had absorbed as much nitrogen as the rye-faba bean mixture in the MSCC modality, at their respective destruction dates, despite its much shorter cycle and lower biomass production. In 2021-22, however, the ECC and MSCC took up significantly more nitrogen than the mustard, perhaps due to the increased proportion of rye in the mixture.

Table 8. Above-ground biomass produced ($t\ DM\ ha^{-1}$) and nitrogen assimilated in the above-ground biomass ($kg\ N\ ha^{-1}$) of cover crops during the field experiment. MSCC: multi-services cover crop, ECC: energy cover crop, CC: catch crop. Different letters indicate a significant difference ($P < 0.1$) between treatments.

Part of the trial	Treatment	Species	Biomass 2020-21	Absorbed N 2020-21	Biomass 2021-22	Absorbed N 2021-22
"Summer"	MSCC	Sorghum	5.96	68	1.45	23
		Vetch	0.24	9	0.63	23
		Weeds	0.46	10	0.47	10
		Σ	6.66 a	87 a	2.55 a	56 a
	ECC	Sorghum	8.77	98	2.90	53
		Vetch	0.13	5	0.26	8
		Weeds	0.42	13	0.23	6
		Stubble	/	/	1.04	21
		Σ	9.32 b	116 b	4.43 b	88 b
	CC	Mustard	2.16	91	3.50	63
		Weeds	0.06	2	0	0
		Σ	2.22 a	93 a	3.5 a	63 a
	MSCC	Rye	5.95	75	5.20	52
		Fababean	0.11	3	2.78	69
		Weeds	1.51	22	0	0
		Σ	7.57 b	100 a	7.98 b	121 b
	ECC	Rye	5.43	95	4.76	69
		Fababean	0	0	2.38	66
		Weeds	0.88	18	0	0
		Stubble	1.57	23	0.92	9
		Σ	7.88 b	136 b	8.06 b	144 b

2.3.1.2 Impact on the following crop

Table 9. Soil mineral N content (SMNC; kg N.ha^{-1}) and soil water content (SWC; mm.ha^{-1}) at the end of the fallow period the first year of experiment.

Part	"Summer"			"Winter"			
Date	2020/09/21			2021/04/05			
Treatment	BS	MSCC	ECC	BS	CC	MSCC	ECC
SMNC kg N.ha^{-1}							
0-30 cm	90	42	48	33	42	21	33
30-60 cm	71	20	20	17	25	13	16
60-90 cm	84	35	33	16	19	8	9
$\Sigma 0-90 \text{ cm}$	245	97	101	66	86	42	58
SWC mm.ha^{-1}							
0-30 cm	80	65	64	83	84	80	75
30-60 cm	92	68	63	85	88	76	75
60-90 cm	96	77	61	86	92	89	88
$\Sigma 0-90 \text{ cm}$	268	210	188	254	264	245	238

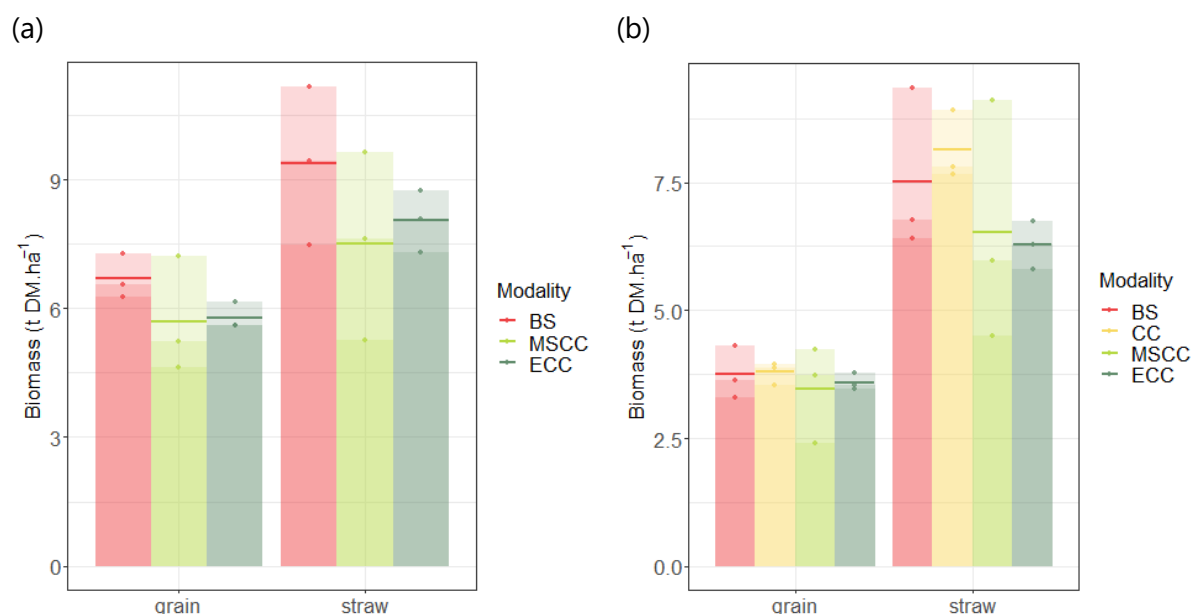


Figure 6. Grain and straw production (t DM.ha^{-1}) of winter barley in the "summer" part (a) and sunflower in the "winter" part (b) following each type of fallow management (bare soil, CC: catch crop, MSCC: multi-services cover crop, ECC: energy cover crop). Each point represents a field measurement, and the horizontal line the mean of the measurements. In the analysis of variance, the treatment effect was not significant for any of the represented variables so we did not perform a comparison of means.

In the "summer" part, SWC as well as SMNC just before barley sowing were strongly decreased by the cover crops, whether they were fertilized or not (-60% for SMNC and -25% on average

for SWC; Table 9). However, the grain and straw yields of barley did not decrease significantly, despite a downward trend (6.7 t DM.ha⁻¹ of grain and 9.4 t DM.ha⁻¹ of straw produced in the BS modality compared to 5.7 and 7.5 t DM.ha⁻¹ in the MSCC modality and 5.8 and 8.1 t DM.ha⁻¹ in the ECC modality; Figure 6a). The MSCC and ECC modalities differed in the variability of production: the standard deviation of grain and straw production were respectively 1.4 and 2.2 t DM.ha⁻¹ for the MSCC modality against 0.3 and 0.7 t DM.ha⁻¹ for the ECC modality.

In the “winter” part, the differences between the treatments on the soil condition before sowing sunflower were very small (Table 9). The SWC in the first horizon and in the whole profile were decreased in the order after CC (84 mm in 0-30 cm and 264 mm in 0-90 cm), bare soil (83 mm in 0-30 cm and 254 mm in 0-90 cm), MSCC (80 mm in 0-30 cm and 245 mm in 0-90 cm) and ECC (75 mm in 0-30 cm and 238 mm in 0-90 cm). The SMNC was decreased also in the same order: after the CC (86 kg N.ha⁻¹ in 0-90 cm), the bare soil (66 kg N.ha⁻¹), the ECC (58 kg N.ha⁻¹) and MSCC (42 kg N.ha⁻¹). However, these differences were not significant and did not significantly impact sunflower yields (Figure 6b). The sunflower grain yields were 3.75, 3.79, 3.46, and 3.60 t DM.ha⁻¹ in BS, CC, MSCC, and ECC modalities, respectively. The vegetative parts accounted for 7.51, 8.13, 6.52, 6.28 t DM.ha⁻¹ for BS, CC, MSCC, and ECC modalities, respectively. The shallow burial of the abundant MSCC residues degraded the sunflower seedbed, reducing sunflower emergence density (Table 29) but did not affect the average grain production per hectare. The sunflower production of grain and straw in this modality was nevertheless more irregular than in the other modalities with a standard deviation of 0.9 t DM.ha⁻¹ for grain and 2.4 t DM.ha⁻¹ for straw against 0.5 and 1.6 in the BS modality, 0.2 and 0.7 in the CC modality, and 0.2 and 0.5 in the ECC modality.

2.3.1.3 Ammonia volatilization

Ammonia volatilization following digestate application was the lowest for the summer ECCs due to digestate injection (6% of applied N-NH₄⁺). In winter, after surface application on ECC, 19% of N-NH₄⁺ on average was lost with a high variability (from 7% to 36% of N-NH₄⁺). At the same period, the digestate applied on barley in the ECC modality emitted 13% of the N-NH₄⁺. In comparison, the winter application of ammonium nitrate to barley in the MSCC modality emitted only 4% of the applied N-NH₄⁺.

2.3.2 STICS model calibration

The sorghum calibration provided satisfactory results for SWC and SMNC after cover crops destruction, biomass production and nitrogen uptake of the cover crops (Figure 28). The differences in production between the two years was well represented. However, the difference in production between the MSCC and ECC modalities remained underestimated. The model predicted a difference of 1.8 t DM.ha⁻¹ in 2020 against 2.6 t DM.ha⁻¹ in reality, and 0.1 t DM.ha⁻¹ in 2021 against 1.8 t DM.ha⁻¹ in reality. On the contrary, the model slightly overestimated the yield of barley following sorghum: + 0.7 and + 1 t DM.ha⁻¹ in MSCC and ECC, respectively.

The rye calibration also provided satisfactory results for SWC and SMNC after cover crops destruction, and for cover crops biomass production. However, the amount of N absorbed by the rye was underestimated (Figure 29). The calibration was done only on the year 2020-21. This year the faba bean dominated the mixture until it froze in January. Thus, on the model, for this particular year, we simulated a rye–faba bean succession by forcing the end of the juvenile phase of rye in early January. When testing the model on the year 2021-22 with a pure rye cover crop, we still obtained good results on SWC and cover crops biomass production. However, the simulated data of post-destruction SMNC and of the amount of N absorbed by the rye were largely underestimated compared to the mixture in the field (results not shown). In reality, the faba bean grew well within the mixture up to represent half of the biomass produced at the destruction. It can therefore be assumed that the symbiotic N fixation by the faba bean contributed substantially to the total amount of N absorbed by the cover crop mixture and explain the decrease of N absorption from the soil. The model strongly underestimated the yield of sunflower following rye: 0.68 t DM.ha⁻¹ in the MSCC modality and 1.22 t DM.ha⁻¹ in the ECC modality against 3.46 and 3.60 t DM.ha⁻¹ in reality. The nitrogen nutrition index was lower in these two modalities than in the bare soil and CC modalities, and the sunflower also suffered an early water stress (Figure 30).

2.3.3 Analysis of the additional variables simulated on the trial from the model

Table 10. Summary of the environmental outputs of the STICS model over the first year of experimentation. Transpiration and evaporation during the fallow period. Drainage, leaching and N₂O emissions during the fallow - main crop succession.

	Drainage	Transpiration	Evaporation	Leaching	Leaching	N₂O emissions
	mm.ha ⁻¹	mm.ha ⁻¹	mm.ha ⁻¹	kg N-NO ₃ ⁻ .ha ⁻¹	% of initial SMNC	kg N-N ₂ O.ha ⁻¹
"Summer"						
BS	216	0	136	184	96	8.7
MSCC	184	58	113	50	50	3.5
ECC	148	73	119	68	51	5.6
"Winter"						
BS	164	0	249	52	52	4.9
CC	161	33	220	12	17	4.5
MSCC	169	159	176	30	33	1.9
ECC	165	159	167	47	37	4.1

2.3.3.1 Water consumption and drainage

In the "summer" part, the simulated drainage began in early October 2020 for the summer BS modality and only in early December for the others, and ended in mid-February 2021. The simulated drainage compared to BS decreased of 15% in the summer MSCC modality and 30% in the summer ECC modality (Table 10). The MSCC had consumed 58 mm of water while 73 mm by the ECC. In addition, the summer cover crops did not reduce evaporation while they were in place. However, after their destruction, evaporation was reduced by 20 mm compared to bare soil thanks to their residues.

In the "winter" part, the simulated drainage occurred from early December 2020 to mid-February 2021 without any difference between the modalities (Table 10). In addition, no differences were simulated in evapotranspiration before and during the drainage period among the modalities. Between the sowing of the cover crops on 2020/09/18 and the end of the drainage period on 2021/02/22, all modalities had evapotranspired approximately 175 mm of water. The different cover crops decreased evaporation compared to bare soil but plant transpiration increased ending in similar water losses in all modalities. From March onwards,

evapotranspiration and especially transpiration of the MSCC and ECC modalities increased very rapidly. At the end, the CC consumed 33 mm, the MSCC 159 mm and the ECC 159 mm of water.

2.3.3.2 Nitrate leaching and N₂O emissions

According to the model, all cover crops, summer or winter type, reduced nitrate leaching during the winter 2020-21 (Table 10). Summer cover crops reduced nitrate leaching by almost 50% without difference between MSCC and ECC despite the addition of digestate. In winter fallow, mustard was twice as effective as the rye-faba bean mixture in the first year of the trial. The MSCC and ECC modalities reduced leaching by 19% and 15% respectively, while the CC modality reduced it by 35%. Digestate application on the winter ECC also had no impact because it was applied after the drainage period that year.

Summer cover crops were also effective in reducing simulated N₂O emissions during the fallow-main crop succession (Table 10). Nevertheless there was a difference between both modalities. The ECC modality with digestate return decreased the emissions by 36% while the MSCC modality without fertilization decreased them by 60%. In the “winter” part, the cover crops also reduced N₂O emissions but the MSCC modality was much more efficient than the ECC and CC modalities. The MSCC reduced N₂O emissions by 61% compared to 16% for the ECC modality and 8% for the CC modality. The CC reduced emissions more effectively than the other cover crops when it was in place until 2020/11/24. However, spring emissions rapidly increased until they exceeded the emissions of the other cover crops modalities.

2.3.4 Repetition over 30 years

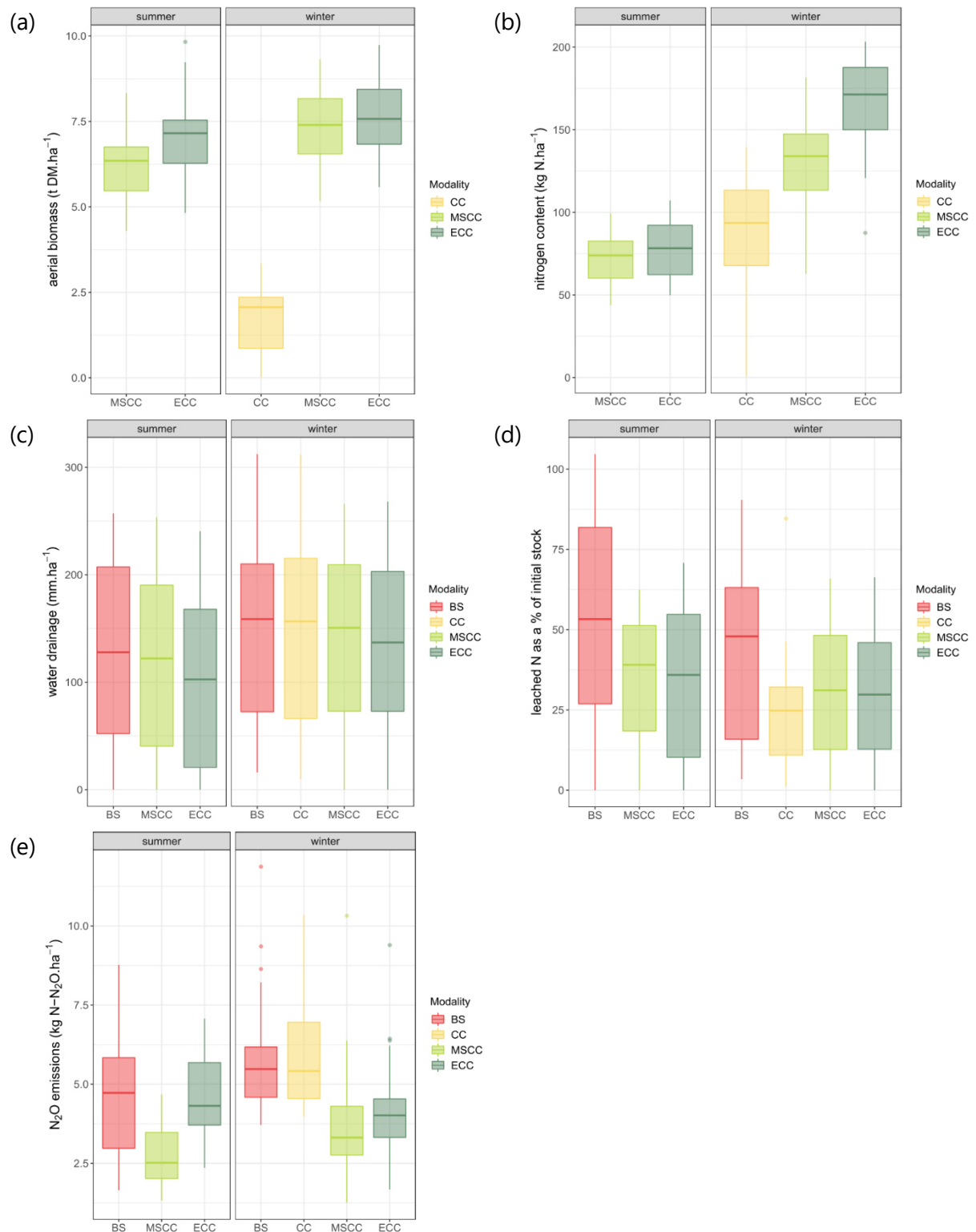


Figure 7. Climatic variability effect on (a) cover crops aerial biomass ($t MS \cdot ha^{-1}$) and (b) its nitrogen content ($kg N \cdot ha^{-1}$), (c) drainage ($mm \cdot ha^{-1} \cdot yr^{-1}$), (d) leaching (% of initial mineral content of the soil), and (e) N_2O emissions ($kg N-N_2O \cdot ha^{-1} \cdot yr^{-1}$) as simulated by STICS with initial data taken from the first year of field experiment.

2.3.4.1 Cover crops biomass production

When the first year of the trial was repeated over the last 30 climatic years, the difference in biomass and acquired N between the summer MSCCs and ECCs was greatly reduced compared to the experiment (Figure 7a, b, Table 8). In the simulations, the average biomass of MSCC was 6.2 t DM.ha⁻¹ with 72 kg N.ha⁻¹ absorbed and that of ECC was 7.0 t DM.ha⁻¹ with 78 kg N.ha⁻¹ absorbed, thus very close to the field production of MSCC in 2020 (Table 8). Interannual variability could be high with a minimum in 2014 for the MSCC modality of 4.3 t DM.ha⁻¹ and a maximum in 2019 for the ECC modality of 9.8 t DM.ha⁻¹ (Figure 31). In some very dry years during the fallow period (50.5 mm in 2003 and 42.5 mm in 2009), cover crop growth was severely limited, despite irrigation at emergence. More than the amount of water received during the crop cycle, it is the amount of water received 10 days after sowing that seemed to correlate better with the biomass produced (Pearson correlation coefficients at 0.22 and 0.31 respectively). For example, in 2015 sorghum received 158.5 mm of rainfall and irrigation between sowing and harvest but only 21.5 mm of water in the 10 days after sowing. The cover crop was therefore not very productive: 5.3 t DM.ha⁻¹ in the MSCC modality. The sum of the global radiation during the crop cycle was also correlated to the produced biomass (correlation coefficient of 0.29). The correlation coefficients of these two factors remained low, and were not sufficient to explain all the interannual variability of the production.

In the 30-year simulations of the “winter” part, the average biomass obtained for the CC modality was 1.7 t DM.ha⁻¹ with 85 kg N.ha⁻¹ absorbed, for the MSCC modality 7.3 t DM.ha⁻¹ with 133 kg N.ha⁻¹ and for the ECC modality 7.6 t DM.ha⁻¹ with 169 kg N.ha⁻¹ (Figure 7a, b). These results were close to what was obtained in the experiment (Table 8). The interannual variability was almost as high as for the summer cover crops with a minimum of 5.2 t DM.ha⁻¹ for the MSCC modality and a maximum of 9.7 t DM.ha⁻¹ for the ECC modality (Figure 31). The average was still higher than for the summer cover crops. The main limiting factor for the biomass production of winter MSCC and ECC seemed to be the temperature sum from February 1st (correlation coefficient of 0.85).

2.3.4.2 Water consumption and drainage

Repeating the trial over 30 different climatic years showed that the drainage in year 2020 was slightly above the average of the last 30 years for both parts of the trial (Figure 7c). In the “summer” part, MSCCs reduced drainage compared to BS by an average of 15% (i.e. 6 mm),

and ECCs by 34% (i.e. 14 mm). The cover crops slightly reduced evaporation compared to BS (by 5 and 10 mm on average) but water was mostly lost by transpiration (56 and 63 mm on average). Irrigation at emergence was systematic and rarely below 20 mm.

In the "winter" part, there were no differences between the BS and CC modalities (151 and 149 mm on average respectively). The MSCC and ECC slightly reduced the drainage compared to bare soil: -13% for the first one with -5 mm and -18% for the second one with -8 mm. Over the last 30 years we had a high inter-annual variability of drainage with some years at 0 mm and others at more than 300 mm (Figure 31). In addition, the transpiration of ECC and MSCC, which were quite similar, was higher than that of CC, while evaporation under these cover crops was reduced. In the end, evapotranspiration in the CC modality was almost equal to evapotranspiration in the bare soil modality and the same for evapotranspiration in the ECC and MSCC modalities with a few exceptional years.

2.3.4.3 Nitrate leaching and N₂O emissions

All summer and winter cover crops consistently reduced nitrate leaching over the 30 simulated years (Figure 7d). In the "summer" part, ECC and MSCC reduced leaching by one-third compared to bare soil, slightly less than in the experimental year, with an average of 35% of the initial mineral N stock leached with these modalities versus 55% with BS. The additional reduction in drainage by the summer ECCs compared to the summer MSCCs rarely resulted in an additional reduction in leaching. However, sometimes the reduction could be substantial: from 20 to 100% of leaching (Figure 31). The addition of digestate in July on the ECC and in March on the following barley did not seem to have any impact on leaching, although sometimes leaching was slightly higher with the ECC modality compared to the MSCC modality (e.g. 1996, 2008). In "winter" part, CC was more effective than MSCC and ECC. The average leaching was 44% of the initial mineral N stock with BS, 24% with CC, 31% with MSCC and 30% with ECC. The same hierarchy of modalities was found as in the field experiment year (Table 10). The digestate applied to the winter ECC in early March did not seem to have any impact on leaching.

Over the 30 simulated years, summer MSCCs consistently decreased N₂O emissions compared to bare soil and ECCs (with one exception in 2008; Figure 31). Unlike during the experimental year, the ECC modality did not always decrease N₂O emissions relative to BS. It increased them in 15 out of 30 years. Emissions were higher both during the presence of the ECC and during

the following barley, which received digestate inputs. On average, cumulative emissions over the year were 4.7 kg N-N₂O.ha⁻¹ in the BS modality, 2.8 kg N-N₂O.ha⁻¹ in the MSCC modality and 4.6 kg N-N₂O.ha⁻¹ in the ECC modality (Figure 7e). In the “winter” part, MSCC and ECC always decreased N₂O emissions compared to bare soil and CC modalities (Figure 31). The average difference between the MSCC and ECC modalities over the 30 years simulated was lower than in the simulated year of the field experiment: 3.8 and 4.2 kg N-N₂O.ha⁻¹ emitted on average, respectively. On average, the CC modality did not reduce emissions compared to bare soil: 5.9 kg N-N₂O.ha⁻¹ emitted in these two modalities. CC did reduce emissions during the fallow period but emissions were then increased from spring onwards due to mineralization of mustard residues.

2.4 Discussion

2.4.1 Predictive capacity of the model

The STICS model complemented the field data. The model results on drainage in the “summer” part were consistent with what could be predicted from field data. Soil analysis conducted on November 23, 2020, at the very beginning of the drainage period, showed significantly lower moisture content in the last soil horizon (60-90 cm) in the MSCC and even more in the ECC modalities compared to the bare soil modality. Consequently, the model predicted a decrease in drainage in the ECC modality compared to the MSCC modality, and in the MSCC modality compared to the bare soil modality. Similarly, drainage in the “winter” part was consistent with soil analysis conducted during the same period; there was no significant difference between modalities neither in the field nor in simulations. The model outputs for leaching were also consistent with the field data. In the “summer” part, SMNC at the beginning of the drainage period was lower in the ECC and MSCC modalities compared to the BS modality. Although the difference was not significant in the field, the model did simulate a decrease in leaching. In the “winter” part, before the drainage period, the CC had absorbed much more N than the other cover crops. As a result, the SMNC, especially in the 30-60 cm and 60-90 cm horizons, was depleted compared to the other modalities. The model had simulated a significant decrease in leaching. However, the difference in experimental SMNC between the BS modalities and MSCC and ECC was not significant, so we could not predict the difference in leaching between the BS modality and these last two modalities. The STICS model tested with cover crops under the

same experimental conditions as ours already reproduced well the dynamics of nitrogen and water in the soil (Plaza-Bonilla et al. 2015; Meyer et al. 2020) and even the fluxes of drained water and leached N when measured in the field (Constantin et al. 2012). We are therefore confident in these STICS-simulated results.

A strong limit of the STICS model was its incapacity to simulate species mixtures. This was particularly impactful for the rye–faba bean mixture of the winter 2021-22 that we could not reproduce. The simplification with monospecific rye cover crops made possible to reproduce the water dynamics and biomass production but the N dynamics were far from reality. Another limit was the lack of valorization of nitrogen applied with digestate by sorghum in summer cover crops. The model did not reproduce the difference in biomass between the MSCC and ECC modalities observed in the field. This can be explained by the fact that in the studies used to calibrate the N response of sorghum, the varieties used had low sensitivity to N (Lemaire et al. 1996; Barbanti et al. 2006). This species would deserve additional calibration work, knowing that to our knowledge the STICS model had never been tested on summer cover crops.

2.4.2 Cover crops biomass production

The simulated aerial biomass production of cover crops were quite variable from one year to the next. They ranged from 4 t DM.ha⁻¹ to almost 10 t DM.ha⁻¹ for ECC and MSCC and from 0 to 3.4 t DM.ha⁻¹ for CC. This is within the range of values that exist in Europe for catch crops and energy cover crops without reaching the highest values (Launay et al. 2022); the winter ECCs produced on average four times more biomass than the CCs. In contrast to the CC modality, the MSCC modality corresponds only to marginal situations in temperate cropland (Bergtold et al. 2017; Lamichhane and Alletto 2022; Kathage et al. 2022). It corresponds to the type of cover crops used in conservation agriculture to cover the soil as long as possible. These covers are supposed to provide ecosystem services other than the reduction of leaching, such as erosion reduction, carbon storage, nutrient recycling, provision of habitats and resources for animal biodiversity... (Scopel et al. 2013; Du et al. 2022). Apart from carbon storage which is starting to be remunerated (De Pinto et al. 2010), these services are not remunerated so the expensive cost of implementation is not compensated (Lamichhane and Alletto 2022). As a result, farmers are reluctant to use them (Lamichhane and Alletto 2022; Kathage et al. 2022). The payment of the biogas produced from the ECC would be a big incentive to implant these long cover crops.

ECCs must reach a certain level of production to be profitable. This level of production varies according to the biogas buy-back price, the cost of implementation and harvest of the ECC. In France, it is estimated to be between 5 and 6 t DM.ha⁻¹ harvested (from technical expert knowledge), knowing that there was 1 to 1.5 t DM.ha⁻¹ under a cutting height of 10 cm in the field experiment (Table 8). On our experimental site, we would have been profitable only 20 years out of 30 with winter ECCs and 16 years out of 30 with summer ECCs. Irrigation at emergence of summer ECCs and MSCCs was often necessary to exceed the threshold (without irrigation the threshold was exceeded only 8 years out of 30 by the ECC), but irrigation of summer cover crops is not always technically or legally possible. The cutting height could be adjusted to increase yield, but this would be done at the expense of soil carbon storage.

In our field experiment, digestate fertilization was not always well valorized by the ECC. On summer cover crops, the gain was visible in both years of the trial even with contrasting weather conditions. On the other hand, the effect on biomass was not visible on winter cover crops. In the second year of the trial, the application was clearly too late (only one month before the harvest of the ECC), but in the first year the application took place two months before the harvest. That year, about 10 kg N.ha⁻¹ was lost through volatilization, leaving 57 kg N-NH₄⁺.ha⁻¹ in the digestate. Our conditions were non-limiting in nitrogen, there was between 50 and 75 kg of mineral nitrogen left in the soil at the end of winter on the ECC and MSCC plots. The mixture used was not very responsive to fertilization because it contained faba bean and rye which is a cereal with low nitrogen requirements.

2.4.3 Impact on the main crop

In our field experiment, we did not observe any yield loss following summer or long winter cover crops despite slightly lower SWC and SMNC after harvesting/destroying MSCC and ECC. The only study we know of that investigated the impact of summer ECC on the yield of the following winter crop did not observe a difference in yield between the ECC, MSCC and BS modalities (Szerencsits 2014). However, late destruction of winter cover crops has been reported to cause pre-emptive competition for nitrogen or water, resulting in yield losses especially in dry climate (< 700 mm.yr⁻¹) (Justes et al. 2012; Abdalla et al. 2019; Garba et al. 2022). This was well predicted by the STICS model for sunflower in the simulations of the first year of the trial. But sunflower express a moderate drought tolerance and high plasticity (Debaeke et al. 2021), which was not showed by the model. Regarding nitrogen, the model-

simulated nitrogen nutrition index of sunflower was higher after ECC than after MSCC. Moreover, the SMNC measured on field was slightly higher after the destruction of the ECC than after the MSCC certainly due to the digestate input on the ECC. Finally, the model predicted less N immobilization from plant residues and digestate during the sunflower cycle after ECC than after MSCC burial (difference of 35 kg N.ha⁻¹). For the same destruction date, ECCs are therefore less likely to enter pre-emptive competition for N with the main crop than MSCCs. Marsac et al. (2019) previously observed this result in an experiment with a larger number of winter cover crop species comparing the ECC and MSCC modalities.

2.4.4 Environmental impact

2.4.4.1 Water cycle

The water used by summer cover crops did not contribute to the water table recharge during the following winter. MSCC reduced the drainage by 12 mm on average and ECC by 28 mm in the 30-years simulations. This could generate consequences on the water supply at the watershed scale if these cover crops become widespread. In addition, they required irrigation to emerge properly. This water problem is not specific to the experimental site. Because they are sown in dry period in France, it is difficult to consider obtaining profitable and stable yields without irrigation. Breeders could develop the drought resistance of cover crop species. Long winter cover crops did not systematically reduce drainage (this was not the case in our field trial, for example), but they did reduce it by 10 to 16 mm on average in the 30-years simulations, with a maximum of 50 mm in 2019-20. This was consistent with the results of the meta-analysis of Meyer et al. (2019) which found that cover crops reduced drainage, on average, of 27 mm. They also found a slightly greater reduction in drainage with summer-sown cover crops than with fall-sown cover crops like we did. Short winter catch crops did not have an overall impact on drainage. In fact, before the drainage period started in the simulated field experiment (in early December), the water consumption of the winter cover crops was not high enough to impact SWC; the reduction in soil water evaporation by the canopy was sufficient to compensate for its transpiration. Water use increased in early spring as the long cover crops started to grow again. When the precipitations were still abundant in spring, as it occurred sometimes in the 30-years simulations, the increased water consumption of long cover crops decreased drainage.

2.4.4.2 Ammonia volatilization

We measured ammonia emission factors of 6% N-NH₄⁺ after digestate injection in summer and 17% N-NH₄⁺ after surface application of digestate in winter. Studies that measured volatilization following land application of mono-fermented or co-fermented digestate from crop residues found emission factors ranging from 6 to 29% of applied N-NH₄⁺ (Launay et al. 2022). Direct injection of digestate is known to be a very effective practice to reduce volatilization (Wulf et al. 2002; Maris et al. 2021). In winter, despite surface spreading, we were in rather good conditions to limit volatilization: average temperature of 9-10°C, average wind speed of 3-4 m.s⁻¹, and cumulative precipitation over the 20 days following spreading of 24 mm the first year and 86 mm the second year. Therefore, our emission factor for surface applied digestate is not in the highest range of literature values. The simulations did not produce more results since we did not trust the model to simulate volatilization following raw digestate applications. Its performance in a previous study was not good on this point (Moinard 2021).

2.4.4.3 Nitrate leaching

Mustard was the most effective cover crop in reducing leaching. Its effectiveness, due to its very fast implantation and growing and its high nitrogen absorption capacity, is well known (Couëdel et al. 2018b). Winter ECCs are unlikely to achieve the same efficacy because they do not include species with this trait. Farmers prefer to mix species with higher potential for aboveground biomass production with less N needs. However, they still reduced leaching compared to bare soil without reducing drainage in the field experiment. Moreover, ECCs were as effective as the unfertilized MSCC. It has already been reported in the literature that a spring fertilized winter cover crop was as or more effective than an unfertilized cover crop destroyed earlier (Launay et al. 2022). Summer cover crops prior to a winter cereal are not required by the European Nitrate Directive to reduce nitrate leaching, yet they substantially reduced leaching in the field experiment and in the 30-years simulations. Winter cereals absorb little N early in their cycle, which had two consequences: i) a lot of leaching in the BS modality and ii) a low impact on barley yield of SMN depletion by MSCC and ECC. Thus, almost all of the SMNC in the early fall is likely to leach out. Organic fertilization in early summer on the summer ECC did not result in additional leaching; all mineral N and organic N mineralized during the summer months must have been taken up by the plants.

2.4.4.4 N₂O emissions

The MSCCs reduced N₂O emissions the most compared to BS in the simulation of the field experiment and in the 30-years simulations. They were effective in reducing the SMNC over time. CC, while more effective at reducing leaching by taking up a lot of N in the fall, did not reduce N₂O emissions at the succession time scale. The effect of cover crops on N₂O emissions is discussed in the literature (Blanco-Canqui et al. 2015; Kaye and Quemada 2017; Abdalla et al. 2019; Guenet et al. 2020). There is no consensus on the net effect of cover crops on these greenhouse gas emissions. The authors agree that N uptake during the cover crop cycle does decrease the SMNC and thus N₂O emissions (Blanco-Canqui et al. 2015; Abdalla et al. 2019). However, when studies follow the succession beyond the fallow period they observe under certain conditions increased N₂O emissions compared to a soil left bare during fallow (Blanco-Canqui et al. 2015; Kühling et al. 2022). The low C: N ratio of residues and their incorporation into soil accelerate the dynamics of residue mineralization and of N₂O emission risks. Mustard had a C:N ratio of 10 on average in our simulations, so it mineralized quickly after its destruction in November by releasing nitrogen too early for the following sunflower. The rye of the MSCC modality had an average C:N ratio of 24 in our simulations and was destroyed in early May. Their degradation provoked N organization after incorporation into soil and during the sunflower cycle and thus did not undergo nitrification or denitrification. The export of these high C:N residues in the ECC modality increased mineral N production in soil followed by nitrification and denitrification. In addition, the application of digestate twice a year in the “summer” crop succession and once a year in the “winter” crop succession brought a surplus of nitrogen compared to the other modalities. This surplus can also be the cause of the increased N₂O emissions compared to the MSCC modality and sometimes to the BS modality in the “summer” part. For example, Häfner et al. (2021) and Holly et al. (2017) observed a peak in N₂O emissions in the first few weeks after digestate application.

2.5 Conclusion

Our study compared the impact on water and N cycles and on the production of the following crop of energy cover crops versus bare fallows, catch crops and multi-services cover crops. In summer fallow, compared to BS, ECCs reduced leaching but not always N₂O emissions because of the high frequency of digestate return. For the same reason, they increased NH₄⁺ volatilization. Their impact on the water cycle was negative since they required irrigation at

sowing and reduced drainage the following winter. The summer MSCCs had a better environmental balance in the field than the summer ECCs since they greatly reduced N₂O emissions and reduced drainage less. These summer MSCCs and ECCs had no impact on the yield of the following winter crop. The winter ECCs reduced N losses by leaching and nitrification/denitrification compared to BS but increased NH₄⁺ emissions due to the return of digestate. They also slightly decreased drainage. As in summer, winter MSCCs decreased leaching in the same way as ECCs and decreased drainage less. However, with the lower frequency of digestate return, the winter ECCs were as effective as the winter MSCCs in reducing N₂O emissions. The winter ECCs and MSCCs did not impact the following crop in the field experiment due to the plasticity of the chosen species, but the soil N status after the cover crop was destroyed/harvested suggested that the following crop could be N-stressed, especially after the MSCC. The short CC tested in the winter fallow more effectively reduced leaching than the ECC without reducing drainage. However, it did not reduce N₂O emissions.

Learnings

After reviewing the literature, we hypothesized that ECCs would reduce N_2O emissions relative to cover crops left in place thanks to residues exportation. In this chapter, we have seen that there are many nuances to this assertion. It is true if the cover crop has residues with a low C:N ratio and are therefore rapidly exposed to nitrification and denitrification. It is no longer true if the residues have a high C:N ratio that leads to the organization of mineral N. Moreover, the fertilization of ECCs goes rather against this assertion since the digestate is rich in mineral N (more than the cover crop residues) and is therefore quite sensitive to nitrification/denitrification.

As expected, ECCs did reduce nitrate leaching compared to bare soil despite the additional fertilization. However, they did not reduce leaching more than other cover crop types despite the sometimes higher biomass production. The cover crop species chosen had a strong impact on this service.

In this experiment, ECCs did not reduce drainage much. The summer ECCs had more impact than the winter ECCs, probably with an effect of the amount of biomass produced. In winter, there was not much difference between MSCCs and ECCs, but these long cover crops had more impact than the short catch crop.

In this Chapter, we studied the field impacts of ECC AD on the N and water cycles. Unfortunately, we cannot use this field experiment to study the C cycle, as the changes in the SOC stocks are too slow. We therefore decided to couple this field experiment with a laboratory experiment that will allow us to characterize the impact of AD on the C and N of cover crop biomass and their fate in the soil. Chapter 3 will be strongly linked to Chapter 2.

Chapter 3 – Influence of cover crop anaerobic digestion on soil organic matter

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3.1 Introduction

Maintaining or increasing soil organic matter (SOM) stocks is useful both for climate change mitigation and for soil fertility (Lal 2004; Bünemann et al. 2018). Soils participate in climate change mitigation when they sequester C, i.e. when they remove CO₂ from the atmosphere, not only when soil organic carbon (SOC) stocks increase (Chenu et al. 2019). Raw SOM storage is useful for nutrient cycling, water retention, physical stability of the soil, and feeding and habitat for living soil organisms (Bünemann et al. 2018). Ultimately, storing C in degraded soils increases food production (Lal 2004). The stoichiometry of the soil microorganisms implies to bring N with the C (Bertrand et al. 2019). If the residues returning to the soil do not contain enough N available to the microorganisms in relation to the amount of C supplied, then the microorganisms will use the mineral N in the soil, i.e. provoke N organization. The depletion of mineral N from the soil by microorganisms can have a deleterious effect on crop production on the short term if they are unable to meet their N requirements.

Inserting cover crops into cropping systems is a practice known for its effectiveness in storing and sequestering C in the soil (Poeplau and Don 2015; Jian et al. 2020; Launay et al. 2021a). These cover crops are inserted between two main crops, mainly in winter fallow before a spring crop, and are neither harvested nor fertilized. The levels of biomass production per hectare are therefore rather low in Europe (Justes et al. 2012).

Recently, the use of cover crops to produce biogas by anaerobic digestion (AD) is developing in Europe (Szerencsits et al. 2015; Dale et al. 2016; Marsac et al. 2019). Using these cover crops does not cause spatial competition between food and energy use of crops. They are sown in summer or winter fallows and managed to produce more biomass than conventional cover crops (Launay et al. 2022). Since a large part of the carbon in the biomass is transformed into CH₄ and CO₂ during AD (between 43 and 80% ; Launay et al. (2022)), they return little C to the soil. But the remaining C is considered more recalcitrant than the C in the plant biomass because i) it is the fraction that was not degraded during AD and ii) the degradation of the labile fraction during AD resulted in the formation of more complex organic compounds (Béghin-Tanneau et al. 2019). In the end, the few studies that compared SOC storage with digestate return to direct return of plant residues for the same amount of initial plant biomass do not reach the same conclusions: Thomsen et al. (2013) conclude to a similar storage, Béghin-

Tanneau et al. (2019) conclude to more storage with the digestate, Levavasseur et al. (2023) conclude to less storage with digestate.

As the impact of AD of plant residues on soil organic matter storage is not clear, we explored the issue with the case of AD of energy cover crops. To quantify i) the amount of carbon potentially stored in the soil and ii) the amount of nitrogen made available by digested and non-digested cover crops, an experimental anaerobic digestion was conducted with the above-ground biomass of ECCs in the laboratory, then monitored the C and N mineralization dynamics of the plant biomass or the digestate during incubation in soil and controlled conditions. By fitting a mathematical model to the C mineralization dynamics, we estimated the amount of stable C in each substrate. Finally, we put these results into perspective with the biomass of cover crops produced in the field.

3.2 Material and Methods

3.2.1 Production of plant biomass

We used the cover crops grown in the field experiment described in Chapter 2. In summary, the experiment was divided into two parts, one with two summer fallows in the crop succession and the other with two winter fallows in the crop succession. In the summer fallow, an energy cover crop (ECC) modality was compared to a multi-services cover crop (MSCC) modality. The species sown were the same but the ECC was fertilized and exported unlike the MSCC. In winter fallow, the ECC and MSCC were also compared to a catch crop (CC) modality, i.e. a short, fast-growing cover crop destroyed and incorporated directly into the soil. After calibration with field data, the STICS model (Beaudoin et al. 2022) allowed us to estimate the amount of carbon in the biomass left in the field after the harvest or the destruction of the cover crops (stubble or the all above-ground biomass and roots) in all modalities.

3.2.2 Anaerobic digestion of plant biomass

The plant biomass of the summer cover crop mixture (96% sorghum– 4% vetch) harvested in 2020 was digested using mesophilic liquid anaerobic digestion in a pilot reactor at the INRAE Transfert Environnement. The plant biomass was characterized before digestion (dry matter (DM) content, C content, total N and N-NH_4^+ content, biochemical methane potential). After seeding the reactor with a suitable inoculum, only the above-ground biomass of the sorghum-

vetch mixture was used to feed the reactor at a flow rate of 9.55 kg fresh matter (FM).m⁻³.d⁻¹ to maintain a residence time of about 100 days. After 181 days of anaerobic digestion, the process was stable. The produced biogas were continuously measured during the process and the final digestate was characterized (DM content, C content, total N and N-NH₄⁺ content).

3.2.3 Kinetics of carbon and nitrogen mineralization in soils

The kinetics of organic C and N mineralization from initial plant biomass (sorghum–vetch mixture) and final digestate were measured during incubation of soil-organic substrate mixtures and compared to control soil alone in controlled conditions. The soil used was sampled in the surface layer of the field experiment. The incubations were conducted for 91 days at 28°C according to FD U44-163 standard. The plant biomass was dried and ground, but fresh digestate was used. At the end of these incubations, the carbon mineralization of the plant residues had not reached a plateau unlike the carbon mineralization of the digestate. In order to approximate the amount of long-term remaining carbon from both organic substrates, a three-compartment mineralization model (Lashermes et al. 2009) was optimized on the carbon mineralization curves. The first two compartments C_1 and C_2 (as percentages of total C supplied by residues) degrade according to first-order kinetics with different degradation rates (k_1 and k_2 in day⁻¹) while the last compartment $1-C_1-C_2$ remained in the soil:

$$C_{\text{mineralized}} = C_1 \cdot (1 - e^{-k_1 t}) - C_2 \cdot (1 - e^{-k_2 t}) \quad (1)$$

The model was fitted to the experimental results using the nls() function on R version 4.0.3.

3.3 Results

3.3.1 C and N fate of harvested and digested biomass

The mass balance of the AD of the sorghum–vetch cover crop is presented in Table 11. The biogas production was 117.42 m³ of biogas produced for one ton of substrate. Since biogas is composed of 56.38% CH₄ and 43.62% CO₂, the substrate expressed almost all the potential production of methane measured (67.20 m³ CH₄.t FM⁻¹). During AD, 66% of the biomass carbon was transformed into biogas and 36% of the organic nitrogen was mineralized. While the plant biomass contained only 0.6 g N-NH₄⁺.kg⁻¹, the digestate contained 1.9 g N-NH₄⁺.kg⁻¹.

Table 11. Mass balance (kg) of the anaerobic digestion of the summer ECC in a pilot reactor. The reference unit is the treatment of one ton of substrate.

	Plant biomass	Digestate
Fresh matter	1000	853.95
Dry matter	233.45	93.56
Total N	3.43	3.43
Organic N	2.83	1.81
NH ₄ ⁺ -N	0.60	1.62
Organic C	109.26	37.49
C : N (without unit)	31.9	10.9

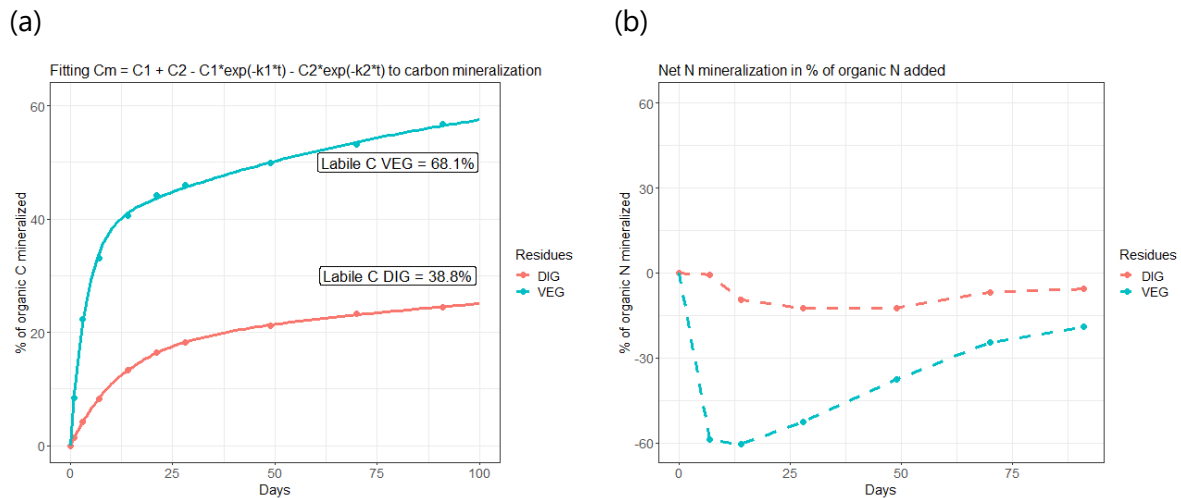


Figure 8. Net carbon (a) and nitrogen (b) mineralization dynamics of cover crop biomass (blue) and cover crop-based digestate (red) during 91-day incubation in the laboratory (dots) and fitting of a three-compartment kinetics model on the C dynamic (line).

The equations of the three-compartment kinetic models fitted to the organic carbon mineralization kinetics are for cover crop biomass (2) and digestate (3):

$$C_{\text{mineralized}} = 37.87 \cdot (1 - e^{-0.259 \times t}) - 30.20 \cdot (1 - e^{-0.0105 \times t}) \quad (2)$$

$$C_{\text{mineralized}} = 17.41 \cdot (1 - e^{-0.084 \times t}) - 21.36 \cdot (1 - e^{-0.0045 \times t}) \quad (3)$$

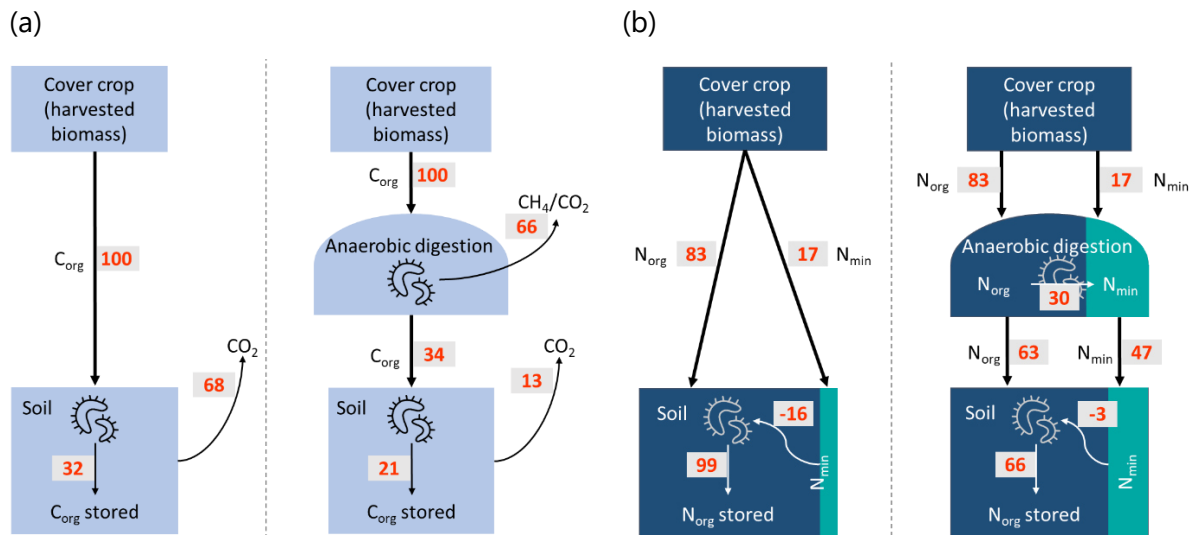


Figure 9. Carbon balance (a) and nitrogen balance (b) of direct incorporation of cover crops' exported biomass on the left and anaerobic digestion before incorporation on the right. Carbon fluxes are expressed as a percentage of the amount of organic C (C_{org}) harvested. Nitrogen fluxes are expressed as a percentage of the amount of total N harvested separated into organic N (N_{org}) and mineral N (N_{min}).

The models determined that 31.9% of the carbon of the cover crop biomass and that 61.2% of the digestate carbon would remain in the soil in the long term after incorporation (Figure 8a). The carbon balance including C degradation during AD and/or after incorporation into the soil is summarized in Figure 9a. It would result that the direct incorporation of the above-ground biomass of cover crops would store 32% of the carbon of the original biomass in the soil while only 21% of the initial C would remain in soil in the long-term after AD of cover crops then incorporation of the digestate in soil. We did not count the carbon losses during digestate storage.

After incorporation in soil, the degradation of cover crop above-ground biomass would be responsible for the organization of mineral N (Figure 8b). After 91 days of incubation, the equivalent of 18.8% of the plant organic N was still organized. The incorporation of the digestate also caused the organization of mineral N but to a lesser extent since after 91 days of incubation, 5.5% of the digestate organic N was organized. The N balance of AD and/or incorporation into the soil is presented in Figure 9b. Without considering the N losses during digestate storage and after digestate application, the digestion of plant biomass greatly increased N availability. The mineral N available in the field after one year (equivalent to 91

days of incubation in the laboratory) would be 1% of the total N of the cover crop residues if they were directly incorporated or 44% if they were digested before incorporation.

3.3.2 SOC storage expected in the field

Table 12. Above-ground biomass produced by each type of cover crop and quantity of carbon returning to the soil via aerial and root residues during the four fallows of the field experiment. Results of the simulation of the experiment on STICS.

		Aerial biomass t DM.ha ⁻¹	Aerial residues t C.ha ⁻¹	Roots t C.ha ⁻¹	Digestate t C.ha ⁻¹	SOC storage t C.ha ⁻¹
Summer 2020	MSCC	7.62	3.10	0.83	/	1.26
	ECC	9.23	1.26	0.95	0.85	1.23
Summer 2021	MSCC	3.91	1.64	0.56	/	0.70
	ECC	4.02	0.34	0.57	0.46	0.58
Winter 2020-21	CC	2.09	0.87	0.19	/	0.34
	MSCC	7.70	3.14	0.36	/	1.12
	ECC	7.93	1.13	0.37	0.72	0.93
Winter 2021-22	CC	1.75	0.87	0.17	/	0.33
	MSCC	8.29	3.35	0.41	/	1.20
	ECC	8.56	1.21	0.43	0.77	1.00

The biomass produced simulated by the model in the ECC and MSCC modalities was never significantly different despite the ECC fertilization (Table 12). The model estimated that one-third of the above-ground biomass C remained in place after harvesting the ECC. When added to the estimated roots C, the ECC would contribute 0.9–2.2 t C.ha⁻¹ to the soil (Table 12). It did not reach the amount of C provided by the MSCC (2.2–3.9 t C.ha⁻¹), even with the digestate return (0.5–0.9 t C.ha⁻¹), but exceeded the amount of C provided by the CC (1.1 t C.ha⁻¹).

If we had been able to return the digestate produced from the ECCs to the field we could have expected to store almost as much SOC over the long term as with direct incorporation of the MSCCs in the summer of 2020 (Table 12). In other years, direct incorporation of MSCC was more advantageous (+20% of SOC storage). The particularity of the summer of 2020 is that the ECC produced much more biomass than the MSCC that year (+1.6 t DM.ha⁻¹). Moreover, in winter fallow, the CC producing 3.8–4.9 times less biomass than the ECC would have stored 2.7–3 times less C.

3.4 Discussion

We found that 32% of the carbon in the sorghum–vetch mixture was stabilized in the soil. This was in the same order of magnitude as the 32–41% found by Justes et al. (2009) for 5 different cover crop species or the 28% found by Constantin et al. (2010) in long-term trials. After AD, only 21% of this carbon contributed to SOC stocks. Thus, we found that digested plant biomass did not contribute as much to the SOC stock as directly returned plant biomass. Previous studies using a methodology very similar to ours did not reach the same conclusions. Thomsen et al. (2013) found that incorporation of crop-based digestate stored 12% of the carbon initially contained in the plant biomass. Béghin-Tanneau et al. (2019) found that corn-based digestate stored 23% of the initial carbon. These figures obtained with the digestate were close to ours. The difference was therefore on the amount of C stored with their plant biomass before digestion: 14% of the carbon of the forage of Thomsen et al. (2013) and -4% of the corn of Béghin-Tanneau et al. (2019). In contrast to these two studies, a more recent study using AMG simulations found that AD of cover crops did not store as much C as if they were directly incorporated into the soil (Levvasseur et al. 2023a). According to their study, a digested cover crop would store more C than a cover crop left in place as long as it produced at least twice as much biomass. This threshold was a little lower in our study since with a difference of a factor of 1.2 we obtained almost the same storage with and without AD (summer 2020, Table 12). The digestate on their model was calibrated from Levvasseur et al. (2022b) who calculated the long-term remaining carbon of a sample of several raw cover crop-based digestates in the laboratory. It averaged 50% of the digestate carbon with a range of 29% to 67% depending on the digestate composition. With 61% of the digestate carbon remaining in the soil in the long term, our digestate is close to the average obtained by Levvasseur et al. (2022b). The high proportion of cover crops in our digestate explains why we are above average.

Because of their high C:N ratio (Justes et al. 2009), the incorporation of cover crop residues resulted in N immobilization. Incorporating these residues just before sowing the next crop could have a depressive effect on it. The peak of organization was reached after 14 days of incubation (i.e. between one and two months outdoors) with the immobilization of the equivalent of 60% of the organic N of the residues, i.e. approximately 56 kg N.ha⁻¹ in the field in the summer MSCC modality of 2020. Exporting these residues for AD avoids this N immobilization and thus the potential depressive effect on the following crop. Moreover,

transforming the residues into digestate made N more available for the crop that will receive it thanks to the mineralization of N during digestion and the lower N organization once applied. The first point had already been measured before (Möller and Müller 2012; Bareha et al. 2018) but the second had not yet been identified, studies having rather compared the direct incorporation of grass–clover leys (with very probably a low C:N) to their AD before incorporation of the digestate (Möller and Stinner 2009; Brozyna et al. 2013).

3.5 Conclusion

While the literature suggested that cover crop AD might not have an impact on SOC storage, we observed that AD of a sorghum–vetch mixture left less long-term stable C in the soil after the digestate was returned to the soil than after the direct return of the plant biomass. AD of cover crops can still allow for more C to be stored if the AD cover crop produces more biomass than in the practice prior to AD (e.g. no cover crop, short cover crop, unfertilized cover crop). AD of productive cover crops (i.e. high C:N grasses) also has the advantage of increasing N availability compared to direct incorporation of this type of cover crop.

Learnings

The results of the laboratory incubations on the dynamics of N mineralization following the incorporation of cover crop residues tend to confirm the results obtained on STICS in the field experiment in Chapter 2. STICS predicted a high N immobilization after the incorporation of rye biomass grown in winter fallow causing N stress on the following sunflower. This N immobilization was partially avoided if the rye was harvested for AD, leaving only stubble in place.

In terms of C storage, contrary to our hypothesis, AD of ECCs does not seem naturally beneficial. It is necessary to produce a much higher biomass with an ECC than with an unharvested MSCC.

The proportion of stable C in the cover crop-based digestate calibrated in the STICS model is very close to what we found with the 100% cover crop-based digestate. In the following chapters, we will keep the calibration of cover crop digestate made from the results of Levavasseur et al. (2022b) in further simulation of cover crop-based digestate.

After the analytical and experimental Chapters 2 and 3 that showed some advantages and disadvantages of the AD of ECCs, we will use these results in a real case study. We have seen in these two chapters that depending on the control used (e.g. bare soil, catch crop, cover crop with the same production than ECC but not harvested,...) and the practices implemented with AD (e.g. frequency of digestate input, choice of species,...), the results of the AD implementation could be better than those of the control or have negative impacts. We will therefore start from the practices of farmers in a given territory and simulate the likely introduction of AD in this territory in order to see how farmers' practices change and what the environmental impacts are.

Chapter 4 – Introducing anaerobic digestion in an area dominated by field crops: coupling soil–crop model and an anaerobic digestion model to estimate C and N fluxes

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4.1 Introduction

Currently, European (Directive 2009/28/CE on the promotion of the use of renewable energies) and French (Ministère de la transition écologique et solidaire 2020) public policies encourage the development of anaerobic digestion (AD) of organic waste, mainly to produce renewable energy while mitigating climate change (substitution of synthetic nitrogen fertilizers by digestates limiting the use of fossil fuels for their manufacture, reduction of emissions from effluent storage). These organic waste can be of agricultural origin such as livestock effluents or crop residues, of industrial origin with food industry wastes and of municipal origin with food wastes from households and restaurants.

However, in addition to energy production, a biogas plant can have many indirect effects, for example by modifying the organic products spread on soils. AD allows the treatment of organic waste other than livestock effluents and therefore brings exogenous matter into the farming system. For example, the French law requires that all individuals and all professionals will have a practical solution for sorting their biowaste at source in order to valorize it by January 1 2024 (LOI No. 2020-105 of February 10, 2020 on the fight against waste and the circular economy). These matters are sources of carbon (C), nitrogen (N), but also of other nutrients and potentially of contaminants. During digestion, organic C and N are mineralized in the form of CO_2 and CH_4 for C (between 20 and 95% depending on the substrate (Möller and Müller 2012)) and mainly in the form of NH_4^+ for N (between 15 and 82% (Bareha et al. 2018)). Part of the C is therefore used to produce a renewable energy source while N in mineral form is more available to the plants. Other indirect effects of AD are linked to its impact on cropping systems such as the modification of spreading periods, the exportation of crop residues that return to the soil only after AD, or the modification of crop succession to introduce energy cover crops (ECC) instead of short growing period catch crops (Möller 2015).

Introducing AD will then lead to changes in the C and N cycles that can have an impact on climate change through, for example, a modification of C stored in agricultural soils (Möller 2015; Levavasseur et al. 2023a), the substitution of synthetic fertilizers by digestates (Pastorelli et al. 2021; Grillo et al. 2021; Álvarez-Alonso et al. 2022) or the increase in N_2O emissions through nitrification due to high NH_4^+ inputs (Möller and Stinner 2009). The changes can also impact other environmental and agronomic issues. Indeed, there are two main issues concerning N within the farm that are the N autonomy, in relation with the substitution of

mineral synthetic fertilizers by N from digestate, and the control of N losses (nitrate leaching, ammonia volatilization, etc.). On the other hand, the soil organic matter content may be modified in relation with AD, thus the related agronomic and environmental services such as water retention, soil structure stability, supply of nutrients (Wiesmeier et al. 2019).

Thus, depending on the priorities of the territories, the development of AD could either combine climate change mitigation and energy production with the improvement of the N balance and soil quality of the farms, or gave priority to certain issues to the detriment of others. For example, only considering energy production would lead potentially to environmental impacts because of excess ammonia volatilization, decreased soil organic matter content, etc. To ensure a sustainable development strategy for AD, it is important to answer three main questions. First, can AD really reduce the use of synthetic fertilizers if AD is based on the insertion of fertilized ECCs in the cropping systems? What are the N losses of these cropping systems related with AD development? Can the influx of new matter into the biogas plant external from the farm increase SOC storage on farms using the digestate? To answer these questions, we took the case study of the Versailles plain in France, mainly a field crop area. Several projects of AD development are under study in this area. We set up a modeling chain to study the impact of each of these projects on the C and N flows in each type of farm in the area.

4.2 Material and Methods

4.2.1 The studied area and choices of representation

4.2.1.1 Description of the area

Table 13. Average characteristics of the surface horizon (0-30 cm) and depth of the two major soil types adapted from Dhaouadi (2014).

Soil	Clay (%)	Organic C (%)	C/N	CaCO₃ (%)	pH	Depth (m)
Clay-limestone	30	1.5	10.4	23	8.4	0.6
Deep loam	17	1.1	9.3	0	7.1	1.2

The Versailles plain is located in norther France, 15 km west of Paris. We considered here the area delimited by the 27 municipalities belonging to the “Association de la plaine de Versailles” in 2023, representing 232 km². The climate is degraded oceanic (Cfb in the Köppen

classification), with a mean annual temperature of 11.5°C and mean annual precipitation of 694 mm (Trappes climatic station 48°77'N, 2°01'E, 1981-2010). This region, bordered by large urban areas, contained 11,100 ha of agricultural land in 2021 according to the Land Parcel Identification System (LPIS) mostly on two main soil types. These soil types described by Dhaouadi (2014) are: i) deep loamy soils, decarbonated luvisols, located on the lower and upper plateaus, ii) clay-limestone soils, rendosols and calcosols, of varying depth. The average properties of the surface horizon and the average depth of these two main soil types are presented in Table 13. Deep loam soils are present on 48% of the studied territory while clay-limestone soils are present on 33% of the territory.

Most farms in the territory are devoted to field crops, occupying 86% of the agricultural land according to the LPIS in 2021. The remaining area is occupied by farms specialized in market gardening, arboriculture or livestock. The livestock farms are mainly horse farms and a large dairy cattle and sheep farm of 400 ha. The main crops in LPIS 2021 were wheat, rapeseed, barley and grain corn, which accounted for 39%, 15%, 14% and 8%, respectively. Corn is grown slightly less on the shallower clay-limestone soils. The analysis of crop successions from LPIS indicates mainly successions of one to several years of cereals in rotation with rapeseed and/or corn.

The farming practices of the territory have been well described by the data of the agricultural census of 2010 completed by various surveys of farmers from 2015 to 2018 (Moinard et al. 2021). Spring crops (spring barley, corn) are generally sown after full inversion tillage to a depth around 25 cm, while wheat and rapeseed can be sown either after full inversion tillage or after a shallow tillage. Crop residues are mainly buried. During long fallow period before spring crops, cover crops are almost systematically grown, in compliance with the European Union Nitrate Directive (91/676/EC). The farmer surveys indicate that white mustard is the main cover crop despite large variability (oat, legumes, mixtures of species, etc.). The cover crops are mostly sown at the end of the summer (end of August/beginning of September) and destroyed by mechanical work in the fall (from mid-November). The European Union Nitrate Directive (91/676/EC) also limits the spreading periods according to the characteristics of the fertilizers and imposes a maximal amount of 170 kg N ha⁻¹ per crop for organic fertilizers. N fertilization is mainly provided by mineral fertilizers, in particular ammonium nitrate and urea ammonium nitrate solution. Green waste compost is both the most used and the most produced

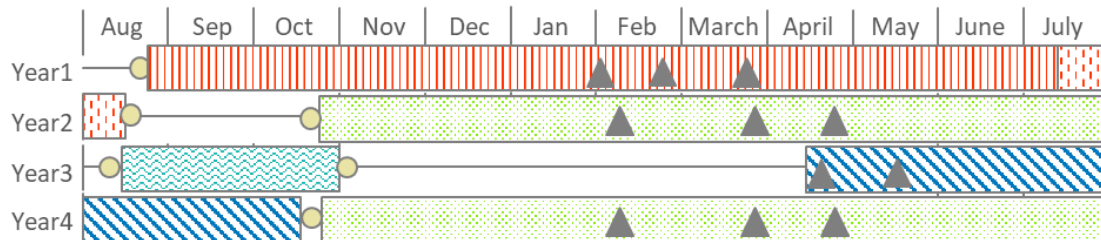
exogenous organic matter (EOM) in the Versailles plain (Moinard et al. 2021). It can be applied either on cover crops (before spring barley or grain corn), or before wheat and rapeseed. Horse manure is the second most used EOM, in the same conditions as compost (rates and insertion in the crop succession). Finally, some sewage sludge (produced on the territory or nearby) and dried pig manure (imported from Brittany), also called Humival, are also used. Both are mostly spread before rapeseed or before wheat for sludge. The large cattle and sheep farm has enough field surfaces to spread its own produced manure and slurry.

4.2.1.2 Representation of the area

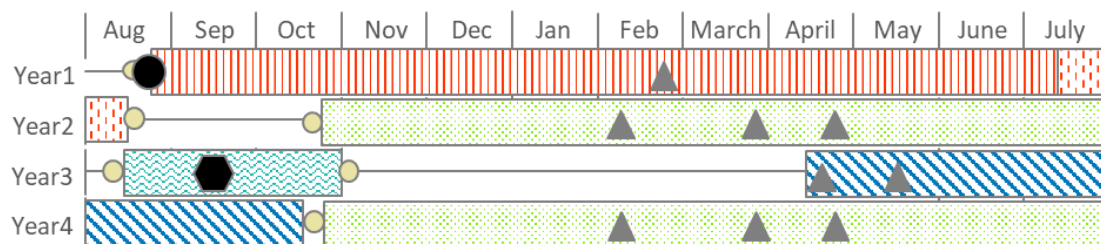
We did not have a spatially explicit representation of the area. We simulated each different cropping system. Since 66% of successions were rapeseed or corn with straw cereals, we focused in our study on a single crop succession: rapeseed-wheat-corn-wheat and simulated only this succession. This succession was used at 58% on deep loam soils and at 28% on clay-limestone soils. We only took into account these two types of soils, which we extrapolated to the area of field crops considered. Thus, the clay-limestone soil represented 32.5% of the simulated surface and the deep loam soil 67.5%. We created 4 typical farms from the practices identified in the area: i) field crop farms using only mineral fertilization, ii) field crop farms using green waste compost and dried pig slurry supplemented with mineral fertilizer, iii) field crop farms using horse manure supplemented with mineral fertilizer, iv) and one large dairy cattle and sheep farm using its own manure. The area occupied by these farms in the simulated region was limited by the amount of EOM available (Moinard et al. 2021). The cattle farm occupied 400 ha, the field crop farms using horse manure occupied 640 ha, the field crop farms using green waste compost occupied 3,400 ha and the rest, 3,360 ha, was occupied by field crop farms using only mineral fertilization. We chose ammonium nitrate as mineral fertilizer. The reference mineral N rate to reach the yields observed in the region for rapeseed was 180 kg N.ha⁻¹, for wheat 170 kg N.ha⁻¹ and for corn 160 kg N.ha⁻¹ (Dhaouadi 2014). We simplified the cattle farm compared to the reality by not considering the grassland area and by assimilating the sheep manure to the cattle manure. The choices of management practices for each farm type are presented in Figure 10 and the characteristics, doses and available amounts of the EOMs actually used are presented in the Table 14. The characteristics of cattle manure and slurry were measured on the cattle farm, those of Humival were obtained from Dhaouadi (2014) and those for horse manure and green waste compost were obtained from Moinard et

al. (2021). The proportion of stable C (i.e. which mineralizes at the same rate as SOC in STICS) in each EOM came from Levavasseur et al. (2022a).

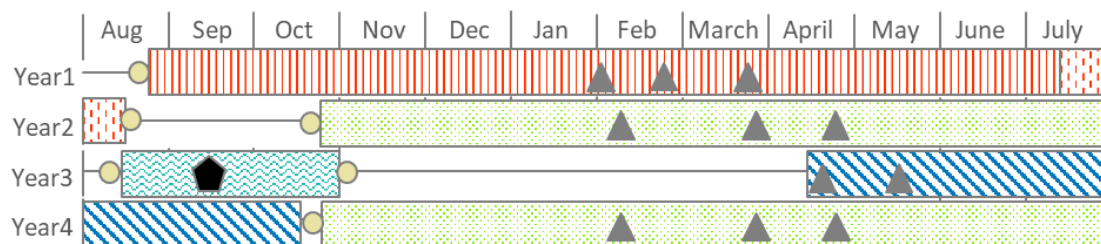
a) Field crop farms with mineral fertilization



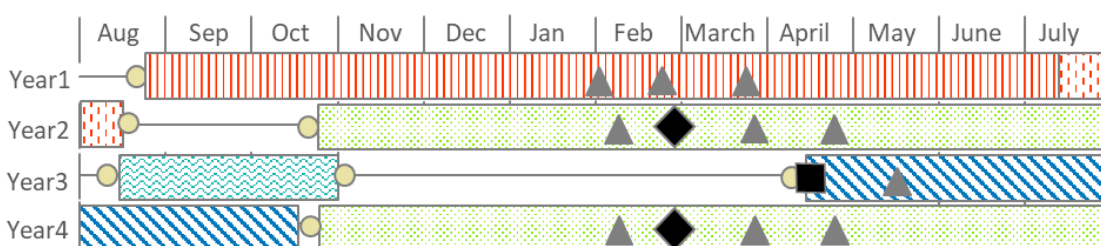
b) Field crop farms with green compost and dried pig slurry



c) Field crop farms recycling horse manure



d) Cattle farm



Legend:

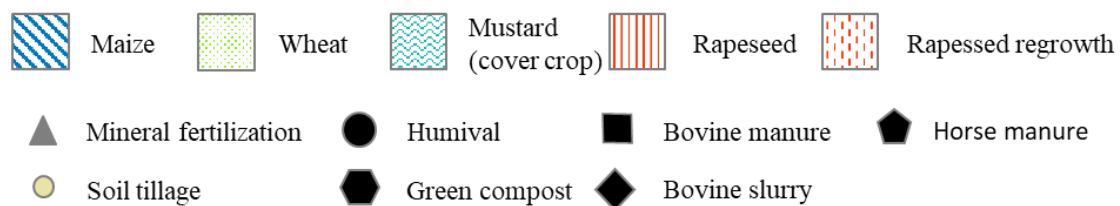


Figure 10. Schematic representation of the crop management practices defined for the dominant crop succession for each farm type.

Table 14. Exogenous organic matter (EOM) available quantities in the territory, doses and characteristics used in simulations. When the quantity is not limiting, the box is crossed out. The digestates correspond to potentially available EOM in case of the development of anaerobic digestion in the territory. Two different rates for ECC-based raw digestate are referenced in this table, the lower rate for exchange with ECC harvest and the higher rate for bovine manure and slurry substitution.

	Available quantity	Dose	C content	Total N content	Mineral N content	Organic C: total N ratio	Dry matter content	Stable proportion
	t or m ³	t.ha ⁻¹ or m ³ .ha ⁻¹	% of fresh matter	% of fresh matter	% of fresh matter		% of fresh matter	% of C content
Green waste compost	17 000	20	12.50	0.84	0.037	14.9	49.10	87
Humival	/	3.5	35.40	4.60	0.050	7.7	88.00	69
Horse manure	4 000	25	12.51	0.31	0.010	39.8	34.64	76
Bovine manure	2 800	25	8.23	0.51	0.020	16.2	17.76	76
Bovine slurry	5 000	25	4.67	0.58	0.150	8.1	11.06	58
Manure-based raw digestate	8 385	27	4.77	0.55	0.240	7.7	11.07	76
Manure-based solid digestate	774	7.5	13.77	0.97	0.140	14.1	31.26	76
Manure-based liquid digestate	7 540	25	3.17	0.49	0.240	5.4	8.16	76
ECC-based raw digestate	84 225	18.5; 37.8	3.07	0.44	0.210	6.9	8.27	64
Food waste-based raw digestate	/	25	0.44	0.61	0.350	0.7	1.90	76

4.2.2 Integration of anaerobic digestion

4.2.2.1 Three scenarios of anaerobic digestion

Three independent biogas plants projects are proposed in the area : i) the AD of food waste produced by the surrounding cities, ii) a biogas plant on the cattle farm that would only treat its effluents and brown water (recycled rainwater on outdoor paved exercise areas), iii) an alternative to this second project that would additionally treat horse manure and rely on the development of energy cover crops (ECCs) in the area. The plain of Versailles being close to the Parisian metropolis, the AD of the food waste could produce an important source of organic fertilizers for the farmers.

The second project concerns only the large cattle and sheep farm. This smaller-scale unit would enable the farm effluents to be converted into energy and the digestate would be returned to the farm fields. While livestock effluents are not necessarily stored with a cover, digestate must be covered since recently in France to limit ammonia losses (Decrees of November 10, 2009 and August 12, 2010 regarding the requirements applicable to anaerobic digestion facilities). We took this regulation into consideration in our scenarios. The potential phase separation of digestate leads to divide this scenario into two alternatives: one with phase separation and one without. In this project, there would be no exchanges outside the farm.

In the third project, the biogas plant would be larger and would use the effluents from the cattle farm, the effluents from the horse farms and the energy cover crops sown instead of the regular cover crops from the field crop farms. There would be exchanges of substrate for digestate with all the farms involved in the area. This scenario is therefore a territorial agricultural AD scenario.

4.2.2.2 Transformation of cropping systems and fertilization practices

Food waste AD

Field crop farms with raw food-based digestate

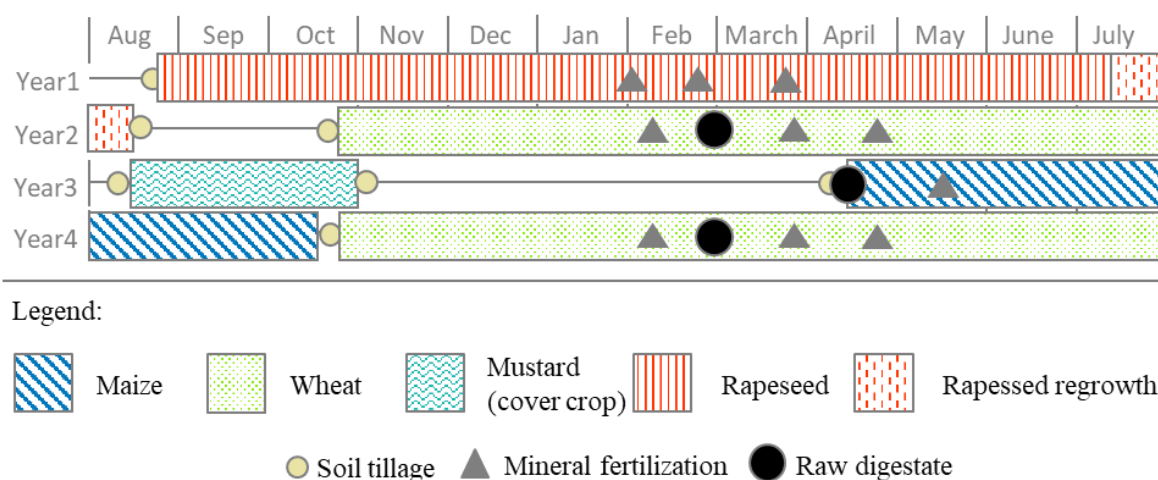
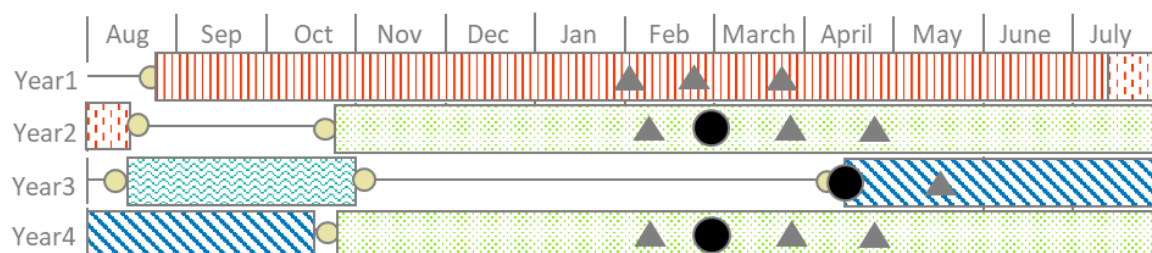


Figure 11. Schematic representation of the crop management practices of all the field crop farms modified by the food waste anaerobic digestion.

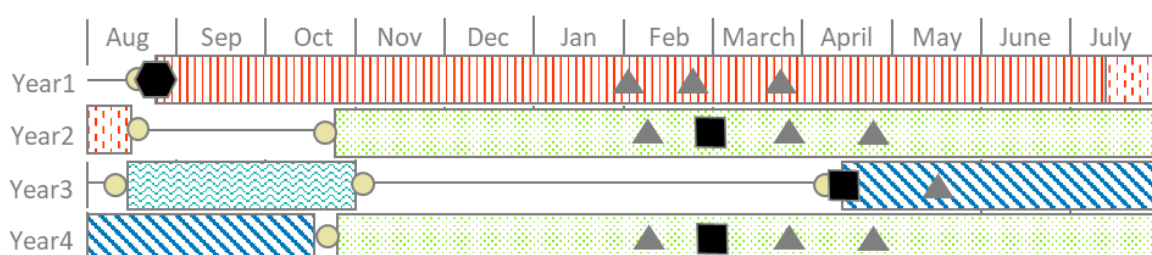
In the scenario of AD of food waste, we considered that all field crop farms would use the digestate produced in the region. This would be produced in non-limiting quantities given the population density around the Plain (Table 14). This scenario did not affect the cattle farm. We chose to replace all EOMs and part of the mineral fertilization by the digestate (Figure 11, Table 30). It should not lead to modifications of the main crop succession because all the crops in the rotation are suitable to receive the digestate. The raw digestate was spread in priority on wheat and corn, then on rapeseed. We applied average rates of $25 \text{ m}^3 \cdot \text{ha}^{-1}$ to meet the $170 \text{ kg N} \cdot \text{ha}^{-1}$ per crop limit of the nitrate regulation.

Cattle effluents AD with or without phase separation

a) Cattle farm with raw manure-based digestate



b) Cattle farm with solid/liquid manure-based digestate



Legend:

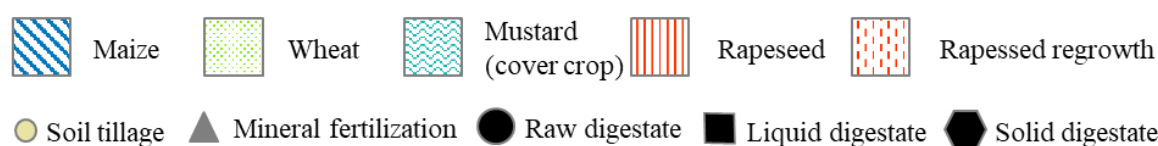
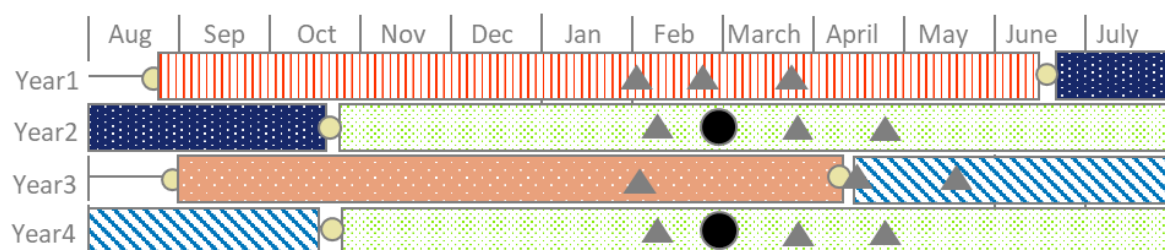


Figure 12. Schematic representation of the crop management practices of the cattle farm modified by the on-farm anaerobic digestion without phase separation (a) and with phase separation (b).

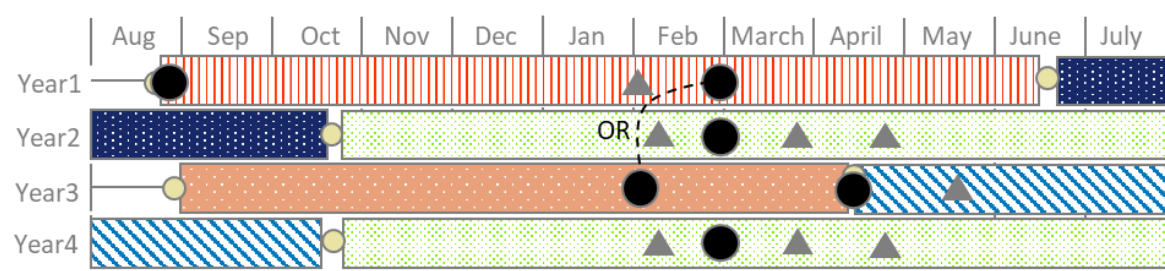
This scenario only concerned the cattle farm. The introduction of an on-farm biogas plant led to the replacement of manure and slurry by raw digestate (Figure 12a, Table 30). The application times did not change. Since the amount of digestate produced on the farm was higher than the amount of livestock effluent (due to the avoided loss of matter in effluent storage and the use of brown water, Table 14), the application rates were slightly higher than the manure and the slurry. In the scenario including phase separation, we calculated the doses of digestate in order to respect the solid digestate: liquid digestate ratio of 1:10 (Table 14). The solid digestate was spread at the sowing of rapeseed and the liquid digestate on wheat and corn (Figure 12b).

Animal effluents + energy cover crops AD

a) Field crop farms with ECC and raw territory digestate



b) Cattle farm with energy cover crops and raw territory digestate



Legend:



Figure 13. Schematic representation of the crop management practices of all the field crop farms (a) and the cattle farm (b) modified by the agricultural territorial anaerobic digestion.

In this scenario, all farmers in the area introduced ECCs in their succession (Figure 13). The crop succession allowed the insertion of a summer ECC after rapeseed and a winter ECC instead of the mustard cover crop (Figure 10). In the area, farmers engaged in AD mainly choose corn as a summer ECC because the water regime is favorable. For winter ECC, they choose among several species, mainly winter cereals. We also chose to test the insertion of a legume as ECC to bring N into the system as recommended by some authors (Stinner 2015). Since mixtures of winter cereal and legumes could not be simulated, we decided to consider faba bean on half of the surface in deep soil, the other half and the shallow soil being sown with oat. The ECC were sown and harvested as close as possible to the previous and following crops. The target harvested yield was 6 t DM.ha⁻¹ based on yields obtained in the region.

Regarding fertilization, the raw digestate replaced all EOMs and part of the mineral fertilization (Table 30) and we allocated the digestate coming from the territorial agricultural biogas plant between the cattle farm and the field crop farms in order to guarantee N equivalents between the substrates leaving the farms and the digestate entering the farms. Thus, the cattle farm first recovered the nitrogen supplied by its effluents, which allowed it to replace the manure and slurry inputs by $37.8 \text{ m}^3 \cdot \text{ha}^{-1}$ of digestate (Table 14). Then, the rest of the digestate was exchanged with the cattle farm and the field crop farms for ECC for a dose of $18.5 \text{ m}^3 \cdot \text{ha}^{-1}$ of digestate against a harvest of ECC, which exported about $70 \text{ kg N} \cdot \text{ha}^{-1}$ on average. The raw digestate was spread in priority on wheat and corn, then on rapeseed and oat. The faba bean and the corn ECC never received digestate (Figure 13), the first one because it was a leguminous crop, the second because of the short period between the harvest of rapeseed and the sowing of corn.

4.2.3 The modeling chain

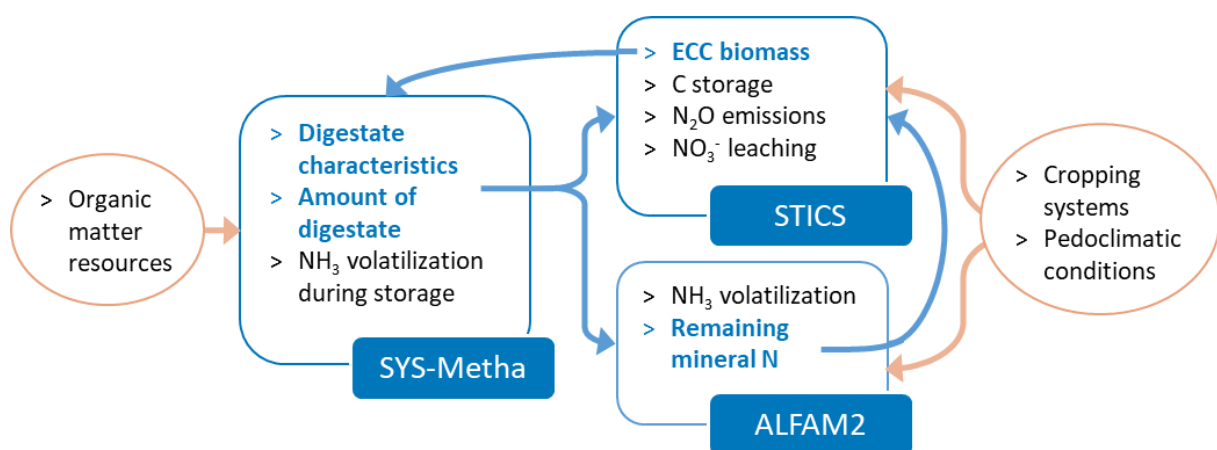


Figure 14. Schematic representation of the modelling chain. Models inputs are represented in the pink boxes and models outputs are represented in the blue boxes. The outputs written in bold in blue are also used as inputs to the other models by following the arrows.

We chose to simulate the different cropping systems with the soil-plant-atmosphere model STICS (Brisson et al. 2003). This model simulates crop growth in interaction with soil characteristics, daily weather and agricultural practices. In addition to crop productivity indicators, it provides output indicators related to the nitrogen cycle (nitrate leaching, N_2O emissions, amount of nitrogen uptake by plants, mineralization of nitrogen from humus, EOMs or residues), the water cycle (evapotranspiration, drainage) and the carbon cycle (root and

aerial carbon, soil carbon storage). It offers a wide choice of crops and has been validated in a wide range of soil and climatic conditions (Coucheney et al. 2015). Recent developments have improved the representation of carbon and nitrogen cycles by adding a dynamic description of nitrogen demand and root mortality (Beaudoin et al. 2022). The representation of EOMs has also been improved with a formalism change and a new calibration of a large number of EOMs including anaerobic digestion digestates (Levavasseur et al. 2022a, b). Since STICS was not able to correctly simulate ammonia volatilization following EOM application (Moinard 2021), we used the ALFAM2 model to estimate ammonia volatilization (Hafner et al. 2019). This model was calibrated on pig and cattle slurry but we also used it for bovine and horse manure, composts and digestates because we did not have a more complete model. Moinard 2021 also showed its good performances in simulating ammonia volatilization after digestate application. Simulations on the STICS model were performed over 30 successive years by repeating the same average climatic year for 30 years (the year 2014) to see the evolution of the SOC stock. We chose to use only one climatic year to be able to set the crop management upstream according to the climate, which the model does not allow to do during the simulation. The sowing and fertilization dates were set before the simulations according to the weather. Main crop yields were kept stable over the 30 years of simulations by adjusting mineral fertilization every 10 years; when the N supply of the soil increased, we reduced manually the amount of mineral fertilizer applied. In this way, we looked at the impact of AD on the amount of fertilizer used over time to maintain food production at its original level.

The additional step of the modeling chain in the AD scenarios was the SYS-Metha model (Bareha et al. 2021b) which predicted the characteristics of a digestate from the feedstock entering the biogas plant (Figure 14). By simulating the mass balance of mineralization of organic matter in a digester, this model is able to simulate the mineral nitrogen, organic nitrogen, dry matter and carbon content of a digestate depending on the characteristics of the substrates and the biogas plant (e.g., mean retention time, type of digestate storage). Substrate properties were provided by on-farm analyses for livestock manure and by STICS simulations for ECCs. For the latter, a first run without AD was necessary in order to estimate the potential production of energy cover crops that could be fed to the digester. In this preparatory run, the fertilization was entirely mineral. The digestate characteristics obtained by the SYS-Metha model for the two agricultural AD scenarios are presented in Table 14. For the food waste AD

scenario, we used the characteristics of a digestate produced not far from the study area (Table 14). The proportion of stable C in the digestates came from Levavasseur et al. (2022a) and Levavasseur et al. (2022b) for the ECC-based digestate.

We averaged the simulation results (i.e., C storage, NH_3 volatilization, NO_3^- leaching, N_2O emissions, mineral fertilizer consumption) either i) per hectare per year for each type of farm, soil type and scenario, or ii) cumulated over the entire area of the depicted territory per scenario in order to reconstitute the flows into and out of the territory. In the second case, the surface of the cattle farm and the surface of the field crop farms do not vary according to the scenarios. We considered in the N input fluxes the mineral fertilizers, the organic fertilizers (classic EOMs and digestates) and the symbiotic N fixation. In the N output fluxes, we considered N exported from the fields with crop biomass (main crops and ECCs), nitrate leaching, N_2O emissions, NH_3 volatilization from field and from manure/digestate storage. The difference between the N inputs and the N outputs would be N remaining in the soil.

4.3 Results

4.3.1 EOMs characteristics

The characteristics of the digestates differed from those of EOMs used in the area. The green waste compost, dried pig slurry and horse manure were very rich in C, more than the digestates. The compost also had a high proportion of stable C. The digestates, on the other hand, were much richer in mineral N. The raw manure-based digestate was less rich in C than the cattle manure but slightly richer than the slurry. The organic matter in such digestate is referred as more stable than slurry but not than manure (Levavasseur et al. 2022a). It contained more mineral N than the effluents, as expected with N mineralization during AD (Möller and Müller 2012). On the other hand, the liquid digestate was not richer than the raw digestate in mineral N, whereas the phase separation was supposed to concentrate the mineral N in the liquid fraction (Möller and Müller 2012). The food waste-based digestate was the most N-rich digestate while the ECC-based digestate presented the lowest N content among the digestates. The difference in N content between the manure and ECC-based digestates was probably because both cattle manure and slurry were richer in mineral N than the ECC and horse manure used in the territorial digestate and because this mineral N was conserved during AD (Bareha et al. 2018). These differences between digestates and conventional EOMs suggest that they

will not have the same impact on SOC storage and N availability; classical EOMs are expected to be more effective at storing C while digestates are expected to be more effective at substituting synthetic N fertilizers.

4.3.2 Carbon storage

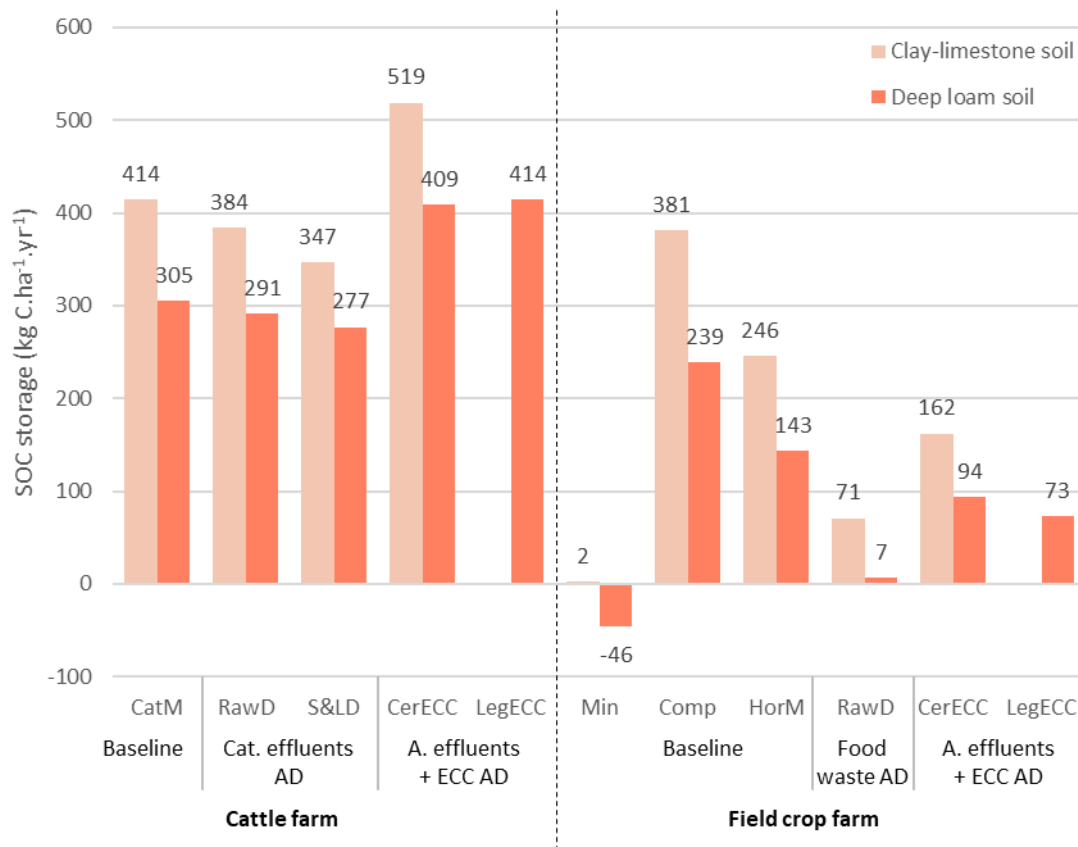


Figure 15. Average annual carbon storage (in kg C.ha⁻¹.yr⁻¹) over the 30-year simulation of the different scenarios for the cattle farm on the left and the field crop farms on the right. The cattle farm uses, i) in the baseline, cattle manure and slurry (CatM), ii) in the scenario of on-farm anaerobic digestion (Cat. Effluents AD), raw digestate (RawD) or liquid and slurry digestate (S&LD), iii) in the scenario of territorial agricultural anaerobic digestion (A. effluents + ECC AD), raw digestate on a succession where either cereal ECCs (CerECC) or legume ECCs (LegECC) are introduced. The field crop farms use, i) in the baseline, only mineral fertilization (Min) or green waste compost and dried pig slurry (Comp) or horse manure (HorM), ii) in the scenario of food waste anaerobic digestion (Food waste AD), raw digestate, iii) in the scenario of territorial agricultural anaerobic digestion (A. effluents + ECC AD), raw digestate on a succession where either cereal ECCs (CerECC) or legume ECCs (LegECC) are introduced.

4.3.2.1 On the cattle farm

In the on-farm AD scenario on the cattle farm, AD without phase separation slightly decreased SOC storage, by 30 and 14 kg C.ha⁻¹.yr⁻¹ on clay-limestone and deep loam soils, respectively (Figure 15). Overall, the digestate inputs in this scenario provided less carbon than the manure and slurry inputs (-122 kg C.ha⁻¹.yr⁻¹; Figure 32) because digestate contained less carbon than manure (Table 14) due to the release of CO₂ and CH₄ during AD. The increased C stability of digestate compared to slurry (Table 14) did not compensate for the losses. Separating phases of manure-based digestate accentuated slightly this soil C decrease (-37 and -14 kg C.ha⁻¹.yr⁻¹ for each soil type compared to raw digestate) because of higher losses during digestate storage leaving less carbon to return to the fields (-128 kg C.ha⁻¹.yr⁻¹ compared to raw digestate; Figure 32).

On the contrary, the territorial agricultural AD scenario increased strongly SOC storage by +100 kg.ha⁻¹.yr⁻¹ compared to the baseline cattle farm in both soil types (Figure 15). The ECC-based digestate had a lower C:N ratio than the manure-based digestate (Table 14), and the inputs were reasoned at equivalent N quantity, so for the same N quantity, the ECC-based digestate provided less C. Despite the provision of less C through the digestate, the ECCs remained crops more efficient to store C than the mustard initially grown in the baseline. They provided more C both through their roots (+137–190 kg C.ha⁻¹.yr⁻¹) and stubble remaining on site (+177–220 kg C.ha⁻¹.yr⁻¹) and through the additional inputs of digestate produced from their above-ground biomass (+54–62 kg C.ha⁻¹.yr⁻¹; Figure 32).

4.3.2.2 On the field crop farms

In the baseline scenario, field crop farms that used only mineral fertilization did not store SOC, or even destored C on the deep loam soil (+2 and -46 kg C.ha⁻¹.yr⁻¹, respectively; Figure 15). In contrast, farms that received horse manure or composts in the baseline situation stored SOC (+183–246 kg C.ha⁻¹.yr⁻¹ with horse manure and +239–381 kg C.ha⁻¹.yr⁻¹ with composts). Like on the cattle farm, SOC storage was greater on clay-limestone soils than on deep loam soils due to the higher clay and limestone contents (parameters that decrease the soil organic matter mineralization in STICS; Beaudoin et al. 2022). In the food waste AD scenario, raw digestate application was able to store carbon on farms that used only mineral fertilizers but was much less effective than compost or horse manure (Figure 15). It increased SOC storage by 52–69 kg C.ha⁻¹.yr⁻¹ compared to mineral fertilization but decreased it by 136–310 kg C.ha⁻¹

yr^{-1} compared to the other EOMs depending on soil type. Its very low C:N ratio resulted in low C contribution to the soil (Table 14, Figure 32).

In the territorial agricultural AD scenario, the introduction of ECCs into the succession along with the arrival of a territorial biogas plant resulted in more SOC storage than with food-based digestate or 100 % mineral fertilization, but still less than with compost or horse manure (Figure 15). It increased SOC storage by 66–160 $\text{kg C}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ compared to mineral fertilization and food-based digestate but reduced it by 49–219 $\text{kg C}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ compared to compost and horse manure. In this case, farms that used to apply horse manure and replace it by the digestate would decrease SOC storage. ECCs also returned C to the soil but less than composts and horse manure (-591– +54 $\text{kg C}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$; Figure 32). In addition, organic matter was returned in a less stable form than that of amendments, both from digestate and ECC roots and stubble.

4.3.3 Nitrogen flows

4.3.3.1 On the cattle farm

Table 15. Evolution of the synthetic fertilizer use, nitrate leaching, N_2O emissions and ammonia volatilization from fields ($kg\ N\ ha^{-1}\ yr^{-1}$) of the cattle farm during the first (0-10 yrs) and last (20-30 yrs) ten simulated years.

Soil type	Scenario	Fertilizer	Mineral N use		N leaching		N_2O emission		NH_3 volatilization	
			0-10 yrs	20-30 yrs	0-10 yrs	20-30 yrs	0-10 yrs	20-30 yrs	0-10 yrs	20-30 yrs
Clay-limestone	Baseline	CatM	133	105	19	46	1.5	2.5	21	19
	Cat. effluents AD	RawD	133	78	22	46	1.7	2.5	25	21
		S&LD	115	88	21	48	1.6	2.5	26	24
	Effluents + ECC AD	CerECC	80	58	3	12	1.1	2.0	30	28
Deep loam	Baseline	CatM	133	88	2	4	2.1	2.7	13	11
	Cat. effluents AD	RawD	118	68	1	4	2.0	2.8	18	16
		S&LD	108	78	2	4	2.0	2.8	19	18
	Effluents + ECC AD	CerECC	103	85	0	0	1.6	2.4	26	26
		LegECC	103	85	0	0	1.6	2.4	27	28

Anaerobic digestion of the cattle farm effluents

The AD of the cattle farm effluents, with or without phase separation, allowed a slight reduction in the use of synthetic fertilizers especially in the deep loam soil (Table 15). During the first ten years of simulation, the replacement of manure and slurry by raw digestate saved $15 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ of mineral fertilizer on the deep loam soil. Phase separation initially decreased the mineral fertilizer use compared to the use of raw digestate, by $18 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ on the clay-limestone soil and by $10 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ on the deep loam soil. These fertilizer savings were due to both the higher N availability in the digestate than in the manure and slurry, and the lower N losses during digestate storage than during effluent storage (Table 31). The savings increased over time (Table 15) as more N was stored in the soil with the use of digestates. The difference between the N inputs and outputs (i.e. N storage in the soil) the first ten years was slightly bigger with the use of raw digestate, they represented 32.2% of N inputs, than with the use of raw effluents (31.0% of N inputs) or solid and liquid digestate (31.1% of N inputs; Table 31). The last ten years of simulation, the difference in mineral fertilizer consumption between the initial situation and the use of raw digestate has increased (-20 and $-27 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ in the deep loam and clay-limestone soils respectively) while the phase separation of digestate has become less profitable than the use of raw digestate ($+10 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ compared to raw digestate; Table 15).

AD of cattle effluents had no impact in the short and the long term compared to the initial situation on leaching (Table 15) and N_2O emissions (Table 15), but ammonia volatilization after land application was higher (Table 15). The use of raw digestate increased volatilization by 2 to $5 \text{ kg N-NH}_3.\text{ha}^{-1}.\text{yr}^{-1}$ compared to the use of manure and slurry depending on the soil type and the period of application. Phase separation slightly accentuated this last point ($+0.2\text{--}3 \text{ kg N-NH}_3.\text{ha}^{-1}.\text{yr}^{-1}$ compared to the use of raw digestate). The liquid digestate applied to corn emitted more NH_3 than the raw digestate. With or without AD, N losses through leaching and N_2O emissions increased over time ($+2\text{--}27 \text{ kg N-NO}_3^-. \text{ha}^{-1}.\text{yr}^{-1}$ for leaching and $+0.6\text{--}1.0 \text{ kg N-N}_2\text{O}.\text{ha}^{-1}.\text{yr}^{-1}$ for N_2O emissions; Table 15).

Anaerobic digestion of ECCs and animal effluents

Including ECCs in the succession with the return of more digestate saved more fertilizer than the digestion of farm effluents only (53 and $33 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ compared to the situation without AD on the clay-limestone and deep loam soil respectively; Table 15). The cattle farm benefited

somehow from the N supplied to the biogas plant by the horse manure, the leguminous crops and the mineral N applied on ECCs in the field crop farms. However, the total amount of N entering the fields was higher than in the scenario of AD of farm effluents (116 t N.yr⁻¹ at the scale of the farm versus 101.6 and 97.3 t N.yr⁻¹ in the previous scenario with and without phase separation respectively; Table 31). In parallel, N exports were also higher with the export of ECCs. The mineral fertilizer savings increased over time but to a lesser extent compared to the initial situation: they increased by 18–22 kg N.ha⁻¹.yr⁻¹ over time in this scenario against 28–45 kg N.ha⁻¹.yr⁻¹ in the situation without AD (Table 15). In fact, fertilizer savings were already maximal at the end of the first ten years of simulation; corn and rapeseed did not receive any mineral fertilizer at the beginning of the simulations, and the dose of digestate quickly became too high for the rapeseed. On the other hand, mineral fertilizer inputs to wheat following the summer ECC increased significantly compared to the situation without AD on the deep loam soil (+50 kg N.ha⁻¹ at the beginning of the simulations).

The main crops exported less N in this scenario (results not shown) because corn suffered more water stress after winter ECC than after mustard, the latter being destroyed much earlier than the former. But the amount of biomass produced and N exported was still higher in this scenario due to the ECCs. The higher N inputs in this scenario were not necessarily accompanied by higher losses: at the farm scale, 12 t N.yr⁻¹ were lost from the fields as in the simple on-farm AD scenario, i.e. slightly more than the 10 t N.yr⁻¹ in the baseline scenario. On the other hand, volatilization losses from effluent storage were reduced by 8 t N.yr⁻¹ (Table 31). The differences between the scenarios were especially visible in the clay-limestone soil, which was more sensitive to volatilization and leaching (Table 15). In this scenario, lower leaching (-17–20 kg N-NO₃⁻.ha⁻¹.yr⁻¹ in the clay-limestone soil) compensated higher losses through ammonia volatilization (+5–9 kg N-NH₃.ha⁻¹.yr⁻¹ in the clay-limestone soil). N₂O emissions were also modestly reduced (-0.3–0.5 kg N-N₂O.ha⁻¹.yr⁻¹ in the clay-limestone soil). The higher mineral N inputs (both as mineral and organic fertilizers) were responsible for the increased volatilization at application. On the other hand, both inserting summer ECC and replacing short winter cover crops with long ones effectively reduced leaching when the soil was shallow. Similar to the other scenarios, volatilization changed little over time (Table 15) while N₂O emissions increased (+0.8–0.9 kg N-N₂O.ha⁻¹.yr⁻¹ in both soil types; Table 15). But unlike the

other scenarios, leaching increased little ($+0\text{--}10 \text{ kg N-NO}_3^-\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ in both soil types; Table 15); the effectiveness of ECCs at reducing nitrate leaching continued over time.

4.3.3.2 On the field crop farms

Table 16. Evolution of the synthetic fertilizer use, nitrate leaching, N₂O emissions and ammonia volatilization from fields (kg N.ha⁻¹.yr⁻¹) of the field crop farms during the first (0-10 yrs) and last (20-30 yrs) ten simulated years

Soil type	Scenario	Fertilizer	Mineral N use		N leaching		N ₂ O emission		NH ₃ volatilization	
			0-10 yrs	20-30 yrs	0-10 yrs	20-30 yrs	0-10 yrs	20-30 yrs	0-10 yrs	20-30 yrs
Clay-limestone	Baseline	Min	170	170	18	29	1.3	1.8	20	22
		Comp	134	93	23	40	1.5	2.2	18	13
		HorM	155	130	17	24	1.3	1.6	19	18
	Food waste AD	RawD	100	85	20	41	1.6	2.5	20	19
	Effluents + ECC AD	CerECC	170	145	3	9	1.2	1.5	28	27
Deep loam	Baseline	Min	170	170	1	1	2.0	2.3	10	11
		Comp	115	78	2	3	2.0	2.5	8	6
		HorM	158	135	1	1	1.9	2.1	10	9
	Food waste AD	RawD	100	65	1	2	2.2	2.6	15	14
	Effluents + ECC AD	CerECC	198	174	0	0	1.8	2.0	20	19
		LegECC	180	163	0	0	1.6	1.9	19	18

Anaerobic digestion of food waste

On field crop farms, since the beginning of simulations, the use of food waste-based digestate was very effective in reducing mineral fertilizer consumption (Table 16). The ten first years, on both soil types, it decreased the consumption by 70 kg N.ha⁻¹.yr⁻¹ on the farms previously using only mineral fertilizers, by 15–34 kg N.ha⁻¹.yr⁻¹ on the farms previously using green compost and dried pig slurry, and by 55–58 kg N.ha⁻¹.yr⁻¹ on the farms previously using horse manure. All farms using EOMs, whether manure, compost or food waste-based digestate, saved more fertilizer over time due to the storage of organic matter in the soil (Table 16). In the end, the farms using digestate were always the most sparing with mineral fertilizers, they used between 65 and 85 kg N.ha⁻¹.yr⁻¹ on the deep loam and clay-limestone soils respectively while the 100% mineral fertilizer control used 170 kg N.ha⁻¹.yr⁻¹. Even with the decrease in mineral fertilizer use and the increase in organic matter storage, this scenario increased the overall N inputs in the territory and on each farm type separately at the end of the 30-year simulation (Table 32).

Whether at the beginning of the simulations or at the end, nitrogen losses by volatilization, N₂O emissions, and leaching from the territory were overall higher with this scenario than in the baseline (+24–28 t N-NH₃.yr⁻¹, +1.6–2.1 t N-N₂O.yr⁻¹, +1–22 t N-NO₃⁻.yr⁻¹ at the beginning and at the end of simulations, respectively; Table 31, Table 32). The high mineral N content of the digestate made it very sensitive to volatilization, N₂O emissions, and the increase in N inputs compared to the initial situation increased the risk of leaching. The N losses slightly differed among the different farm types and soil situation. For example, leaching did not increase on farms that previously used compost and dried pig manure (Table 16). Volatilization only increased on the deep loam soil (Table 16). N losses through N₂O emissions and leaching increased more over time with the food waste AD scenario than in the baseline (Table 16, Table 32). This could be explained by the soil N storage the first years of simulation in this scenario compared to the baseline in farms using only mineral fertilization and in those using horse manure (Table 31). On the other hand, in all situations with organic fertilization, volatilization decreased with the decrease of total N inputs over time (Table 16).

Anaerobic digestion of ECCs and animal effluents

Changing from compost, dried pig slurry and horse manure to digestate produced mainly from ECCs increased synthetic fertilizer consumption by 15–36 kg N.ha⁻¹.yr⁻¹ on the clay-limestone soil and by 22–83 kg N.ha⁻¹.yr⁻¹ on the deep loam soil in the first ten years (Table 16). The

mineral fertilizer consumption did not increase compared to the mineral control in the clay-limestone soil but it did increase by 10–28 kg N.ha⁻¹.yr⁻¹ in the deep loam soil depending on the winter ECC species. With the legume as a winter ECC, the increase in fertilizer use was less than with the cereal (Table 16). The digestate provided slightly more total N than was exported by the ECCs, but winter oat as ECC needed to be fertilized (approximately 70 kg N.ha⁻¹) and the wheat following summer ECC needed to be more fertilized (+60 kg N.ha⁻¹ from mineral fertilizer compared to mineral baseline). The increase in synthetic fertilizer consumption was greater in the deep loam soil; the difference between the two soils could be explained by their different leaching behavior. On the clay-limestone soil sensitive to leaching, the ECCs (especially the summer ECC) strongly reduced leaching (from 13 to 20 kg N.ha⁻¹.yr⁻¹ the first ten years; Table 16). The saved nitrogen benefited the following wheat. Conversely, on the deep loam soil with very little leaching (Table 16), ECC did not provide this service. Beyond the difference between the two soil types, for an equivalent amount of N, the digestate was less effective on wheat following the summer ECC than the mineral N that could have remained in the soil instead of being absorbed by the ECC. Indeed, the N in the digestate returned partly in an organic form that could not be directly assimilated and partly in mineral form that was subject to volatilization. After 30 years, the farms decreased their fertilizer consumption but they would still consume more fertilizer than in their initial situation, except for those using only mineral fertilizers on clay-limestone soils which would decrease their final consumption by 25 kg N.ha⁻¹.yr⁻¹ (Table 16). These same farms on deep loam soils would consume about the same amount of fertilizer as in the baseline (Table 16).

As on the cattle farm, the corn following the winter ECC suffered from water stress, reducing its yield and thus the N export from the field. In the global balance, the additional N export by ECCs masked the decrease in corn yield (Table 31). The increase in N inputs on field crop farms in this scenario was accompanied by an increase in N losses compared to the baseline in the region: 182.5 t N.yr⁻¹ compared to 160.2 t N.yr⁻¹ in the baseline in the fields the first years (Table 31). The losses were nevertheless less important than in the food waste AD scenario (191.5 t N.yr⁻¹). In addition to the losses from the field, there were also increased losses from the storage of digestate: horse manure emitted 4 t N.yr⁻¹ during storage while 10 t N.yr⁻¹ could be attributed to the storage of digestate produced from it and from field crop farms' ECCs (Table 31). Going down to the field level, ECCs effectively reduced leaching and slightly reduced N₂O

emissions on all farm types (Table 16). The first years, they reduced leaching by 13–20 and 1 kg N-NO₃⁻.ha⁻¹.yr⁻¹ on the clay-limestone and deep loam soils, respectively and they reduced N₂O emissions by 0.1–0.3 and 0.2–0.4 kg N-N₂O.ha⁻¹.yr⁻¹. Since farms using green waste compost and dried pig slurry lost the most N₂O and NO₃⁻, changing fertilization and inserting ECCs was most effective on these farms. The overall increase in N losses was therefore due to a significant increase in volatilization on all farm types (Table 16). The higher N inputs, now partly in ammonia form, increased volatilization the first years by 10–12, 8–10, 9–10, 4–8 kg N-NH₃.ha⁻¹.yr⁻¹ compared to farms using green waste compost and dried pig slurry, to those using only mineral fertilization, to those using horse manure and to those using food waste-based digestate respectively. As in the other scenarios, volatilization slightly decreased over time with the decrease in N inputs (-0.3–1.5 kg N-NH₃.ha⁻¹.yr⁻¹; Table 16) and N₂O emissions slightly increased (+0.2–0.4 kg N-N₂O.ha⁻¹.yr⁻¹; Table 16). Unlike other scenarios, leaching did not increase in the deep loam soil (Table 16) and increased very slightly compared to others in the clay-limestone soil (+6 kg N-NO₃⁻.ha⁻¹.yr⁻¹; Table 16). At the end of the simulations, thanks to this evolution, the field crop farms of the territory did not lose much more N than if they had kept their initial practices: 192.5 t N.yr⁻¹ with the territorial agricultural AD against 190.5 t N.yr⁻¹ with the baseline, without counting the storage emissions that did not change over time (Table 32).

4.4 Discussion

4.4.1 Interest of coupling models

The coupling of the three models (SYS-Metha, STICS, ALFAM2) allowed us to evaluate the impact of the introduction of anaerobic digestion on the N and C fluxes at the plant biogas and farm level of a well-characterized region. Instead of using the characteristics of an average digestate, SYS-Metha allowed us to adapt the characteristics of the digestate to the substrates available in the area according to the two agricultural AD scenarios. Moreover, by injecting directly in SYS-Metha the outputs of STICS on the production and the characteristics of the ECCs, we preserved the C and N flows in the field-biogas plant system. STICS also had the advantage of simulating the impact of soil type, weather and agricultural practices on C storage, leaching and N₂O emissions. ALFAM2 allowed us to overcome the limitations of STICS on ammonia volatilization following organic fertilizer applications (Moinard 2021) at least for

the liquid and raw digestates and for the bovine slurry. However, the solid organic fertilizers (horse manure, cattle manure, green waste compost, solid digestate) are outside the scope of both STICS and ALFAM2 (Hafner et al. 2019; Moinard 2021). Our results on the volatilization of solid EOMs were therefore less robust than others but given their low ammonia N content, the error is probably small. Moreover, we did not have a more suitable model or an accurate emission factor for these products in the area. Regarding volatilization following mineral fertilizer applications, not all mineral fertilizers have the same sensitivity. We have chosen to use ammo-nitrate which is much less sensitive to volatilization than urea for example (Pan et al. 2016). The increase in volatilization with the substitution of mineral fertilizers by EOMs that we observed is therefore highly dependent on this choice.

4.4.2 Anaerobic digestion of food waste

The use of food waste-based digestate, in this study, had increasingly replaced the use of synthetic fertilizers over time. This scenario was the most effective in reducing the use of synthetic fertilizers on the territory because it was the only AD scenario that brought N from outside onto the agricultural fields from the studied territory. This N, currently incinerated, would therefore benefit the farmers. The high N content of food waste-based digestate is an asset for its use as fertilizer as shown by Álvarez-Alonso et al. (2022). However, its use caused also more nitrogen losses (gaseous and leaching) than synthetic fertilizers, compost and horse manure. Consistently, O'Connor et al. (2022) reported significant N losses from food waste-based digestate through volatilization (18–42% of total N applied) and leaching (0–25% of total N applied). Nicholson et al. (2017) also reported higher NH_3 and N_2O emissions from food waste-based digestate compared to compost. They also found significantly more leaching after digestate than compost when applied in the fall. However, in our study, digestate did not increase leaching more than compost due to two main factors that could explain this difference. First, the timing of application in the succession was adapted to the EOM: compost was applied at the end of summer while the digestate was applied in spring. Second, compost induced greater increase in soil organic matter and soil N mineralization in the fall just before the drainage period than food waste-based digestate, increasing the risk of N leaching.

Using digestates to profit from their fertilizing effect without increasing the impact of agriculture on the environment implies to spread them, in accordance with crop N need, in spring, to reduce the risk of leaching. If it could be buried before spring crops to also reduce

volatilization risk, it is not as easily possible for winter crops. The solutions for these crops could be, in decreasing order of effectiveness: i) the shallow injection, ii) the trailing shoes, iii) the bandspreading (Nicholson et al. 2017; Nicholson et al. 2018). For N_2O emissions, food-based digestate meets all the criteria favorable for N_2O emissions related with nitrification: it is rich in NH_4^+ and its C:N ratio is low, meaning a high availability for microorganisms. The solution that is considered by several authors could be the processing of the digestate before its application either to reduce its ammonium content, to adjust its C:N ratio, to adsorb ammonium, and inhibit the nitrification (Du et al. 2018; Sánchez-Rodríguez et al. 2018; Manu et al. 2021).

Because of its low C:N ratio, a particularity of food waste-based digestates (Table 14; Dutta et al. 2021, Álvarez-Alonso et al. 2022), it brought little carbon in the soil leading to little SOC storage. In our food waste AD scenario, we did not combine the use of digestate with the use of compost and horse manure. Compost with its higher stability and horse manure with its higher C:N ratio were on the contrary very efficient to store SOC as reported in the literature (Martínez-Blanco et al. 2013; Maillard and Angers 2014). If this food waste AD scenario were to develop, it would be interesting to keep the use of these EOMs in addition to the digestate to maintain the current levels of soil organic matter.

4.4.3 Anaerobic digestion on the cattle farm

Anaerobic digestion on the cattle farm allowed to save N mineral fertilizer in the first years while slightly increasing exports. Phase separation saved slightly more fertilizer than using raw digestate. The savings are explained by a decrease in NH_3 volatilization during effluents storage since in the scenario, according to French regulation, biogas plants must cover their digestate tanks, which is not the case for livestock effluents. This cover has a big impact on NH_3 emissions, as found experimentally (Sommer 1997; Clemens et al. 2006) and also reduces by almost 80% the greenhouse gas emissions (i.e. CH_4 and N_2O) if it is associated with a biogas recovery system (Malet et al. 2023). In terms of field emissions, AD of farm effluents did not increase leaching but increased ammonia volatilization. These two results are predominant in the literature and have been well described (Möller 2015; Launay et al. 2022).

Studies have investigated the increase in N bioavailability in digestate and found similar or better performance than undigested feedstock (Nkoa 2014). Cavalli et al. (2016) for example

found higher N recovery rates in corn and rye-grass with cattle slurry-based digestate than with undigested slurry. In our study, the decrease in N losses from effluent storage is the main reason for the decrease in synthetic fertilizer use. But it is possible that the increase in N bioavailability found in the literature explains a small part of the savings.

The results with and without phase separation were quite similar while usually, phase separation increases the concentration of mineral N in the liquid phase, thus increasing the availability of N (Nkoa 2014). Phase separation did save some fertilizer initially, but less in the last years because it reduced C and N storage a little. It also slightly increased volatilization contrary to what was observed by Moinard (2021). In their experiment, the lower dry matter content of the liquid digestate than that of the raw digestate enabled a faster infiltration into the soil and therefore reduced volatilization. In our experiment, part of the liquid and the raw digestates was directly buried, canceling a potential effect of the infiltration speed rate of the digestate.

On-farm AD slightly destored SOC compared to the baseline: by 5-7% and 9-16% without and with phase separation respectively, depending on the soil type. In the literature, Thomsen et al. (2013) did also predict a slight reduction of SOC by 14% but Wentzel et al. (2015) found only one case out of five in accordance with us. However, in our simulations, the soil organic N was maintained since AD allowed more N to be stored in the soil leading to increased fertilizer savings over time. Consistently, Wentzel et al. (2015) also found that digestate applications decreased the soil C:N ratio in contrast to raw slurry applications.

4.4.4 Anaerobic digestion of livestock effluents and energy cover crops

4.4.4.1 At the regional scale

Overall, the insertion of a territorial agricultural biogas plant increased N inputs to the territory by 25% (mineral, organic fertilizers and symbiotic fixation combined) and N losses by 12% the first years compared to the baseline. Some of these additional inputs were recycled by the ECCs in the digestate. At the end of the 30 years, N inputs were still increased but N losses were almost the same as in the baseline. SOC increased 12% less per year compared to the baseline. This lower C increase was mainly due to the replacement of green waste compost by digestate. However, we could have kept the use of green waste compost that is both compatible and

complementary to the use of digestate to better maintain C storage. The consequences of territorial AD varied within the farm types. Using horse manure in the biogas plant rather than in some farms would benefit all farms collecting digestate, except those that used to apply horse manure directly. As a result, the cattle farm reduced its mineral fertilizer consumption (with the exception of the first year when a mineral fertilizer was necessary on the ECC to start the production) and stored additional C. Field crop farms using only mineral fertilizers stored C but did not necessarily reduce their fertilizer consumption. Synthetic fertilizer consumption increased when pre-emptive competition for N between ECCs and next crops was strong. Farms previously using horse manure increased their mineral fertilizer consumption and destored C. By losing the horse manure, they lost an EOM that allowed storing SOM and therefore providing N in the long term instead of mineral fertilizers. The digestate they received was not really a substitute for mineral fertilizers since it simply compensated for the additional N consumption of ECCs.

4.4.4.2 ECCs impacts

The insertion of ECCs in cropping systems tended to increase SOC storage compared to the baseline, except on farms that used to apply horse manure and compost. Despite C losses during AD, C inputs were higher on average over the 4-year succession with a summer and winter ECC than with a winter catch crop, due to the much higher biomass production of ECCs (15–19 t DM.ha⁻¹ aboveground and 2.3–2.9 t DM.ha⁻¹ belowground in cumulative) than of catch crop (2–3 t DM.ha⁻¹ aboveground and 0.5–0.7 t DM.ha⁻¹ belowground). Levavasseur et al. (2023) estimated that for an equivalent biomass production, soil C storage was higher when cover crops were directly buried rather than applied back to soil after AD. However, once a certain production differential was reached (at least twice as much biomass), the digestion of the cover crops no longer destored SOC. Here, the differential between our catch crops and ECCs was more than sufficient (difference of at least a factor six). The ECCs yields were in line with what was observed in the field (Marsac et al. 2019). The summer corn ECC sown in deep loam soil was the ECC that produced the most biomass (about 10 t DM.ha⁻¹). This corn benefited from a large soil water capacity and sufficient rainfall (39 mm in the 10 days following sowing), which would not have been the case everywhere in France. Corn simulated under the same conditions but on clay-limestone soil produced 3 t DM.ha⁻¹ less.

The insertion of ECCs in the cropping systems increased N inputs to compensate for their N exportations, including mineral fertilizers and N_2 fixation. Moreover, mineral fertilizer inputs increased on wheat following summer ECC on all farm types, on deep loam soils. On this soil, the summer ECC uptook a lot of soil mineral N that would have been available for the following wheat. On the clay-limestone soil, N uptake by the summer ECC would have been leached and lost for the following wheat anyway. Szerencsits (2014) found no impact of the summer ECC on the yield of the following winter crop. Yet they observed the same level of leaching as those we observed on the clay-limestone soil (20-30 kg N.ha⁻¹.yr⁻¹ in situations without ECCs). The effectiveness of ECCs in reducing N leaching, including summer ECCs, had already been observed by various authors (Launay et al. 2022). Oat harvested in spring was even slightly more effective than mustard destroyed in fall to reduce leaching in our results. The corn following the winter ECC was also affected due to water stress. Since the corn was not irrigated in this area, the water stress impacted yield. Studies on long winter cover crops showed risks of water and nitrogen stress on the following crops (Alonso-Ayuso et al. 2014; Blanco-Canqui et al. 2015) but these risks have not been demonstrated experimentally for winter ECCs in Europe on the few studies available with late termination date of cover crops (Launay et al. 2022). If the period of ECCs harvest, i.e. April-June, were to become drier with climate change, the risk of water stress during the vegetative phase of the following crop would become higher, with potential effect on crop yield.

4.4.4.3 Contribution of the legume ECC

The introduction of a legume as a winter ECC on a part of the surfaces added 87 t N.yr⁻¹ in the territory via the symbiotic fixation of N_2 , which represented between 50 and 60% of total N in these ECCs. Knowing that 87% of the plant N was exported in the biogas plant, the symbiotic fixation brought almost 20% of N that fed the biogas plant. This was more than the horse manure and the effluents from the cattle farm combined. Legume ECCs are therefore interesting in these cropping systems since it does not required mineral N fertilizer and provides N into the system. However, their yield was lower than that of oat; it exported 4–5 t DM.ha⁻¹ against 5–7 t DM.ha⁻¹ for oat on the same soil type. Combining those two species in an ECC mixture could benefit from both N inputs through symbiotic fixation as compared to the non-legume and higher yield than the legume. In fact, Dordas and Lithourgidis (2011) and Dhima et al. (2014) obtained better yields with the mixture than with the faba bean sole crop

but not as well as with the sole crop oat. The yield of the mixture was better when the faba bean–oat ratio was 25:75 than when the faba bean increased in the ratio. This was not the case in these studies, but mixing species on the same area may increase the biomass produced compared to the same species grown on separate areas. Thus, the ECC mixture could be optimized and gave better results by choosing the adapted species. For example, Dordas and Lithourgidis (2011) had better yields with triticale–faba bean combinations than with oat–faba bean. Using spring faba bean as summer ECC, Fan et al. (2006) found in a colder and drier climate than France that corn associated with the faba bean facilitated the growth and the nitrogen fixation of the latter without decreasing its own yield.

4.4.5 Anaerobic digestion in general

In light of our results on the three scenarios of introduction of AD in the region, some general lessons could be drawn. First, the digestate produced by the biogas plants is highly dependent on the incoming substrates (Guilayn et al. 2019). Thus, the food waste-based digestate was much richer in organic and mineral N with a much lower C:N than the manure-based and ECC-based digestates. The impact on N and C fluxes on the fields was therefore different.

Secondly, the consequences of AD implementation varied with the scale of C and N fluxes within the farm or including exchange outside the farm. When only animal effluent were digested in the on-farm AD scenario, i.e. with no external input nor export of matter outside the farm, there was little changes in the N and C balances: the farm stored slightly less C because of the biogas production and consumed slightly less fertilizer because it lost slightly less ammonia throughout the effluent treatment chain. On the other hand, when we moved to territorial AD, either non-agricultural (i.e. AD of food waste) or agricultural (i.e. AD of livestock effluents and ECCs), the exchanges of matters between the farms and between the farms and outside the farms were modified and, depending on the case, had a strong impact on the farms' N and C balances. The cattle farm would benefit from territorial agricultural AD scenario. Inserting ECCs to produce more biogas rather than treating only its livestock effluents would store more C, save a little more mineral fertilizer, reduce leaching and N₂O emissions but increase ammonia volatilization in field. On field crop farms, however, changing fertilization practices was not always beneficial; the digestate did not provide the same services as the initial organic amendments used, nor the mineral fertilizer. These negative impacts must be weighed against the fact that the introduction of ECCs allows for the production of biogas,

which replaces fossil gas and therefore reduces greenhouse gases (GHG) balance. A global GHG assessment or even an LCA study such as Esnouf et al. (2021) can provide a perspective on these impacts in the field. It is also possible that territorial agricultural AD led to competition for organic matters, since farms that used to supply themselves with horse manure no longer have access to it. If they want to continue to be supplied with EOMs, they can choose to be supplied by the territorial biogas plant, but must comply with the new conditions of the biogas plant manager. On the other hand, they would no longer need to buy dried pig slurry imported from another region that was a cost to the farm.

Table 17. Advantages and disadvantages of each scenario for farmers compared to the current situation.

	Food waste AD	Cattle farm AD	Territorial AD	
	Field crop farms	Cattle farm	Field crop farms	Cattle farm
Advantages	<ul style="list-style-type: none"> • Less synthetic N fertilizer used • C storage (on 100% N synthetic farm) 	<ul style="list-style-type: none"> • Less synthetic N fertilizer used • Biogas income • Air quality improvement 	<ul style="list-style-type: none"> • No more organic fertilizer purchased • Biogas income • C storage (on 100% N synthetic farm) • Less leaching • Less N₂O emissions 	<ul style="list-style-type: none"> • Less synthetic N fertilizer used • Biogas income • C storage • Air quality improvement • Less leaching • Less N₂O emissions
Disadvantages	<ul style="list-style-type: none"> • No extra income • SOC lower increase (if the use of territorial EOMs is stopped) • Air pollution • More N₂O emissions 	<ul style="list-style-type: none"> • SOC lower increase 	<ul style="list-style-type: none"> • SOC lower increase • Air pollution • More synthetic N fertilizer used 	

4.5 Conclusion

The modelling chain used in this study allowed us to fully answer our research questions, which concerned the modification of N and C flows in a region after the introduction of AD. We explored three development scenarios for AD: AD of food waste, on-farm AD of livestock effluents, territorial AD of livestock effluents and ECCs. Both scenarios of agricultural AD have the advantage for the farmers to get them the income of the biogas. Concerning the substitution of mineral fertilizers, the scenario of food waste AD was the most effective (-65% of mineral fertilizer in 100% mineral farms after 30 years). On the other hand, in the territorial AD scenario, introducing whether fertilized or not ECCs in the succession cancelled out these gains or even increased the use of mineral fertilizers on the field crop farms. The other disadvantage is that they reduced the yield of corn by 5 to 10%. In exchange, ECCs greatly reduced nitrate leaching and moderately reduced N₂O emissions that could be increased by the use of food waste-based digestate. The legume ECCs still contributed a significant amount of atmospheric N to the biogas plant since it represented 20% of the N ration. All AD scenarios increased ammonia volatilization in the field. On the other hand, the introduction of AD with a digestate cover on the cattle farm strongly decreased the volatilization by 80% during storage, making the farm balance better with AD. Finally, in terms of soil carbon storage, we can conclude that the digestates used had a better fertilizing effect than amending effect. Digesting livestock manure decreased SOC storage of the cattle farm by 0.8 to 3.5% in 30 years. In contrast, introducing ECCs had a positive impact on SOC storage on this farm (+5.5 to 6.2%) and on field crop farms that did not use EOMs (+8.5 to 10.7%). Keeping green waste compost, which is not digested, in the territorial AD scenarios would have maintained the C stocks in the other field crop farms. Thus, in this region of the plain of Versailles, AD can be an asset to reduce the use of mineral fertilizers, but the flows of C must be considered so as not to damage certain lands.

Learnings

This chapter focused on the impact of introducing AD in a region dominated by field crops. The issue was therefore broader than that of ECCs. Nevertheless, we can draw some conclusions about them.

The insertion of two ECCs in a succession instead of one CC increased the return of C to the soil despite the production of biogas and stored much more C.

Summer ECCs greatly reduced leaching under the following wheat. Winter ECCs (barley) reduced leaching a little compared to CCs (mustard). Only one climatic year was tested. During that year, heavy rainfall between late April and early June caused additional drainage in the spring. As the CC was destroyed on October 31, the N of its residue mineralized in the spring was leached. The water consumption of the ECC from the end of winter avoided this second period of drainage.

The insertion of ECCs tended to increase mineral fertilizer use on field crop farms that used only mineral fertilization with a soil type effect. On the soil with low leaching susceptibility, the summer ECCs caused N stress on the following wheat that was compensated by increased N fertilization resulting in the overall increase in mineral fertilizer use. Thanks to the increase in soil organic matter, this increased consumption decreased slightly over time and sometimes fell below the consumption of the baseline.

The storage and spreading of digestate produced from ECCs increased ammonia volatilization compared to a situation without ECCs with 100% mineral fertilization. The increased use of mineral fertilizers also increased volatilization.

Despite the overall increase in N inputs to the fields, N₂O emissions were reduced with the insertion of ECCs probably because the increase in N exports allowed a lower soil mineral nitrogen content to be maintained throughout the succession.

In addition to the N stress sometimes caused by summer ECCs on the following crop, winter ECCs caused water stress on the following corn, which resulted in yield loss due to absence of irrigation in this region.

Now that we have developed a methodology to simulate the introduction of ECCs in a region, we will use it to study their impacts on the whole of France. By moving to the scale of France,

we will be able to study a greater diversity of soils, climates and crop successions. Moreover, here we simplified the study by using only one climatic year, in the following chapter we will use real climatic series. The comparison of the different AD scenarios highlighted the complementarity of the contributions of different EOMs. At the scale of France, in order to avoid the different biases that could result from the inclusion or exclusion of a given organic matter, we will only digest the ECCs. For the moment, we have examined the impacts on the N, C and water cycles but we have not taken into account the energy production by ECCs in the environmental impacts. However, this is an important point that we will integrate into a GHG balance taking into account several items ranging from the production of ECCs to the injection of methane into the network.

Chapter 5 – Potential production of energy cover crop in France: consequences on food crop production and environmental impacts based on scenarios simulation at high resolution

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5.1 Introduction

Anaerobic digestion (AD) is currently being developed in Europe to produce renewable energy such as gas to replace fossil gas. It is also opportunity for food industries and cities to treat their agro-industrial waste and biowaste. Indeed, selective collection of biowaste and biological treatment of biowaste from inhabitants, food stores and restaurants will be mandatory in France on January 1, 2024. The introduction of main crops (e.g. the crop present on June 1st on a plot; decree n° 2022-1120 of August 4, 2022 relative to crops used for the production of biogas and biofuels) in the biogas plant being limited to 15% of the biogas plant inputs in France, crop farmers without animal breeding have rather few important resources available to feed a biogas plant. Therefore, they use cover crops, which are not currently limited by regulations and are not initially meant to be harvested. Cover crops are grown between two main crops, instead of leaving the soil bare for several months, in order to provide some ecosystem services such as limiting nitrate pollution of groundwater, protecting the soil from erosion, providing nitrogen for the next crop and reduce greenhouse gas emissions by storing C in soils (Blanco-Canqui et al. 2015; Abdalla et al. 2019; Launay et al. 2021a). As they have no market value, farmers in Europe do not easily adopt them (Kathage et al. 2022). However, European Union Directive on Nitrate pollution (91/676/EC) have required the use of catch crops in some watersheds to limit nitrate pollution since 1991. This EU regulation will be extended to all farmland in 2023 with the new Common Agricultural Policy conditionality rules. In addition, the increasing carbon payment for environmental practices inducing C storage in soils could encourage farmers to adopt multi-services cover crops (MSCC; Bamière et al. 2022). When the objective becomes the production of energy, the cover crops are called energy cover crops (ECCs). They are managed to maximize the production of biomass and are harvested to feed the biogas plant. Species are chosen for their potential to produce above-ground biomass rather than for the ecosystem service they provide (e.g., grasses instead of legumes for nitrogen supply or brassicas for leaching reduction or soil-borne disease management). They can be fertilized and sometimes irrigated to increase the biomass production. Such biomass resource could be potentially important in France as shown by two studies conducted by the French Agency for the Environment and Energy Control. Based on simple assumptions and coarse calculation, these two studies estimated an ECC biomass production between 18 and 22.8 Mt

covering 11 to 30% of our gas needs by 2050 (ADEME 2018b, 2021). The figures were estimated with the insertion of ECCs on 5.6 Mha of the 17 Mha of arable land in France in 2010.

However, the extension of cover crops for energy and/or environmental purposes could raise concerns on competition for resources (water and nitrogen) with the following main crop leading to potential decrease of main crop production (Alonso-Ayuso et al. 2014; Alvarez et al. 2017) and increased water consumption (Meyer et al. 2019). In addition, the fertilization of ECCs raises concerns about increased consumption of mineral fertilizers and N leaching. For soil organic carbon (SOC) it is known that cover crops increase storage (Poeplau and Don 2015; Kaye and Quemada 2017; Jian et al. 2020), which is beneficial for both soil fertility and climate change mitigation. The introduction of AD should lead to an increase in biomass production by cover crops with the exportation of a large part of this biomass. The balance on the SOC stock is therefore less predictable. Focusing on SOC, Levavasseur et al. (2023a), showed that ECCs could lead to more C storage than non-exported cover crops when they produced twice as much biomass but this result needs to be strengthened together with a better analysis on the consequences on N cycle, water dynamics and GHG balance as well.

The objectives of our study were i) to propose a new estimate of the biomass production of ECCs taking into account the diversity of climate, soils and cropping systems at high resolution in France, and ii) to estimate the impact of ECC extension compared to MSCC extension on food crop production and environment (i.e. GHG balance, SOC storage, water consumption, mineral fertilizer consumption, and N losses). To meet our objectives, we set up a modeling chain involving four models (i.e. a soil-crop model, an AD model, a volatilization model, a N fertilization balance) to simulate current and alternative French cropping systems at high resolution according to three cover crop development scenarios. The first scenario would correspond to a wider adoption of cover crops for their ecosystem services, the second to the development of AD of ECC without redesigning the cropping systems and the last one to the further development of AD of ECCs with modifications in the cropping systems to increase their production.

5.2 Material and Methods

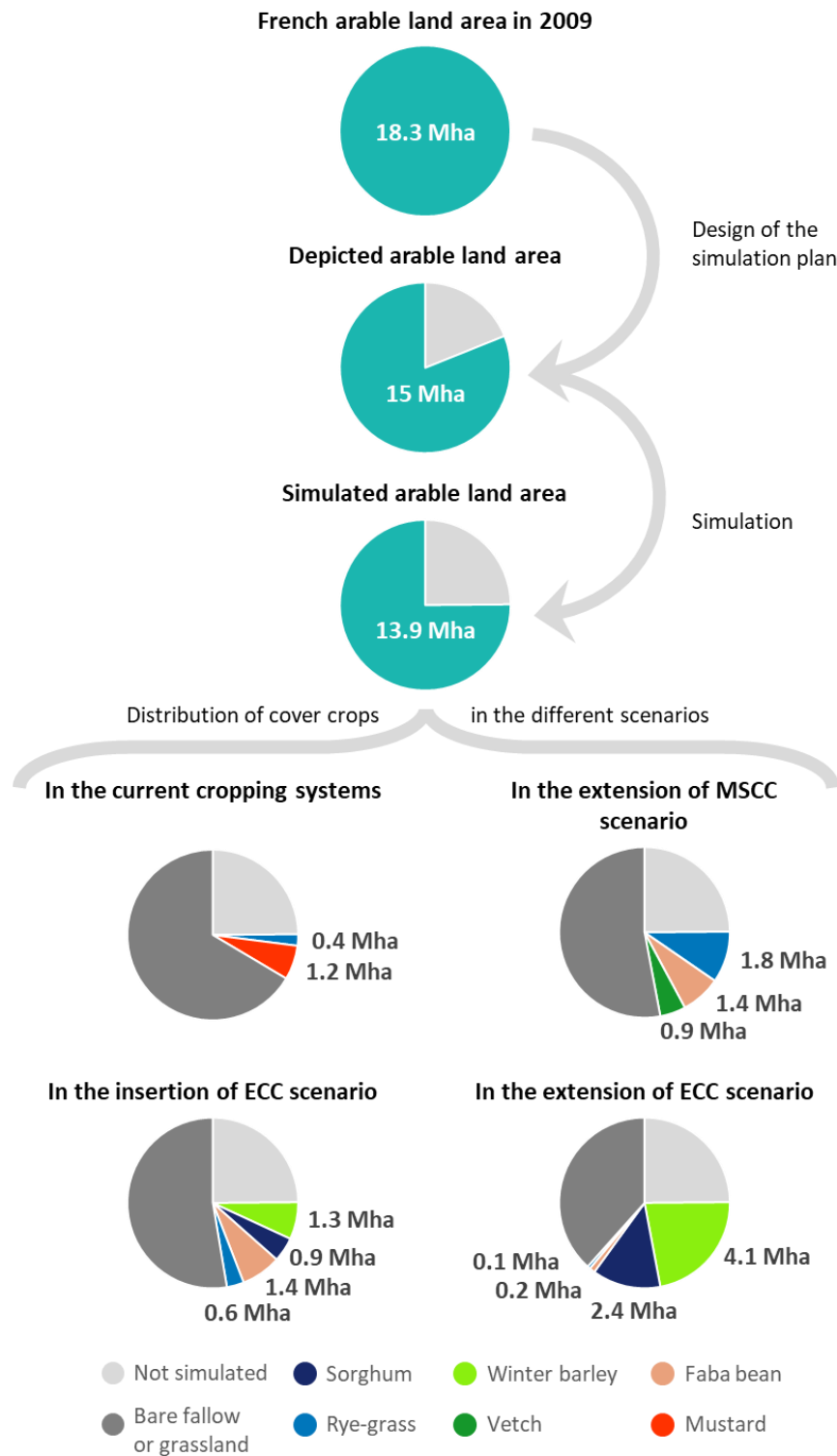


Figure 16. French arable land areas represented in our work and areas of the different cover crops species in the baseline and the three scenarios of development of multi-services cover crops (MSCC) or energy cover crops (ECC).

5.2.1 Current cropping systems in the baseline scenario

The baseline was built from the baseline of Launay et al. (2021a). The metropolitan France was divided into 30,966 pedoclimatic units (PCUs), which correspond to the intersection of the 8km x 8km SAFRAN climate grid (Durand et al. 1993) and the 1:1,000,000 French soil map (Jamagne et al. 1995). Of these 30,966 PCUs, 12,060 PCUs contained at least 10 ha of field crops and temporary grasslands in 2012. They covered 84% of French arable land (i.e. 15 Mha out of 18.35 Mha in 2009; Figure 16). For each PCU, we selected one to three predominant soil types to cover 70% of the area and one to three predominant rotations. The main crop rotations were determined using the database developed by the French National Research Institute for Agriculture, Food and Environment (INRAE; Leenhardt et al. 2012) from the French Land Parcel Identification System (LPIS) over the period 2006-2012. Field crop management practices were defined mainly from the Agricultural Practices Surveys conducted in 2006 and 2011 per crop and former administrative region (Table 33, Figure 33) by the French Ministry of Agriculture, Agri-food and Forestry. Grassland management was divided into 30 different management types constructed by Graux et al. (2020).

5.2.1.1 Crop rotations

A total of 71 different field crop rotations and 274 crop/temporary grassland rotations were simulated. The simulated crops were winter wheat to which all other winter cereals were assimilated, grain and forage corn, rapeseed, sunflower, spring and winter peas to which all annual legumes were assimilated, sugar beet and temporary grasslands (simulated only with grasses). For all these crops except corn, we used the most common variety in France available on the model used (STICS, see below). For corn, we adapted its precocity to the climatic context. The sowing and maximum harvest dates were determined from the Agricultural Practices Surveys 2006 and 2011. Sowing dates were set by crop and region while the crop model simulated harvest dates at physiological maturity.

5.2.1.2 Crop fertilization

We used the Agricultural Practice Surveys to define the area of organic fertilizer use, the type of organic fertilizer (cattle manure, pig slurry or sugar beet vinasse), their rates and frequencies of occurrence in the rotation. They also helped us to establish default mineral fertilization practices (rates and dates). After two years of initialization, mineral fertilizer rates were

calculated using the N balance method of the COMIFER (COMIFER 2013) to take into account the increase in soil N supply with the increase of soil organic matter (SOM) related with C storage. This method is described below with the other selected models.

5.2.1.3 Cover crops

In the current cropping systems, cover crops were included in accordance with the French application of European Union Directive (91/676/EC). They were grown before spring crops for 2 to 6 months (destruction from mid-November to mid-March) only in PCUs located in Nitrate Vulnerable Zones (NVZ) in 2012. No cover crops were sown after grain corn or sugar beet because of their late harvest (Figure 34). The species, either mustard or rye-grass, and dates of sowing and termination were defined by region based on the Agricultural Practice Surveys. Cover crops received no fertilization nor irrigation in the baseline scenario.

5.2.1.4 Irrigation

We used the LPIS to determine in which PCUs irrigation was the dominant practice. In these PCUs, corn, winter wheat and sunflower could be irrigated. To take into account the climatic variability, we let the model calculate during the simulation the dates and doses of irrigation as soon as crop actual transpiration covered less than 85% of water crop need. Wheat and sunflower were only irrigated around flowering with the same threshold. Thus, the crop model simulated the amount of irrigation water supplied depending on climate condition and crop growth.

5.2.2 Multi-services cover crops extension scenario

The first scenario generalized the use of multi-services cover crops (MSCC) to produce different ecosystem services in addition to N leaching reduction, such as providing N to the following crop with legumes or increasing soil carbon storage with grasses. Before spring crops, ryegrass replaced mustard in NVZ and was also sown outside of NVZ. Growing period was extended up to one month before spring crop sowing to maximize C fixation without affecting the yield of the following crop. Faba bean was sown after grain corn at the end of October and terminated a few days before sowing the next spring crop (Figure 34). No cover crops were sown after sugar beet, because of their late harvest (end of November). Finally, we introduced vetch before winter crops when the fallow lasted at least two months and sowing was possible before July 20 to ensure significant growth. With these conditions, it was mainly sown after rapeseed,

winter peas and spring peas (Table 34) and terminated a few days before sowing the winter crop. In irrigated PCUs, vetch was irrigated at sowing if necessary to ensure emergence and growth. Fertilization of the crop following a cover crop was automatically adjusted according to the above-ground biomass production of the cover crop, its C:N ratio and its date of destruction, according to the N balance fertilization model.

5.2.3 Energy cover crops insertion scenario

The second scenario introduced AD of energy cover crops (ECC). The ECC species were chosen for their high biomass production potential; sorghum was chosen in summer fallow, for its drought resistance, instead of vetch and winter barley in long winter fallow, instead of some ryegrass. Winter barley was chosen based on previous simulations for its better production potential than oats or rye in three contrasting French regions (data not shown). To obtain ECC biomass profitable for biogas, we set decision rules for sowing and harvest dates based on technical recommendations (Marsac et al. 2019). As for vetch, sorghum was not sown after July 20 to ensure sufficient growth (Table 34) and was harvested a few days before sowing the following winter crop or by October 31 at the latest to avoid too bad weather conditions at harvest time (Table 35). Winter barley was sown between September 15 and October 15 depending on the previous crop harvest date and harvested a few days before sowing the next spring crop (Figure 34). No ECC was sown after sugar beet and grain corn due to their late harvest (Table 36). The soil therefore remained bare after the sugar beet and faba bean was sown but not harvested after grain corn to provide ecosystem services like in the previous scenario. As spring pea and sugar beet were sown too early, before April, to expect a good harvest of winter ECC (Table 37), a summer ECC was sown before if the previous crop allowed it and harvested on October 31; otherwise a ryegrass was sown but not harvested. In all cases, ECCs were harvested if the harvestable biomass exceeded 5 t DM.ha^{-1} , the threshold at which harvesting is currently considered profitable (based on expert knowledge). If the production of the ECC did not reach this profitability threshold, it was buried into the soil like a MSCC.

In order to obtain high biomass, ECCs were fertilized, almost systematically with mineral fertilizers and sometimes with digestate. The first years of initialization, a dose of 70 kg N.ha^{-1} was applied. In subsequent years, the application rate was calculated using the same method as for main crops (see below). The potential yield and the N needs were given by COMIFER

according to the species sown and its cycle length. Like vetch, summer-sown sorghum was irrigated if needed at emergence in irrigable PCUs, with a maximum of 30 mm of water.

If ECCs were harvested, their exported biomass was directly digested via the SYS-Metha model (see below). The digestate produced was spread on the following crop if it did not already receive organic fertilizer (as organic fertilization practices do not change) or the next one. The applied digestate was used in the calculation of the N balance to reduce the amount of mineral fertilizer applied as a complement. The biogas produced in parallel was valorized directly by injection into the gas network to replace fossil gas. If the ECCs was not harvested, the fertilization of the next crop was also adjusted taking into account the returned biomass as in the previous scenario.

5.2.4 Energy crops extension scenario

This scenario consisted in modifying the crop rotation and crop precocity to integrate more ECCs to produce more biogas. As winter ECCs were more productive than summer ECCs in the previous scenario, we implemented modifications that favored them. First, we changed the precocity of corn and sunflower for earlier varieties to sow them two weeks later and delayed ECC harvest by the same amount of time. The change in precocity for grain corn also allowed an earlier harvest and a sowing of winter ECCs by October 15 instead of faba bean. In the crop succession, sunflower substituted one rapeseed if rapeseed returned more than one year out of three and corn substituted some winter wheat. As winter wheat was harvested too late in most regions to sow a summer ECC, we substituted we substituted one out of two with winter barley when two followed each other. Winter barley was harvested at the latest on July 15. Apart from the increase in sunflower surface, these changes in crop rotation induced by AD have already been reported in a recent study in France (Levavasseur et al. 2023b). The rest of ECC management was the same as in the previous scenario.

5.2.5 The modeling chain for scenarios simulations

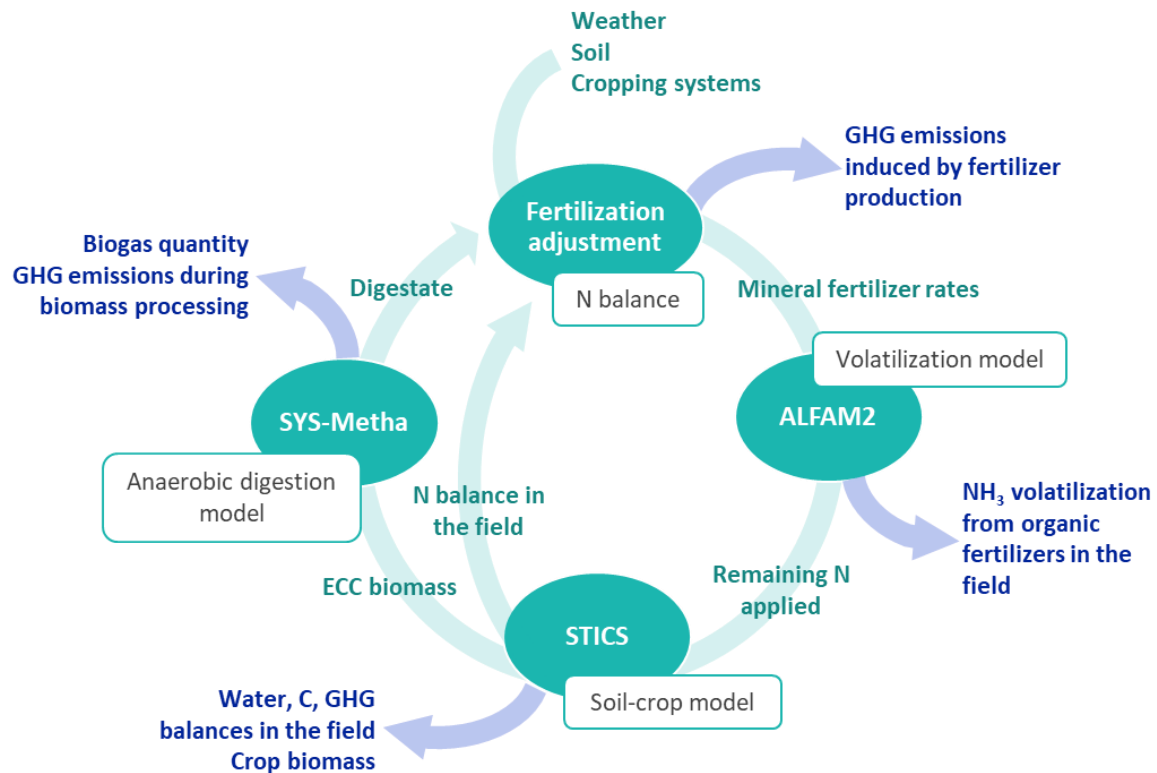


Figure 17. Modeling chain coded in R. The loop is repeated for each crop in the succession until the end of the 30 years of simulation. At the end of each loop, the outputs are saved and used in the next loop to ensure the continuity of the N, C and water balances. Some outputs (in turquoise) are used as inputs in the next model. After the crop simulation with STICS, either the ECC is not anaerobically digested and the chain is finished, or it is digested and the biomass simulated by STICS is transferred to the SYS-Metha model (anaerobic digestion model). When digestates or other organic fertilizers are applied, the ammonia volatilization is simulated by ALFAM2.

We first used the method and the dataset of Launay et al. (2021a, b) to set up the basis of the modeling chain with the crop model. They described as finely as possible (at the scale of a few km²) the current French cropping systems (crop successions and crop management) and the associated pedo-climatic conditions in order to simulate them for 30 years with the STICS crop model. To better adapt N mineral fertilization in all our scenarios, we added the N balance method as described below, considering organic or mineral fertilizer use (COMIFER 2013). In both scenarios with AD we added a module in the modeling chain allowing to transform a cover crop harvest into biogas and digestate which was spread in the crop succession as fertilizer. We used the AD model SYS-Metha (Bareha et al. 2021b) for this purpose. In addition,

given the poor results of STICS on ammonia volatilization following organic fertilizer applications (Moinard 2021), we used the ALFAM2 model in parallel to estimate this output. Figure 17 shows the architecture of the simulation chain. The whole simulation chain was coded on R version 4.0.3 thanks to the *SticsRfiles* (0.4.2) and *SticsOnR* (0.2.2) packages for the STICS part and *ALFAM2* (2.0) package for the ALFAM2 part, and thanks to the work of Malet et al. (2023) for the SYS-Metha part.

A simulation unit was a combination of 1 PCU x 1 soil x 1 rotation and its management x 30-year climate. The simulations ran with past climate from 1987 to 2019. The first two years were used to initialize soil water and mineral nitrogen stocks according to agro-pedoclimatic conditions. For more information on the construction of the simulation units and the initialization of the simulations see Launay et al. (2021a). At the output of the simulation chain, we obtained 75% of the simulation plan with consistent results, after removing aborted simulation or yield failure more than 2 years out of 10. The area represented by the PCUs present at the end was 13.9 Mha or 75.6% of the 18.35 Mha of arable crops and temporary grasslands in 2009 (Figure 16). Of this surface, the MSCC extension scenario modified 10.95 Mha, the ECC insertion scenario 10.95 Mha and the ECC extension scenario 13.6 Mha. The annual area covered by cover crops in each scenario and the baseline is detailed in Figure 16.

5.2.6 Model choice

5.2.6.1 STICS

The Multidisciplinary Simulator for Standard Crops (STICS) model (Beaudoin et al. 2022) is a process-based ecosystem model that simulates daily interactions between water, C and N cycles influenced by crop species, crop management (i.e., sowing, N fertilization, irrigation, tillage, harvest, residue management) and pedoclimate characteristics (weather, clay content, organic N content and C:N ratio of topsoil, water retention properties of the soil profile). The version used (v10) is the most recent to date and has been improved on the dynamic root mortality (Beaudoin et al. 2022) as well as on the formalism of organic fertilizer mineralization (Levvasseur et al. 2021) with the calibration of default parameters for digestate (Levvasseur et al. 2022a). Here we used the digestate decomposition parameters obtained from energy cover crop-based digestates (Levvasseur et al. 2022b). A previous version of the model was validated on more than 1800 field crop production situations in France (Coucheney et al. 2015). A recent study by Moinard (2021) evaluated the prediction quality of STICS on ammonia

volatilization following organic fertilizer applications (solid digestate, raw digestate, liquid digestate, cattle manure, cattle slurry). He concluded that the prediction quality of STICS was rather poor, especially for solid products (solid digestate, manure) or liquid products with rather high dry matter contents (raw digestate). Therefore, we used the STICS output for NH_3 volatilization following mineral fertilizer applications but we used ALFAM2 model for prediction of ammonia volatilization after organic fertilizer application.

5.2.6.2 ALFAM2

We chose this model because it has already been tested in a French context with the same types of exogenous organic matter (EOM) as in our study (i.e. raw digestate, slurry). Moinard (2021) compared it with STICS and obtained better performances on these products. It was originally calibrated on cattle and pig slurry in a wide variety of application conditions (Hafner et al. 2019). In the absence of a more complete model, we chose to use it also outside its range of validity on solid EOMs (>15% dry matter). A first experiment with cattle manure, horse manure and green compost gave plausible results (Chapter 4). The advantage of this model over the use of an emission factor is that it can take into account the impact of temperature, wind, precipitation, NH_4^+ content, and application method on volatilization.

5.2.6.3 SYS-Metha

SYS-Metha models in a simple way the mass balance in a digester (Bareha et al. 2021b). It is able to simulate C and N fluxes during substrate AD, during phase separation if this option is chosen, and during digestate storage. It takes as input a wide range of substrates including some energy cover crops. As output, it is able to provide the quantities and composition of the digestate and biogas produced, including all the parameters needed by STICS to simulate the digestate (i.e. dry matter, NH_4^+ , C content, C:N ratio), and also the emissions of CO_2 , CH_4 , N_2O and NH_3 during storage of the digestate. Malet et al. (2023) directly coupled this model to the calculation of GHG emissions during the functioning of the biogas plant, from the storage of the incoming matter to the injection of the purified methane.

5.2.6.4 N balance

We calculated the mineral fertilizer rates supplementing or not organic fertilization using the N balance method of the COMIFER (COMIFER 2013). This method calculates the amount of mineral fertilizer to be applied to reach a target yield according to the following equation:

$$X = \frac{(P_f - (R_i + M_{CC} + M_{SOM} + M_{EOM} + M_{TG}) + R_f)}{ACU} \quad (1)$$

The amount of mineral fertilizer to be applied X was calculated as the crop requirement P_f (i.e., target yield multiplied by N requirement per unit of production) plus the expected post-harvest N residue R_f minus the initial soil mineral N stock at sowing R_i , the amount of N mineralized from the cover crop residues preceding the crop M_{CC} , the amount of N mineralized from the SOM between the date of the first N application and the harvest M_{SOM} , averaged over the last 3 years to take into account SOM change through time, the amount of N mineralized from the organic fertilizer M_{EOM} , plus the amount of N mineralized from the pasture preceding the crop M_{TG} . We applied an apparent coefficient of utilization of the mineral fertilizer ACU , to take into account gaseous loss of the mineral fertilizer. We chose as target yield the average yield obtained per simulation unit in Launay et al. (2021a). The different N supply items were calculated after the end of the simulation of the previous crop thanks to the STICS outputs. The N requirement per unit of production, R_f , M_{TG} and ACU were provided by COMIFER.

5.2.7 Calculations and extrapolation

The outputs of each simulation unit affected by the scenarios were analyzed from 1989 to 2019 after the 2 year-initialization. They were retrieved at the end of each cropping cycle (i.e. harvest of main crops and energy cover crops or termination of cover crops) and averaged over 30 years. Within each PCU, the results for each simulation unit were averaged by the weight of their respective area and then extrapolated to the entire PCU arable area. We calculated the biomass produced and exported annually by cover crops on the entire simulated French surface (i.e. 13.9 Mha). To evaluate the environmental impacts of the different scenarios we focused on the arable crop systems only (i.e. without grassland and manure application, 8 Mha). In livestock systems, the introduction of AD would necessarily involve the digestion of livestock manure. In our methodology, we could not consider this, yet the transformation of manure into digestate can modify fertilization practices, which in turn can modify nitrogen flows, notably ammonia volatilization.

We performed a sensitivity analysis of the harvested ECC biomass to the assumptions of profitability threshold and cutting height. For these two assumptions, the modalities entered into the model were a harvestable biomass threshold of 5 t DM.ha⁻¹ and an amount of stubble left in place at harvest representing one third of the total aboveground biomass. This meant

that the ECC was harvested if it produced at least 7.5 t DM.ha⁻¹ of aerial biomass. Without re-running simulations, we calculated on the simulation outputs how much biomass could have been exported with other assumptions. We tested a profitability threshold of 4 t DM.ha⁻¹ to simulate a stronger incentive to use ECCs and an amount of stubble left on the field of 1 t DM.ha⁻¹ to represent mowing as close to the ground as possible.

All the results, apart from the cumulative biomass produced on the territory, were expressed as a differential compared to the baseline scenario. The soil carbon storage differential, for example, is the difference between the average annual storage in one scenario (MSCC or ECC) and the average annual storage of the baseline scenario.

We calculated a GHG balance (*GHG balance*, kg CO₂e.ha⁻¹.yr⁻¹) taking into account field emissions and emissions due to anaerobic digestion:

$$GHG\ balance = 296 \times \frac{44}{28} (direct\ N_2Oe + indirect\ N_2Oe) - \frac{44}{12} \Delta SOC_{0-0.3m} + 5.34 \times Nfertilizer + E_{digester} - 63.1 \times Gas\ substitution \quad (2)$$

The GHG balance take into account the amount of N₂O emitted from the soil *direct N₂Oe* (kg N₂O-N.ha⁻¹.yr⁻¹), and the amount of N₂O emitted throughout the N cascade through NH₃ volatilization and nitrate leaching *indirect N₂Oe* (kg N₂O-N.ha⁻¹.yr⁻¹). Direct N₂O emissions were simulated by STICS and averaged over 30 years. Indirect N₂O emissions were estimated using emission factors defined by the IPCC (De Klein et al. 2006): 1% of the N from fertilizers that volatilizes being transformed into N₂O and 0.75% of the leached N being transformed into N₂O throughout the N cascade in groundwater, rivers and estuaries. Soil carbon storage was calculated by STICS to a depth of 0.3m. Outside the field, we took into account the emissions related with fertilizer production (kg CO₂e.ha⁻¹.yr⁻¹), which equaled the amount of mineral *Nfertilizer* applied per ha multiplied by an emission factor of 5.34 kg CO₂e.kg⁻¹ of N for ammonium nitrate. Emissions during the AD process are grouped under the heading *E_{digester}*. They gather the emissions linked to the electricity consumption of the anaerobic digester, the fugitive CH₄ emissions, the CO₂ and CH₄ emissions during the flaring, the emissions during the digestate storage knowing that we consider it as covered with biogas recovery, and the emissions linked to the purification and injection of the biogas. For all these emission factors, we used the assumptions of Malet et al. (2023). Finally, we took into account the substitution

of fossil gas by biomethane *Gas substitution* with the substitution factor of the French mix of 63.1 kg CO₂e.GJ lower heating value (LHV) (Malet et al. 2023).

We constructed correlation matrices with Pearson's test between different agro-environmental variables used as input to the model or resulting from the models (Table 38). The correlation coefficients were visualized with the corrplot R package (v0.92).

5.3 Results

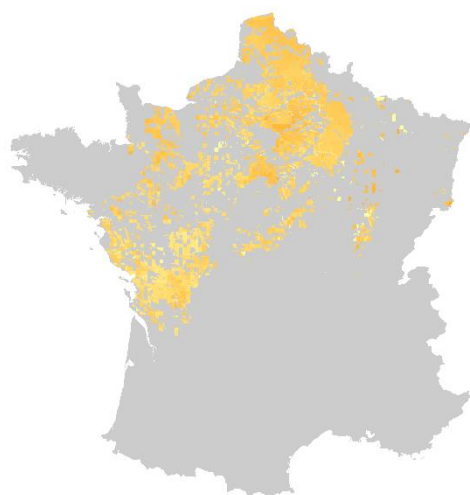
5.3.1 Production of biomass

5.3.1.1 Cover crops production of biomass

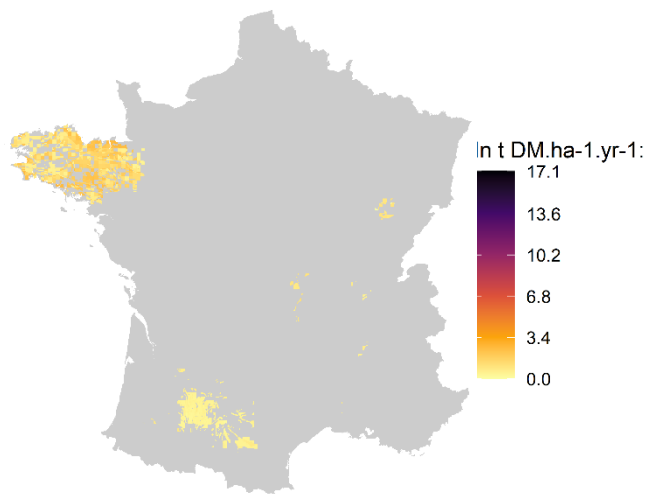
Table 18. Cover crop total aerial biomass in Mt of dry matter produced in the simulated area in the different scenarios. The area covered annually by each species is mentioned in brackets.

	Baseline	MSCC extension	ECC insertion	ECC extension
Mustard	2.1 (1.2 Mha)	.	.	.
Rye-grass	0.5 (0.4 Mha)	3.5 (1.8 Mha)	0.6 (0.6 Mha)	0.1 (0.1 Mha)
Faba bean	.	4.8 (1.4 Mha)	4.8 (1.4 Mha)	0.1 (0.2 Mha)
Vetch	.	2.3 (0.9 Mha)	.	.
Sorghum	.	.	2.2 (0.9 Mha)	6.7 (2.4 Mha)
Barley	.	.	10.3 (1.3 Mha)	36.0 (4.1 Mha)
TOTAL	2.5	10.5	17.8	42.9

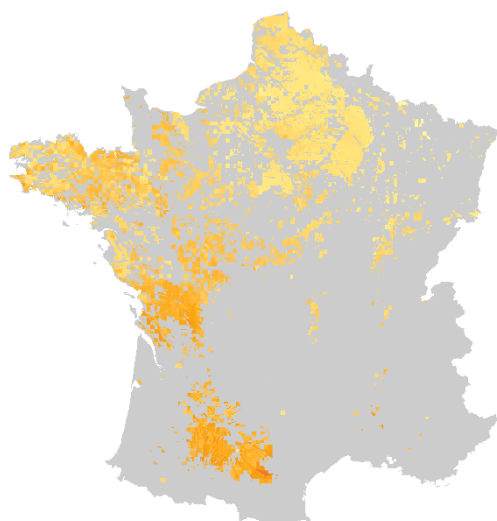
(a) Baseline: mustard



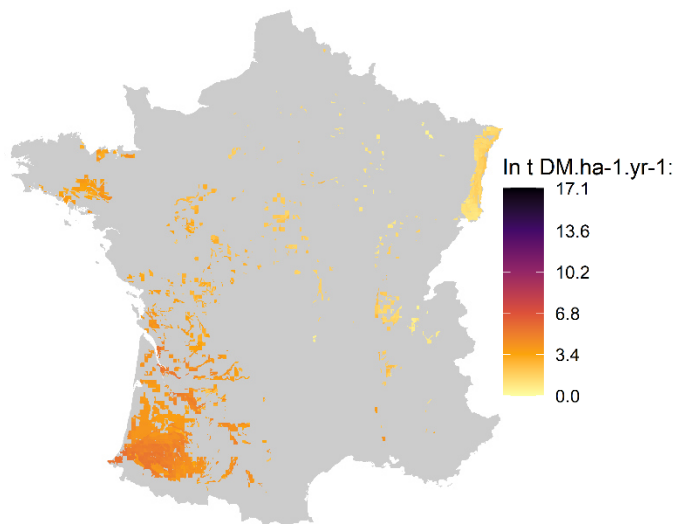
(b) Baseline: rye-grass



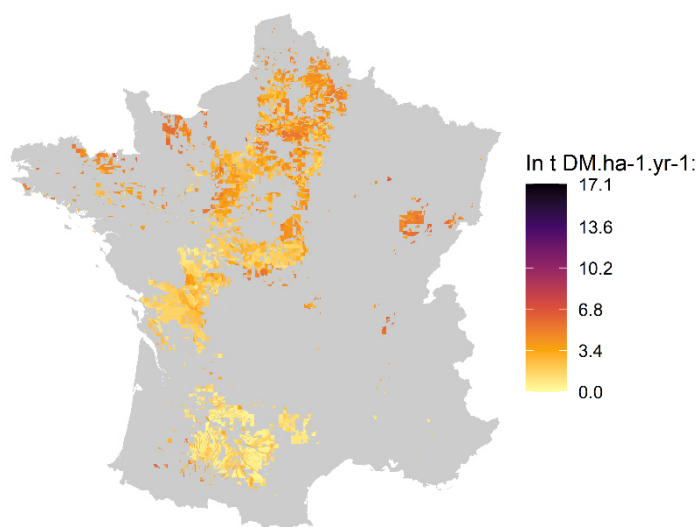
(c) MSCC extension: rye-grass



(d) MSCC extension: faba bean

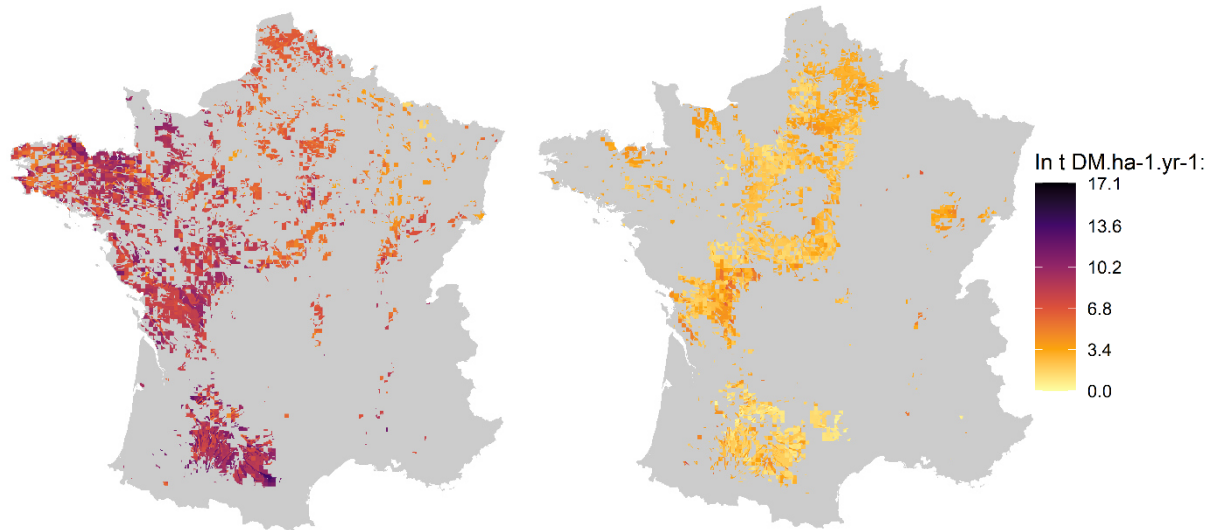


(e) MSCC extension: vetch



(f) ECC insertion: winter barley

(g) ECC insertion: sorghum



(h) ECC extension: winter barley

(i) ECC extension: sorghum

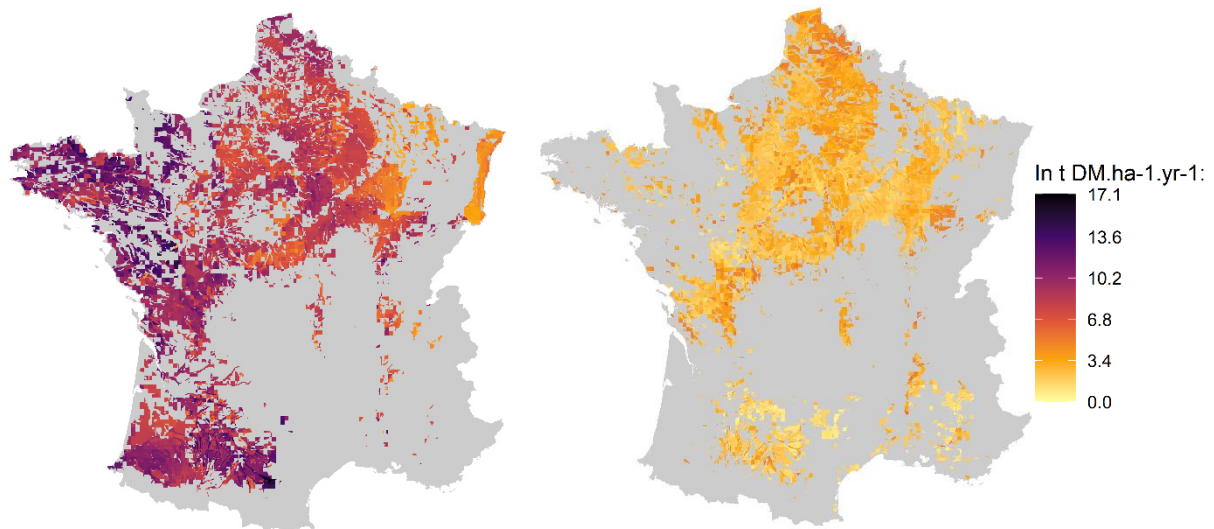


Figure 18. Aerial biomass production (t DM.ha⁻¹) of (a) rye-grass and (b) mustard in the baseline, (c) rye-grass, (d) faba bean and (e) vetch in the MSCC extension scenario, (f) winter barley and (g) sorghum in the ECC introduction scenario, (h) barley and (i) sorghum in the ECC extension scenario.

In the current cropping systems, the average production of cover crops was 1.6 t DM.ha⁻¹ for mustard and 1.2 t DM.ha⁻¹ for ryegrass. Ryegrass was mostly established in Brittany (western France) and part of the southwest (Figure 18a), while mustard was established in a large part of the northern half of the country (Figure 18b). While mustard production was rather homogeneous on the country, except for a few points where emergence was always a failure, ryegrass production was rather heterogeneous and much lower in the southwest with late-

autumn termination than in the west region of Brittany with late-winter termination. The cumulated total aerial biomass produced yearly on the simulated area was rather low (Table 18).

In the MSCC extension scenario, ryegrass, which expanded in surface and terminated later, produced slightly more biomass than in the baseline, averaging 1.9 t DM.ha^{-1} . Delaying ryegrass termination by five months in the southwest was very beneficial since production reached 3.2 t DM.ha^{-1} on average, i.e. 2.7 t DM.ha^{-1} more than in the baseline (Figure 18c). In Brittany, where the extension was only one month because of its already late termination in the baseline, production increased by only 0.6 t DM.ha^{-1} . Overall, there was a northeast to southwest gradient in ryegrass production. The climate in the north of France was less favorable to ryegrass than to mustard, with only 1.1 t DM.ha^{-1} against 1.8 t DM.ha^{-1} , respectively. In this scenario, faba bean was mostly sown every year in the grain corn monocultures of southwestern France and northeast in Alsace (Figure 18d). The production difference between these two regions was important with 4.3 t DM.ha^{-1} for the former and 1.4 t DM.ha^{-1} on average for the latter. Vetch produced in average 2.7 t DM.ha^{-1} and followed a gradient from southwest to northeast; the climate in the north of France being more favorable than in the south (Figure 18e). Total biomass production by cover crops increased fourfold compared to the baseline (Table 18).

In the ECC insertion scenario, barley was mostly introduced in the western half of France (Figure 18f) and produced on average 8.0 t DM.ha^{-1} . In central and eastern France, production was lower than in the west. In this scenario, ryegrass before sugar beet in the north of France and faba bean in the grain corn monocultures of the southwest and Alsace produced the same biomass as in the MSCC extension scenario. In summer fallows, sorghum produced the same biomass on average as vetch: 2.7 t DM.ha^{-1} but the distribution of this biomass was not the same (Figure 18g). Irrigation at emergence in irrigated areas was beneficial to this crop, as it increased its production by 1.0 t DM.ha^{-1} compared to the French average. In rainfed areas, unlike vetch, production was rather homogeneous. Thanks to barley, the cover crops in this scenario produced more cumulative aerial biomass on the simulated area than in the baseline and in the previous scenario by 600% and 70%, respectively (Table 18).

In the ECC extension scenario, barley was sown much more widely than in the previous scenario; it was sown on nearly all the country but with a varying frequency in crop succession

(Figure 18h). It produced an average of 9.3 t DM.ha⁻¹. An east to west gradient was observed, with production levels below 7 t DM.ha⁻¹ in the northeast of France (Alsace and south of Champagne-Ardenne). Sorghum expanded in the Center, North, Northeast and Southwest of France and averaged around 2.9 t DM.ha⁻¹ (Figure 18i). It was rather evenly distributed over the country, but biomass production remained low. Production could be higher in some scattered areas. Almost all the biomass produced in this scenario, 84%, originated from barley (Table 18). Ryegrass and faba bean almost disappeared and despite the large area sown with sorghum, its production was low. This scenario produced 2.4 times more biomass than the scenario of insertion of ECCs without modification of the cropping systems, i.e. 17 times more biomass than cover crops in the baseline.

5.3.1.2 Exportation of energy cover crops biomass

Considering the threshold of 5 t DM yr⁻¹ to harvest and mowing that leaves 33% of the biomass on site, barley was exported on average 6 years out of 10 in the ECC insertion scenario and 7 years out of 10 in the ECC extension scenario, except for northeastern France where yield was too low. The choice of productive species, fertilization and growing period extension allowed the winter ECC to consistently achieve their production target. On the contrary, sorghum as summer ECC did not fulfill its role. It could only be exported in a few situations in the southwest in the ECC insertion scenario and in a few additional situations in the rest of the country in the ECC extension scenario. It produced as much as vetch in the MSCC extension scenario even though vetch was not sown for the same purpose.

On the simulated arable area of 13.9 Mha, barley exported 4.4 Mt DM out of 10.3 Mt DM produced in the ECC insertion scenario, and four times more in the ECC extension scenario, with 18.8 Mt DM harvested out of 36 Mt DM produced (Figure 19). As a result of the low frequency of sorghum exports, the biomass harvested was less than 0.01 Mt DM out of 2.2 Mt DM produced in the ECC insertion scenario and out of 6.7 Mt DM produced in the ECC extension scenario.

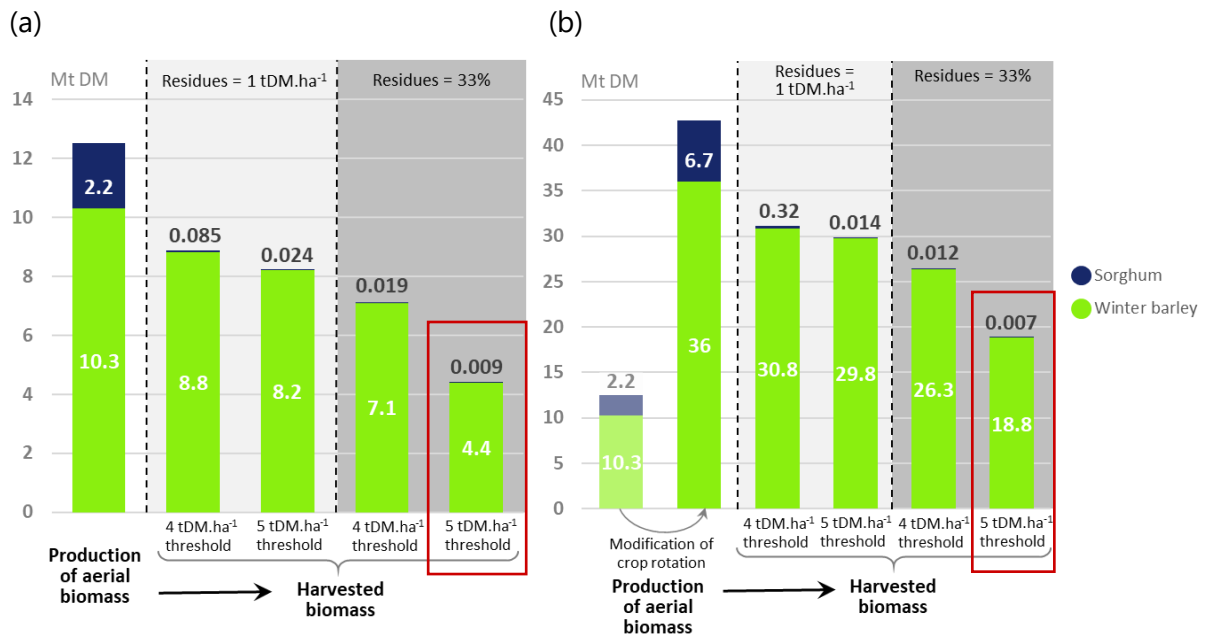


Figure 19. Produced and harvested aboveground biomass of energy cover crops in (a) the energy cover crops insertion scenario and (b) the energy cover crops extension scenario. Biomasses were calculated for the simulated arable area of 13.9 Mha (out of the 18.5 Mha of actual arable area in 2009) and are expressed in Mt dry matter (DM). The harvested biomass was calculated by crossing 4 decision rules: i) the biomass is harvested if harvestable production reaches 4 t DM.ha⁻¹ or ii) the biomass is harvested if harvestable production reaches 5 t DM.ha⁻¹, iii) the mowing is close to the ground and leaves 1 t DM.ha⁻¹ on site or iv) the mowing is higher and leaves 33% of the biomass on site. The hypothesis entered in STICS were the second hypothesis on the harvest threshold and the second one on the cutting height (i.e. the rightmost bar in each graph).

If we had chosen a threshold of 4 t DM.ha⁻¹, we could have increased the harvest by 61% (7.1 Mt DM) and 40% (26.3 Mt DM) for barley in the first and second scenarios respectively, and by 111% (0.02 Mt DM) and 71% (0.01 Mt DM) for sorghum (Figure 19). With another assumption on the cutting height, i.e. if the cut left only 1 t DM.ha⁻¹ instead of one third of the aboveground biomass in the STICS model, then the harvested biomass would have been 86% (8.2 Mt DM) and 59% (29.8 Mt DM) higher for barley in both scenarios respectively and 167% (0.02 Mt DM) and 100% (0.01 Mt DM) higher for sorghum. Depending on the assumptions chosen, the total biomass harvested could therefore vary from simple to double: from 4.4 Mt DM to 8.9 Mt DM in the first scenario and from 18.8 Mt DM to 31.1 Mt DM in the second scenario, mainly due to the increase in barley harvestable biomass.

5.3.1.3 Main crops production of biomass

Table 19. Yield of main crops in the whole simulated France for the baseline and in the area modified by each scenario. Yield is expressed in t dry matter per hectare and differences between the scenarios and the baseline are expressed in percentage of the yield in the baseline in the area concerned by the modifications.

	Wheat	Rapeseed	Sugar beet	Sunflower	Grain corn	Forage corn
Baseline	6.2	2.9	11.6	1.8	8.5	6.8
MSCC	6.4	2.9	11.5	1.6	8.2	6.3
extension	+2.3%	+3.1%	-0.5%	-9.8%	-6.5%	-7.5%
ECC	6.4	2.9	11.5	1.5	8.1	6.1
insertion	+2.6%	+3.9%	-0.9%	-17.3%	-8.9%	-11.7%
ECC	6.3	3.1	11.9	1.6	7.3	5.9
extension	+2.7%	+8.6%	+2.1%	-14.8%	-9.5%	-14.1%

In the MSCC insertion scenario, in the cropping systems modified by the scenario, spring crop yields decreased while winter crop yield little changed. Sugar beet yield decreased by 0.5%, grain corn by 6.5%, forage corn by 7.5% and sunflower by 9.8% in average (Table 19). The insertion and growing period extension of winter cover crops had a negative impact on the growth of the following spring crop, in average. In the irrigated area, the yield of grain corn did not change after faba bean. We could therefore hypothesize that the faba bean mainly caused water stress on the following crop. The vetch sown in summer fallow did not impact the following winter crop.

In the ECC insertion scenario, spring crop yield was slightly more impacted than in the MSCC extension scenario, despite N fertilization adjustment. Sugar beet yield decreased by 0.9%, grain corn by 8.9%, forage corn by 11.7%, and sunflower by 17.3%. Sunflower was thus the food crop most disadvantaged by ECCs. The yield of winter wheat and rapeseed changed little as in the MSCC extension scenario. The insertion of fertilized sorghum thus seemed to have the same impact as the insertion of unfertilized vetch on the production of the following crop.

In the ECC extension scenario, spring crop yield, except for sugar beet, was even more affected than in the previous two scenarios. The yield of grain corn decreased by 9.5%, forage corn by 14.1%, and sunflower by 14.8%. The yield of sugar beet slightly increased compared to the baseline. In contrast to the other two scenarios, thanks to the replacement of wheat by winter barley before the sugar beet, it was mainly a summer ECC that was sown before the

sugar beet instead of a winter MSCC (rye-grass). Due to the earlier termination of the summer ECC compared to the winter MSCC, the sugar beet was no longer stressed. Rapeseed yield increased slightly compared to the baseline and the other two scenarios. It could be due to the reduction in the area of this crop, which would be more concentrated in favorable areas and years.

5.3.2 Environmental impact on non-livestock systems

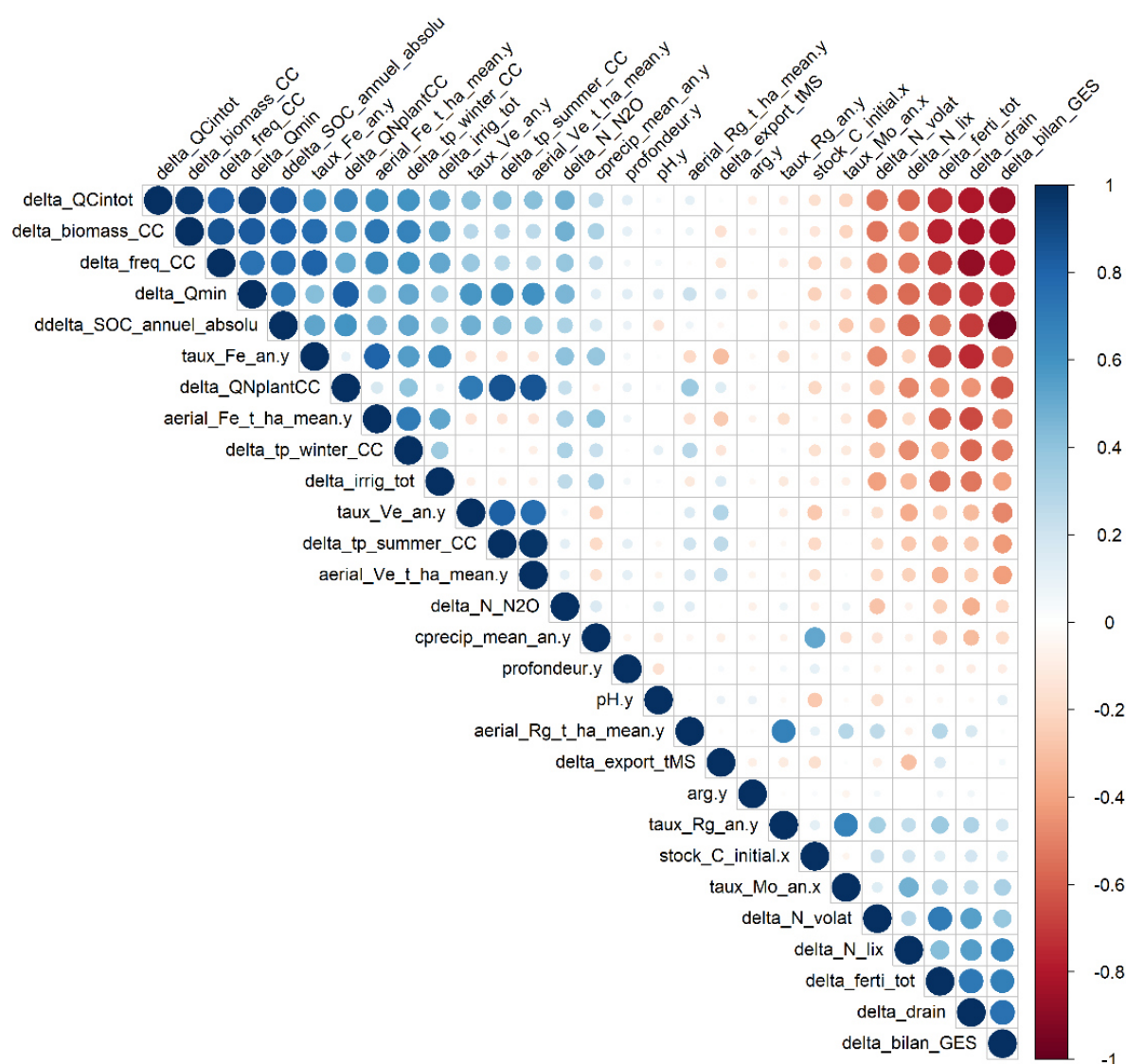


Figure 20. Correlation matrix between different agri-environmental variables in the MSCC extension scenario. The variables preceded by a "delta" are differentials between the scenario and the baseline. Blue circles represent positive correlation coefficients and red circles negative coefficients. The darker and larger the circles, the higher the correlation coefficients. The meaning of the variables is described in Table 38.

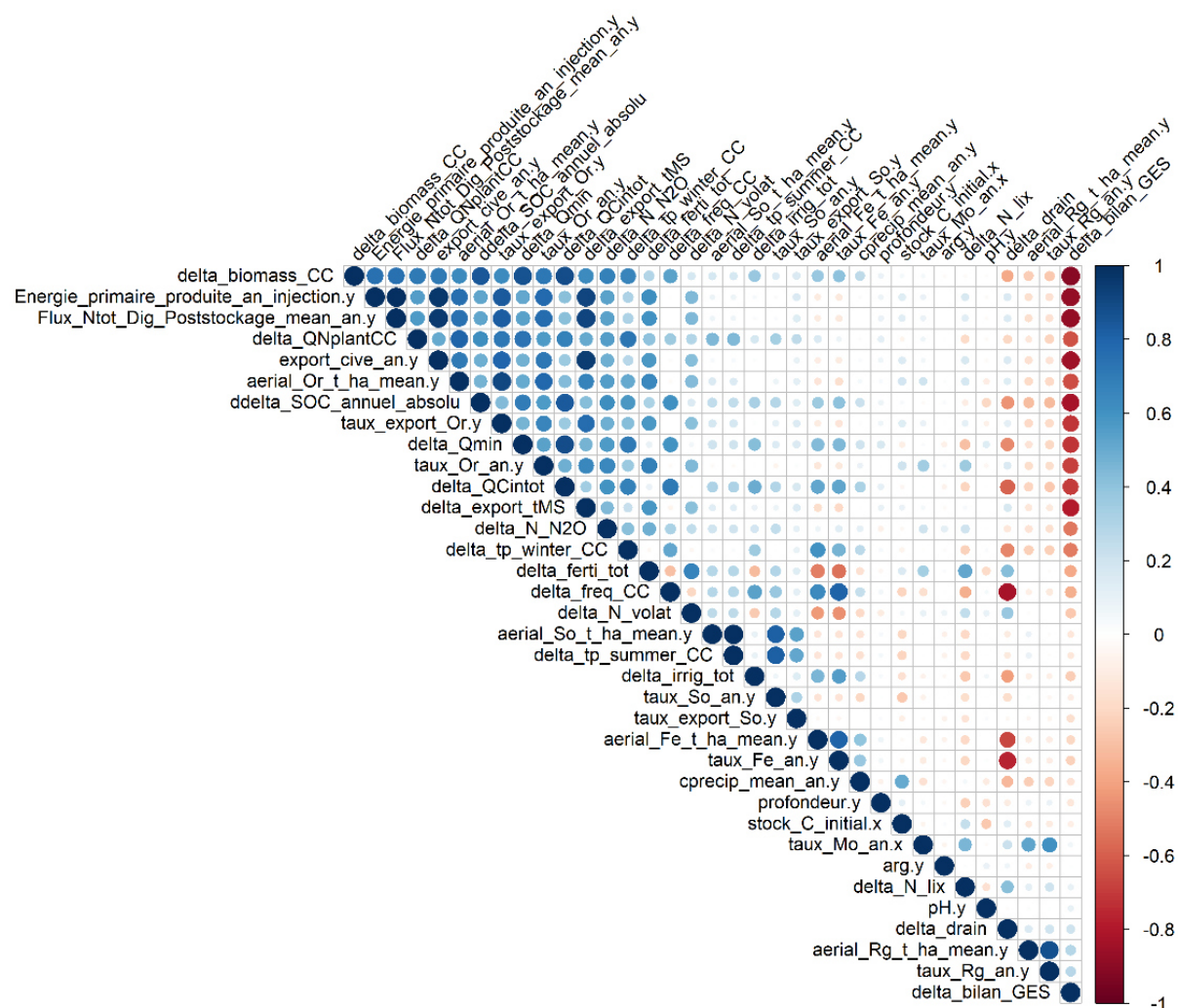


Figure 21. Correlation matrix between different agri-environmental variables in the ECC insertion scenario. The variables preceded by a "delta" are differentials between the scenario and the baseline. Blue circles represent positive correlation coefficients and red circles negative coefficients. The darker and larger the circles, the higher the correlation coefficients. The meaning of the variables is described in Table 38.

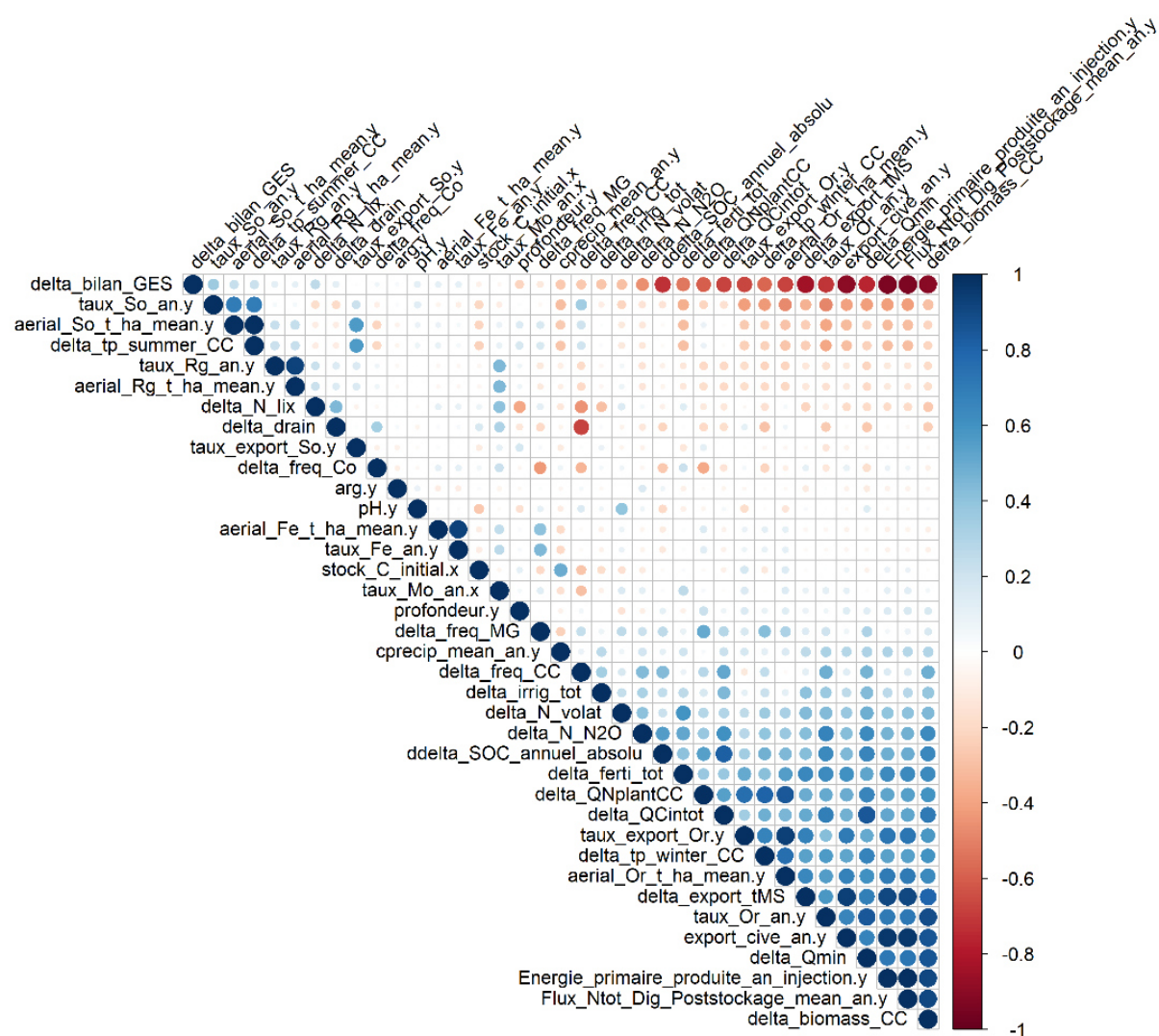


Figure 22. Correlation matrix between different agri-environmental variables in the ECC extension scenario. The variables preceded by a "delta" are differentials between the scenario and the baseline. Blue circles represent positive correlation coefficients and red circles negative coefficients. The darker and larger the circles, the higher the correlation coefficients. The meaning of the variables is described in Table 38.

5.3.2.1 Climate change mitigation

Table 20. Differences in GHG balance expressed as kg CO₂e.ha⁻¹.yr⁻¹ between the different scenarios and the baseline for field crop systems in the area affected by each scenario. The GHG balance is the sum of direct and indirect (ammonia volatilization and leaching) N₂O emissions from the field, emissions compensated by SOC storage, emissions induced by the production of N mineral fertilizer used in the field, emissions from the biogas plant and emissions avoided thanks to the substitution of fossil gas.

	Δ GHG balance	Δ N₂O emissions	Δ SOC storage	Δ mineral fertilizer	Δ digester emissions	Δ energy production
MSCC extension	- 219 ± 371	+ 21 ± 54	- 220 ± 306	- 19 ± 84	0	0
Time extension of MSCC	+ 40 ± 138	+ 19 ± 29	- 1 ± 130	+ 20 ± 26	0	0
Insertion of MSCC	- 407 ± 311	+ 12 ± 60	- 363 ± 263	- 57 ± 82	0	0
ECC introduction	- 447 ± 597	+ 77 ± 85	- 308 ± 318	+ 60 ± 121	+ 177 ± 302	- 453 ± 774
ECC extension	- 1031 ± 1030	+ 110 ± 123	- 471 ± 415	+ 120 ± 112	+ 508 ± 575	- 1298 ± 1470

French current cropping systems (without livestock systems) emitted 1786 kg CO₂e.ha⁻¹.yr⁻¹. Globally, the three scenarios of insertion and extension of cover crops improved the GHG balance of French cropping systems not dedicated to livestock, the decrease ranging from - 219 to -1031 kg CO₂e.ha⁻¹.yr⁻¹ (Table 20). They all stored more C in soil while slightly increasing N₂O emissions. Impacts on fertilizer consumptions was variable from one scenario to another. The two latter scenarios emitted from the biogas plant but also produced energy in addition, leading to different levels of GHG balance improvement.

The **MSCC extension scenario** improved it by 219 kg CO₂e.ha⁻¹.yr⁻¹, i.e. a 13% decrease of the baseline emissions where the scenario could be implemented. This decrease came from the increase in SOC storage (on average +60 kg C.ha⁻¹.yr⁻¹) and to a lesser extent from the decrease in synthetic fertilizer use (-4 kg N.ha⁻¹.yr⁻¹) despite a slight increase of N₂O emissions, by 0.05 kg N.ha⁻¹.yr⁻¹. In the cropping systems where MSCCs were added and not simply extended in time, the difference with the baseline was accentuated; the GHG balance was improved by 407 kg CO₂e.ha⁻¹.yr⁻¹ instead of 219, corresponding to a 23% decrease in baseline emissions in the concerned area, SOC storage increased by 99 kg C.ha⁻¹.yr⁻¹ instead of 60 and fertilizer consumption decreased by 11 kg N.ha⁻¹.yr⁻¹ instead of 4. All three GHG balance components were strongly correlated with the frequency of insertion ($R_2 = 0.77$ for SOC storage, 0.38 for

N₂O emissions and -0.69 for fertilizer consumption) and the differential in biomass produced by MSCCs ($R_2 = 0.81$ for SOC storage, 0.47 for N₂O emissions and -0.76 for fertilizer consumption) between the MSCC extension scenario and the baseline (Figure 20). Additional SOC storage was also correlated with the irrigation difference between both scenarios ($R_2 = 0.35$). Therefore, additional SOC storage was higher in the irrigated southwestern part where faba bean was sown every year and produced significant biomass (Figure 35a). Conversely, this scenario destored SOC compared to the baseline in northern France where ryegrass replaced mustard with a longer growing period but a lower biomass production because it was less adapted to the climate. The slight increase in N₂O emissions was highly correlated ($R_2 = 0.46$) to the increase in soil organic N (SON) mineralization. It was also much more correlated with the frequency of insertion and the biomass produced by faba bean ($R_2 = 0.40$ and 0.33 respectively) than with the other cover crop species ($R_2 < 0.2$). Among all cover crop species, faba bean was also the species most correlated with the decrease in fertilization ($R_2 = -0.64$), with vetch, the other legume cover crop, coming in second ($R_2 = -0.24$). Thus, in the southwest and in northeast where faba bean was inserted, the decrease in fertilizer consumption was the highest (Figure 35b). On the other hand, in the north of France where ryegrass simply replaced mustard without the insertion of legume MSCC, fertilizer consumption increased slightly.

The **ECC insertion scenario** improved the GHG balance by 447 kg CO₂e.ha⁻¹.yr⁻¹ in the field crop systems where it could be implemented, i.e. a 26% decrease of the baseline emissions (Table 20). The improvement of the GHG balance was due to the increase of SOC storage by 84 kg C.ha⁻¹.yr⁻¹ and to the substitution of fossil gas by biomethane which allowed to avoid the emission of 453 kg CO₂e.ha⁻¹.yr⁻¹. Besides, N₂O emissions increased by 0.15 kg N.ha⁻¹.yr⁻¹ and fertilizer consumption by 11 kg N.ha⁻¹.yr⁻¹. Fertilizer consumption tended to decrease on the main crops (-2.5 kg N.ha⁻¹.yr⁻¹) but the fertilization of ECCs increased the overall consumption. These received 57 kg N.ha⁻¹.yr⁻¹ on average for barley and 60 kg N.ha⁻¹.yr⁻¹ for sorghum. In this scenario, the increase in N₂O emissions was as correlated to the increase in SON mineralization as to the increase in cover crop fertilization ($R_2 = 0.56$ and 0.49, respectively, Figure 21). The increase in N₂O emissions was thus visibly higher in some regions in the west (Midi-Pyrénées, Basse-Normandie and Poitou-Charentes) where barley was sown at least one year out of two.

The **ECC extension scenario** improved the GHG balance by 1,031 kg CO₂e.ha⁻¹.yr⁻¹ or a 58% decrease in baseline emissions (Table 20). The substitution of fossil gas by biomethane avoided

the emission of 1,298 kg CO₂e.ha⁻¹.yr⁻¹ in this scenario and the SOC storage increased by 128 kg C.ha⁻¹.yr⁻¹. On the other hand, N₂O emissions and fertilizer consumption also increased more than in the previous scenario. N₂O emissions increased by 0.24 kg N.ha⁻¹.yr⁻¹ on average and fertilizer consumption by 22 kg N.ha⁻¹.yr⁻¹ despite a decrease of 17 kg N.ha⁻¹.yr⁻¹ on the main crops due to the use of digestate. The correlations between fertilizer consumption, N₂O emissions, SOC storage and other variables were globally the same as in the previous scenario, with barley having a much greater influence than sorghum (Figure 22). Additional SOC storage remained higher in irrigated areas. It was rather correlated to the difference in exported biomass between the scenarios ($R_2 = 0.43$), but it was nevertheless rather high in northeast (Alsace), where the barley that was included was almost never harvested (Figure 35a).

5.3.2.2 Air quality improvement

Table 21. Differences in different environmental impacts between the scenarios and the baseline for field crop systems in the area affected by each scenario. The irrigation differential concerns only the irrigated area.

	ΔN₂O direct emissions kg N.ha ⁻¹ .yr ⁻¹	ΔNH₃ volatilization kg N.ha ⁻¹ .yr ⁻¹	ΔNO₃⁻ leaching kg N.ha ⁻¹ .yr ⁻¹	ΔDrainage mm.ha ⁻¹ .yr ⁻¹	ΔIrrigation mm.ha ⁻¹ .yr ⁻¹
MSCC extension	0.05 ± 0.11	-0.29 ± 1.23	-0.8 ± 7.9	-14.8 ± 33.7	35.5 ± 38.8
Time extension of MSCC	0.02 ± 0.05	0.15 ± 0.26	2.5 ± 4.4	5.9 ± 9.1	0.2 ± 2.3
Insertion of MSCC	0.06 ± 0.12	-0.64 ± 1.34	-4.3 ± 6.9	-29.6 ± 31.3	39.3 ± 40.7
ECC insertion	0.15 ± 0.15	0.54 ± 1.62	1.9 ± 10.6	-12.9 ± 34.0	49.0 ± 31.2
ECC extension	0.24 ± 0.24	1.49 ± 1.50	-2.5 ± 11.9	-14.0 ± 23.0	65.8 ± 36.6

In the **MSCC extension scenario**, ammonia volatilization tended to decrease by 0.29 kg N.ha⁻¹.yr⁻¹ on average (Table 21). The change in volatilization compared to the baseline was positively correlated with the change in the amount of mineral fertilizer used ($R_2 = 0.70$, Figure 20). Thus, the decrease in volatilization was stronger in the southwest and Alsace where the insertion of faba bean greatly decreased the use of mineral fertilizer. We found a small correlation with soil pH ($R_2 = -0.17$) that was expected but probably blurred by many interactions. The higher the pH was, the stronger the decrease in volatilization with the decrease in fertilizer use.

In the **ECC insertion scenario**, ammonia volatilization tended to increase by $0.54 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ (Table 21). As volatilization was still highly correlated with the amount of mineral fertilizer used ($R_2 = 0.66$, Figure 21), volatilization increased the most in west (Brittany, Basse-Normandie) and southwest (Midi-Pyrénées) where fertilization had increased the most with the high frequency of ECC insertion. Volatilization was also correlated to a lesser extent with the amount of N provided by the digestate ($R_2 = 0.44$) knowing that the digestate had a low NH_4^+ content (10% of total N) compared to a real digestate.

In the **ECC extension scenario**, ammonia volatilization tended to increase more strongly than in the ECC insertion scenario by $1.49 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ compared to the baseline (Table 21). The relationship between volatilization and pH was more visible than in the other scenarios ($R_2 = 0.39$, Figure 22). Alkaline soils were more sensitive to changes in fertilization. The amount of N provided by the digestate was less correlated with volatilization than the mineral fertilizer use differential ($R_2 = 0.40$ and 0.60 respectively). On average, digestate provided only $19 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$. Focusing on simulations where ECCs were inserted and exported at least one year out of two, the dose rose to $74 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ on average. But only 10% of this nitrogen was in ammonia form.

5.3.2.3 Water resource saving

In the **MSCC extension scenario**, irrigation for vetch emergence was possible but was never triggered by the model that considered non water-limited conditions for this species. Irrigation on the main crops increased by $35 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ on average in the irrigated areas (Table 21). It was correlated with winter MSCC impact on water availability in the soil for the next crop ($R_2 = 0.36$, Figure 20). Thus, spring crops following winter MSCC was more irrigated to avoid water stress in irrigated areas. At the same time, drainage decreased by an average of $15 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ on all cropping systems without livestock. It was unchanged by MSCC extension in time but decreased by $30 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ where MSCC were inserted. This decrease was mainly correlated to the biomass of faba bean but also a little to the biomass of summer MSCC ($R_2 = -0.65$ and -0.23 respectively against 0.17 for ryegrass). Therefore, drainage decreased strongly in the faba bean areas of southwestern France and Alsace and to a lesser extent in the vetch areas of central France (Figure 35c).

In the **ECC insertion scenario**, wherever irrigation was possible the model provided $30 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ to sorghum to facilitate its emergence. This represented an increase of only $3 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ on

the irrigated area due to the frequency of sorghum in crop rotation. Irrigation on the main crops (corn, wheat, sunflower) increased by $46 \text{ mm.ha}^{-1}.\text{yr}^{-1}$, more than in the previous scenario. Drainage decreased by $13 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ over the entire non-irrigated and irrigated area (Table 21). The decrease in drainage and the increase in irrigation on the main crops seemed to be much more correlated with the insertion of faba bean than with the insertion of barley or sorghum ($R_2 = -0.77$ and 0.55 for faba bean, 0.16 and 0 for barley, -0.18 and 0.06 for sorghum, Figure 21). Therefore, faba bean would possibly consume more water before and during the drainage period than these other two species.

In the **ECC extension scenario**, as in the previous scenario, wherever irrigation was possible the model provided 30 mm.ha^{-1} to sorghum to facilitate its emergence. But the impact on the average annual irrigation on the irrigated area remained $3 \text{ mm.ha}^{-1}.\text{yr}^{-1}$. Irrigation on the main crops increased by $62 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ compared to the baseline. As in the MSCC insertion scenario, it was the insertion of winter cover crops before grain corn, in this case barley, that increased irrigation on the following crop. Drainage changed little compared to previous scenarios despite the additional area covered by cover crops; it decreased by $14 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ compared to the baseline (Table 21). It means that on the surface already covered but where the MSCC species changed for an ECC species, drainage decreased. It was much more correlated with the frequency of ECCs insertion than with the biomass produced by these ECCs ($R_2 = -0.68$ against -0.25 , Figure 22).

5.3.2.4 Reduction of groundwater eutrophication

The **MSCC extension scenario** little reduced nitrate leaching on average ($-0.8 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ in all modified cropping systems) although the insertion of MSCCs reduced it by $4.3 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ or 11% of the amount leached in the baseline (Table 21). Extending the MSCCs duration did not reduce leaching because the cover crops were already established at the right time to take up mineral N before the drainage period. The leaching difference was equally correlated with the drainage difference and the N uptake of MSCCs difference ($R_2 = 0.55$ and -0.49 , respectively, Figure 20). Leaching was significantly decreased in the southwest where drainage was significantly decreased (Figure 35d). Concerning N uptake before the drainage period, mustard in the baseline scenario was more effective than ryegrass. Replacing mustard with ryegrass in northern France tended to increase leaching ($R_2 = 0.47$).

The **ECC insertion scenario** slightly increased leaching compared to the baseline (+1.8 kg N.ha⁻¹.yr⁻¹ i.e. +5%, Table 21). This increase in leaching was surely due to the increase in fertilization to which it was highly correlated ($R_2 = 0.51$, Figure 21). In the west (Brittany and Basse-Normandie), where mineral fertilization had increased the most, we observed an increase in leaching (Figure 35d). Furthermore, as with ryegrass, replacing mustard with barley increased leaching ($R_2 = 0.46$).

The **ECC extension scenario** slightly decreased leaching by 2.5 kg N.ha⁻¹.yr⁻¹ or 6% of leaching in the baseline (Table 21). Drainage was still highly correlated with leaching but mineral fertilization was much less correlated than in the previous scenario ($R_2 = 0.44$ and 0.13, respectively, Figure 22). The increase in the frequency of ECC export may have been able to offset the increase in ECC fertilization. Species selection was also found to have an influence. As with the previous scenario, replacing mustard with barley remained correlated with increased leaching ($R_2 = 0.40$). However, by comparing all scenarios, barley was more effective than faba bean used in the MSCC extension and ECC insertion scenarios (Figure 35d). Finally, this ECC extension scenario revealed the effect of soil depth on the leaching difference; the deeper the soil the more the scenario decreased leaching compared to the baseline ($R_2 = -0.39$).

5.4 Discussion

5.4.1 A source of biomass for gas production

Our methodology enabled us to estimate for the first time, on 75% of the French arable land, the potential production of ECC associated with their environmental impacts by taking into account cropping systems and pedo-climatic conditions at high resolution. Depending on the assumptions of profitability threshold, cutting height and modifications of the cropping systems, ECC could provide between 4.4 and 31.1 Mt DM.yr⁻¹ i.e. 16.7 TWh.yr⁻¹ to 115.1 TWh.yr⁻¹ of gas injected into the gas network. The most impactful assumption concerns the more or less widespread adoption of ECCs, i.e. the difference between the ECC insertion and ECC extension scenarios. Production increases by a factor of 3.5 when cropping systems are modified to insert even more ECCs. In a previous study of ADEME et al. (2013), with assumptions close to our first ECC insertion scenario (no change in crop rotation, winter ECCs before spring crops,...), yield calculated with an amount of biomass produced per month by

region, and a mobilization rate (i.e. harvested biomass/ total biomass ratio) of 30% for winter ECCs and 10% for summer ECCs they estimated a harvestable production of about 5 Mt DM, close to our estimation. In a more recent study by Solagro (2016), in which they considered a much larger area with summer ECCs than winter ECCs and a harvest threshold at 4.5 t DM.ha⁻¹, they calculated a harvestable production of 18 Mt DM, i.e. the production obtained in our second ECC extension scenario. Even though they introduced summer ECCs on a large area, they considered that two third was never harvested and that winter ECCs were harvested three years out of five, that is consistent with our findings. In the latest available study (ADEME 2021), the estimate was of the same order of magnitude: from 12 to 22.8 Mt DM that could be harvested depending on the area sown and whether or not fertilizer was used. To our knowledge, no study has made an estimate close to the 31.1 Mt DM that we obtained with the most favorable assumptions. With these assumptions, the mobilization rate of cover crop biomass (all species) is 73%, much higher than the 30% of previous studies or our 25% with the most conservative assumptions. The assumption of 1 t DM.ha⁻¹ of stubble restitution at harvest is more likely than the assumption of 33% of aboveground biomass restitution (expert knowledge). Thus, the low range of our estimate would more likely be 8.2 Mt DM. On the other hand, the assumption of a profitability threshold at 4 t DM.ha⁻¹ is very optimistic. It is difficult to predict what will happen to this threshold in the years to come, but the decrease in the biogas feed-in tariff at the end of 2020 (Decree n° 2020-1428 of November 23, 2020) is not favorable to a lower profitability threshold yield. If we keep the profitability threshold at 5 t DM.ha⁻¹ as it is today, the maximum biomass we could harvest is 29.9 Mt DM against 31.1 with a 4 t DM.ha⁻¹ threshold, showing that the profitability threshold therefore has little impact on overall potential production of biomass for biogas.

In our study, the potential was based entirely on winter ECCs since summer ECCs were not productive enough contrary to the study of Solagro (2016). Indeed, summer ECCs are more risky because they are sown during a dry period that could lead to repeated crop failure and this problem could increase with climate change. Irrigation could solve the problem of water stress but is not encouraged by the public authorities or technical institutes due to water scarcity and quantitative management issues and restrictions during summer (Mazzega et al. 2014). We can nevertheless estimate that the production of summer ECCs could be underestimated compared to reality because of the method we used. First, the sowing date of

summer ECCs is crucial for their success; the earlier it is, the more likely the cover crop will produce biomass (Marsac et al. 2019). To date, STICS does not allow to finely adapt the sowing date of the cover crop to the harvest date of the previous crop at such a large scale with high resolution approach. The second improvement for future estimation is the calibration of sorghum, which has not yet been tested extensively and which showed a poor response to fertilization in Chapter 2. These improvements could lead to a more realistic prediction of summer ECC yield. For winter ECCs, we showed that changing species from classic mustard, rye-grass and faba bean and fertilizing them increased a lot the biomass production. The biomass of the latter was in the expected range for MSCCs since they rarely exceeded 5 t DM.ha⁻¹ (Launay et al. 2022). The ECCs were also in the average of the values already observed in France with the same simulated variability (Marsac et al. 2019). Slight changes in cropping systems, such as advancing the earliness of corn, could greatly increase their production from 8 to 9 t DM.ha⁻¹ on average. The potential of harvestable barley biomass was mainly located in the western half of France (from Normandy to Midi-Pyrénées). When the crop sequences were modified to insert it more widely, another pool appeared in the Center and North of France. On the other hand, barley provided very little biomass in the East, where the climate was less favorable.

5.4.2 Competition with food crops?

Summer cover crops did not reduce the yield of winter crops while winter cover crops, whether MSCC or ECC, tended to reduce the yield of spring crops. This result is maybe a bit pessimistic compared to the literature, for example in the meta-analysis of Marcillo and Miguez (2017) on corn and the meta-analysis of Tonitto et al. (2006) on corn and other spring crops who showed that legume cover crops could increase the yield of the subsequent crop in low fertilized systems and that grasses didn't have any impact. We hypothesized that the yield decrease was probably due to pre-emptive competition for water and nitrogen (Thorup-Kristensen et al. 2003; Blanco-Canqui et al. 2015). Yield of irrigated corn was unaffected by faba bean while rainfed corn yield decreased. The increase of irrigation after faba bean is consistent with the hypothesis of a water competition between this legume cover crop and the following main crops. When winter barley preceded corn in the ECC extension scenario, corn yield was slightly reduced even with increased water inputs, supporting the hypothesis of induced N stress as well. Thus, the insertion of legume cover crops would cause mainly water stress on the

following crops, and the insertion of non-legume cover crop could also cause N stress. The absence of N stress after a legume cover crop is well documented in the literature (Thorup-Kristensen et al. 2003; Tonitto et al. 2006; Marcillo and Miguez 2017). On the other hand, we might have expected ECC fertilization to offset the N export at harvest (79 and 103 kg N.ha⁻¹ on average for both ECC scenarios). Few studies examined yield following a fertilized cover crop, but Szerencsits (2014), in a field experiment, did not observe a decrease in yield of the spring crop if the ECC was properly fertilized. So either the ECC fertilization was not sufficient as it was slightly below what is recommended in some studies (Szerencsits 2014; Marsac et al. 2019) or the fertilization of the following crop was too much reduced. Apart from the difference between irrigated and non-irrigated areas, we did not observe strong regional disparities in the impact of cover crops on the yield of main crops. In the ECC extension scenario, the yield loss was also due to the delay of the sowing date and the earliness of corn that have a lower yield potential. Advancing the earliness of corn and sunflower increased the frequency of insertion of ECCs and their growing period. Sunflower and grain corn production did not seem to suffer much from it, but forage corn production did. Marcillo and Miguez (2017) observed in their meta-analysis that the insertion of a cereal cover crop affected more forage corn than grain corn. In several studies they analyzed, the yield of forage corn, unlike grain corn, decreased due to the delay of its sowing date or the late destruction of the cover crop. Sunflower was the crop that lost the most yield with the insertion of long winter cover crops in our study. Preceding it with a barley harvested just before sowing in the ECC insertion scenario rather than with ryegrass terminated one month before sowing in the MSCC extension scenario had a more deleterious effect. The date of termination of the cover crop appears to be a determining factor on the growth of the following crop according to the literature (Alonso-Ayuso et al. 2014; Acharya et al. 2017). The fact that sunflower reacted more strongly than corn was not expected. Sunflower is a more resilient crop than corn, less sensitive to water stress (Debaeke et al. 2021) so STICS probably over-estimates the impact of cover crops on its yield due to a less robust calibration of this crop in the model. Finally, yield loss following a cover crop may be due to a decrease in population density because of interference of cover crop residues with soil-seed contact (Kaspar and Bakker 2015), because of a decrease in seedbed temperature (Dabney et al. 2001), or because of allelopathic effects that slow down or inhibit germination (Reberg-Horton et al. 2005) but the STICS model does not simulate these processes.

5.4.3 Environmental impact of cover crops

We chose to remove livestock systems (i.e. 6 Mha) from the environmental analysis because the introduction of AD on a livestock farm would have treated livestock manure in addition to ECCs. Consequently, fertilization practices would have been different from those simulated (e.g. better storage practices, different product characteristics, different application periods). It is likely that the impacts on the N cycle and GHG balance would have been different from those simulated (Amon et al. 2006; Rodhe et al. 2015; Vergote et al. 2019). Our method therefore needs to be refined to better take into account the changes induced by anaerobic digestion in livestock systems.

In terms of GHG balance, both the ECC insertion and extension scenarios achieve better results than the MSCC extension scenario due to the substitution of fossil gas and due to the higher SOC storage. They reduced the GHG balance of the baseline by 28 and 51% respectively on the modified arable area without livestock with the most conservative assumptions on the production of biogas. For the second scenario, it is as much as found in Launay et al. (2021a) with the insertion and extension of temporary grasslands, the extension of MSCCs and the mobilization of more organic fertilizers (i.e. food waste and green waste) combined (-54%). It could contribute up to 2% of France's 2050 C-neutrality target (savings of 8 Mt CO₂e out of the 378 Mt CO₂e required) if applied on the entire arable area without livestock. The GHG emissions from the AD process were rather high but did not make the balance negative. The management of the digester is an important factor in the variation of the GHG balance (Malet et al. 2023). Not covering the digestate could have increased GHG emissions during storage by a factor of five, and increasing methane leakage from 1% to 5% could have doubled the balance. The fate of the biogas is also a significant variation factor in the GHG balance. Malet et al. (2023) found that valorizing the biogas by injection as in our study rather than by co-generation was much more effective in reducing GHG emissions. Our results on SOC storage depend on an assumption of high stubble restitution after the harvest of ECCs. With the other hypothesis, the quantity of stubble returned would be lower, thus reducing SOC storage, but the quantity of biomass harvested would be higher, thus increasing fossil gas substitution. In terms of GHG balance, the second option would be better since 1t DM.ha⁻¹.yr⁻¹ exported from ECCs was equivalent to about 700 kg CO₂e.ha⁻¹.yr⁻¹ avoided by the substitution of fossil gas against about 200 kg CO₂e.ha⁻¹.yr⁻¹ stored in the soil if buried into the soil.

Legume MSCCs inserted in the MSCC insertion scenario reduced fertilizer use up to 41 kg N.ha⁻¹.yr⁻¹ for vetch and up to 65 kg N.ha⁻¹.yr⁻¹ for faba bean over 30 years. Tonitto et al. (2006) reported that in half of the studies examined, legumes MSCC provided between 50 and 150 kg N.ha⁻¹ to the next crop. We are thus rather in the low average knowing that faba beans were sown in late autumn almost every year but not vetch. Insertion of legume MSCC did not significantly reduce leaching in our study which is a bit surprising. Although legumes are the least effective species for reducing leaching, they still tended to significantly reduce leaching compared to bare soil in previous studies (Tonitto et al. 2006; Tribouillois et al. 2016a; Couëdel et al. 2018c). In addition, the ryegrass that replaced the mustard to cover the soil longer was less effective than the mustard which is one of the most effective species thanks to its rapid development and its high capacity to absorb N (Thorup-Kristensen et al. 2003; Couëdel et al. 2018c). The ECCs reduced the 30-year average leaching if the export frequency was sufficient to compensate for their fertilization. On the contrary, when too much N arrived at the field without being exported because the ECC did not exceed the profitability threshold then the risk of leaching increased. They were more effective than the faba bean in the MSCC extension scenario but less effective than the mustard in the baseline. Heggenstaller et al. (2008) also observed lower levels of leaching with winter ECC than with bare soil despite the higher levels of nitrogen fertilizer usage across the crop succession (+80 kg N.ha⁻¹.yr⁻¹).

Ammonia volatilization was driven by the amount of mineral fertilizer applied. The digestate input had a lower impact than expected. The composition of the simulated digestate was quite particular since it was very poor in NH₄⁺ and therefore not very sensitive to volatilization compared to other digestates or EOMs. In terms of N mineral content, our digestate was closer to a solid ECC-based digestate than to a raw digestate. The expected N-NH₄⁺/total N ratio of a raw ECC-based digestate is around 41%-68% instead of 10% on average in our study (Levavasseur et al. 2022b). Optimistic results have certainly been obtained regarding volatilization following digestate application. According to field experiments, raw digestates from co-fermented or mono-fermented crops would lose 6 to 29% of their N-NH₄⁺ content (Launay et al. 2022). We may have reached the limits of the SYS-Metha model since all the digesters used to calibrate the model were primarily processing pig or cattle slurry. The N-NH₄⁺ content of the digestate output by the model being very sensitive to the dry matter content of the substrate entering the digester; digesting cover crops that have a DM content

of at least 25% will increase the N-NH_4^+ content compared to digesting slurry (Bareha et al. 2021b).

The impact of cover crops on drainage was the same under all cover crops scenarios even though the biomass produced by the cover crops was different. This result is consistent with the meta-analysis of Meyer et al. (2019) in which cover crop biomass did not explain the variability in drainage reduction. Conversely, in our study, the reduction in drainage was highly correlated with the frequency of insertion of winter cover crops in the rotation and the species used. The observed reduction of $15 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ was close to the average of $27 \text{ mm.ha}^{-1}.\text{yr}^{-1}$ calculated by Meyer et al. (2019). Groundwater recharge could be therefore reduced by the insertion of cover crops, especially if water withdrawals increased in addition. Our study shows that in the irrigated area, the insertion of MSCCs and ECCs increased irrigation on the main crops while irrigation at emergence of summer cover crops had a negligible impact on the total irrigation water amount. However, Tribouillois et al. (2022), using an integrated model called MAELIA, showed that despite the decrease in drainage due to cover crops extension, they induced a low impact on water flows of the main rivers of the studied watershed.

5.5 Conclusion

The modeling chain implemented in this study allowed us to finely evaluate the quantity of biomass and therefore biogas that ECCs could produce in France associated with their environmental impacts. This biomass production was significant since it could provide 4 to 27% of our gas needs, mainly in the west and center of France. It was accompanied by a reduction of nitrate pollution of water and C storage in agricultural soils, but also by negative side effects such as a slight decrease in food production (-9 to -18% of spring crops yields), an increase in water consumption and a probable decrease in groundwater recharge, an increase in the consumption of mineral fertilizers and air pollution. In addition, the other way of developing cover crops, which is to develop MSCCs, does not have as many negative impacts but does not allow to store as much C in soils nor to decrease GHG emissions as much.

Learnings

In this chapter, for the first time in this work we calculated the GHG balance of the AD of ECCs. The results are positive positive, since the AD of ECCs would reduce GHG emissions from field crop systems by at least 28%. By taking more optimistic assumptions on the mobilization of biomass, the emissions could decrease by more than 51%.

The insertion of ECCs stored more SOC than the catch crops currently sown or even than the MSCCs developed all over the territory thanks to the much higher biomass production.

The high biomass production was accompanied by an increased use of mineral fertilizers. This resulted in increased N losses in the form of NH_3 and N_2O . Leaching did not necessarily increase if the N absorbed by ECCs was exported regularly enough.

Irrigation of the ECCs had very little impact on overall water use given its small weight relative to the amount of water applied to the main crops. On the other hand, the water consumption of the ECCs caused water stress on the following crops, which was compensated by an increase in irrigation, which was substantial. The other negative impact of the ECCs' water consumption was the decrease in drainage. Despite their higher biomass production, the decrease was not higher than with the insertion of MSCCs. The species used had a stronger influence than the amount of biomass produced.

As a result of the water stress but also the N stress caused by ECCs, the yield of the spring crops decreased. The MSCCs also reduced the yield of spring crops, but to a lesser extent, probably due to the use of legumes and the earlier destruction of grass cover crops.

General discussion

6.1 Objectives and approach

Anaerobic digestion (AD) has been developing in France since the 2000s in livestock farms to treat their effluents in co-digestion potentially with other organic waste from the region. Since a few years now, AD has also been seen as a solution for treating biowaste and is being considered by cereal farmers as an additional potential income by using energy cover crops (ECCs). ECCs are cover crops sown on bare soil or in place of multi-services cover crops (MSCCs), harvested and managed in such a way as to produce a lot of biomass. They raise several questions that we have addressed in this thesis:

- What is the potential production of these ECCs on a national scale and thus potential biogas production assuming that i) farmers do not change their crop successions or ii) change them to increase the biomass production potential of ECCs?
- What are the consequences of replacing bare soils or MSCCs with ECCs on environmental services and disservices centered on C, N water cycles and main crop yields?
- What are the advantages and disadvantages (for society and for farmers) of the AD of ECCs compared to other types of AD: livestock effluents, food waste?

These questions raised methodological issues:

- At which scale should we study these impacts to take into account what happens in the field and in the biogas plant?
- How to study these impacts? What is the interest of modeling and coupling of models?

After a review of the grey and scientific literature concerning the impacts of MSCCs, ECCs and fertilization with digestate, we made the following hypothesis also summarized in Figure 23:

- ECC with the return of digestate would store as much carbon in the soil as MSCC left in place. With the additional gas production that would meet a good part of the gas needed in a decarbonized France, ECCs would improve the GHG balance of French agriculture.

- The insertion of fertilized ECCs is expected to increase overall N consumption and NH_3 volatilization associated. The use of digestate should create additional NH_3 volatilization.
- Exporting biomass from ECCs and returning it in a more stable form would decrease N_2O emissions compared to MSCCs left in place.
- Even if fertilized, the ECCs would reduce nitrate leaching compared to bare soil and compared to MSCCs due to higher biomass production and residue export.
- ECCs should decrease drainage in the same way as MSCCs compared to bare soil but irrigation of ECCs could increase pressure on the water resource.
- ECCs are more likely to reduce the yield of the following crop than MSCCs because of their late harvest, but more importantly because of changes in farmers' practices who would sow their crops later with earlier varieties.

To test these hypotheses, we implemented an approach combining experimentation and modeling. The purpose of the experimentation was to quantify in the laboratory and in the field, the services and dis-services reported in the literature synthesis in order to consolidate it. On the one hand, modeling provided access to flows not measured in the experiments, on the other hand it allowed to considerably broaden the spatial and temporal scale of the study. The different steps of the approach are shown in Figure 24.

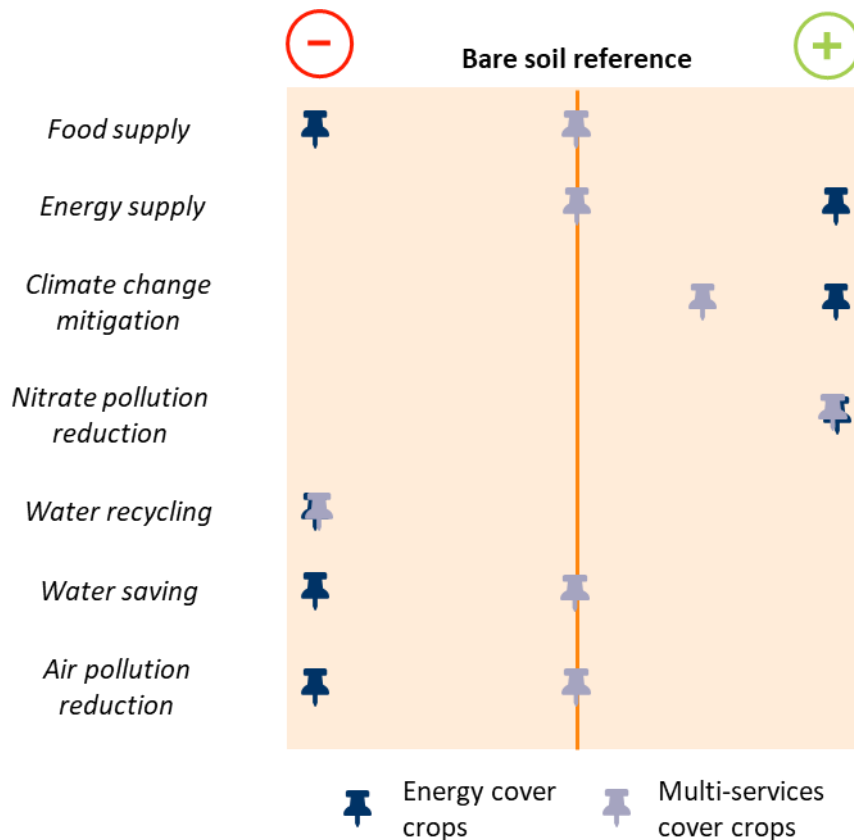


Figure 23. Synthesis of the results of the review on the effect of ECCs and MSCCs on different ecosystem services. The food supply service is measured with the yield of the main crops, the energy supply with the methane produced by ECCs, the climate change mitigation with the GHG balance, the nitrate pollution reduction with the nitrate leaching, the water recycling with the drainage, the water saving with the irrigation water consumption, the air pollution reduction with the ammonia volatilization. Cover crops are compared to bare soil (placed in the middle of the scale). When the modality is placed on the right, it means that its performance is better than the bare soil, when it is placed on the left it means that its performance is worst. The most extreme result of either modality serves as the upper or lower bound against which the other modality is placed.

6.2 Reflections on the approach used in this thesis

6.2.1 Experimental-modeling coupling

Field (Chapter 2) and laboratory (Chapter 3) experimentation were useful for exploring in detail some of the processes affected by ECC AD. Thanks to the lab experiment, we highlighted that the conversion of ECCs to digestate did not store as much C in the soil as the direct return of the same ECC biomass into the soil. We would have only seen the increase in SOC storage associated with the introduction of ECCs in the modeling approaches (Chapter 4 and 5).

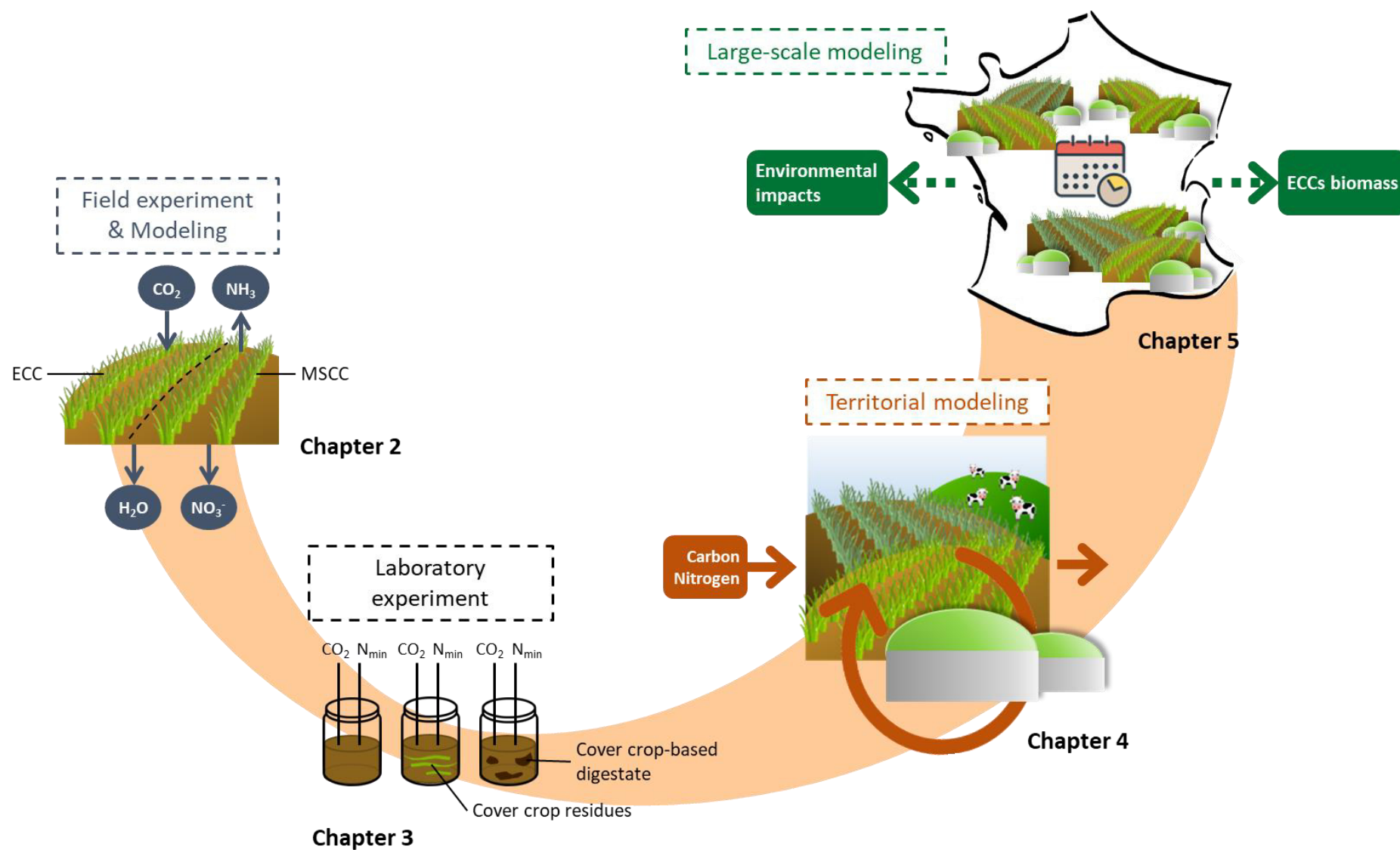


Figure 24. Schematic representation of the approach of the thesis with the division into chapters.

Simulating our experiment on the STICS model (Chapter 3) was a way to both validate the calibration of the model and to complete the measurements made in the field. The faba bean and mustard already calibrated on STICS gave good results with respect to the variables of interest (biomass produced, quantity of N absorbed, N and water stocks in the soil). Rye, which we re-calibrated from the literature and from the experiment, was satisfactorily simulated on all these variables except for the amount of N absorbed, which was underestimated. The calibration of sorghum was not satisfactory enough, we reproduced well the impact of climate variability but we failed to reproduce the response to N observed in the field. For the variables extrapolated by the model from field measurements (leaching, drainage, evapotranspiration), the good consistency between these extrapolations and the field measurements gave us confidence in the model results. Only the impact of long winter cover crops on sunflower was overestimated in simulations compared to reality. In fact, sunflower in STICS was calibrated on few varieties and in a rather limited database of situations, leading to this discrepancy with field results. It is consistent with the findings of Constantin et al. (2015) that STICS tended to overestimate the effect of water stress on yield for this crop, particularly in dry climate. Sunflower is a rather resilient crop in front of stresses and our observations in the field tended to validate that.

We chose not to use the STICS model to estimate ammonia emissions following digestate applications, but we relied entirely on the model for N₂O emission estimates. Our choice not to validate the model predictions on N₂O emissions with field measurements is questionable. However, the ability of the model to simulate N₂O emissions had already been tested on the same experimental site (Plaza-Bonilla et al. 2017). The authors found that the performance of the model was good.

Unfortunately, we were only able to conduct one field experiment over two years. Our results would benefit from being consolidated by additional experiments in various pedo-climatic contexts. This would also be an opportunity to explore other ecosystem services that could provide ECCs, such as erosion protection in sloping plots. The RECITAL trial network could provide good working material to continue the analysis. It would also allow us to consolidate our calibration of sorghum and rye and to calibrate a larger number of ECC species on STICS, or even mixtures to estimate potential production with more adapted species to local climate conditions.

6.2.2 Deployment of modeling at other scales

The three models STICS, SYS-Metha, ALFAM2 fit well together. We coupled them in two different ways: in series (Chapter 4) or integrated and completed with N balance fertilization model (Chapter 5). In series, it enabled to have a simpler device but required to know in advance all the input parameters of the three models. Integration gave flexibility in the model inputs, as some inputs were calculated as the simulation progressed, but it increased the complexity of the simulation chain. To be able to parallelize the numerous simulations of the whole France plan, we put aside the idea of simulating territorial biogas plants which would mobilize all the resources available on several km². Contrary to the series, this integrated approach enabled us to answer the question: what is the environmental impact of the AD of ECCs? but not the question: what is the environmental impact of AD? In the long-term simulations (Chapter 4 and 5) we adjusted the mineral fertilization of crops over time by taking into account the dynamics of soil organic matter. The failure to take into account the increase of SOM stock on soil N supply was a limitation raised by Tribouillois et al. (2018) and Launay et al. (2021) in the long-term simulation studies. By better taking into account the N dynamics, we have improved the predictions on N₂O emissions and leaching.

When the STICS model was deployed on a large scale (Chapter 5), the shortcomings noted when the model was compared to field experiments were found again, i.e. the poor response of sorghum to fertilization and the over-sensitivity of sunflower to water and N stress. Thus, in this approach we often touched the limits of validity of the models. This was the case for STICS but also for ALFAM2 and SYS-Metha in some way. ALFAM2 was used to simulate the volatilization of organic fertilizers, including viscous and solid fertilizers, even though the model was not calibrated specifically for them (Chapters 4 and 5). The alternative option was to use emission factors aggregated by fertilizer type by the European Environment Agency which would have surely overestimated the emissions. We considered the first option to be less false than the second option, because even if the type of product applied was not accurately taken into account, the weather and the method of application were. By using SYS-Metha to simulate 100% ECC-based digestates (Chapter 5) we also went a bit outside its range of validity since no digester digesting only ECCs was used to calibrate it. However, it was calibrated on digesters that co-digested ECCs with other materials. Thus the characteristics of the ECC and livestock

manure-based digestate in Chapter 4 were consistent with the characteristics of digestates found in the literature (Möller and Müller 2012; Guilayn et al. 2019; Levavasseur et al. 2022b).

Large-scale simulations require some simplifications. Already at the scale of the Versailles plain (Chapter 4), we were not able to represent all the diversity of crop rotations and practices of the territory. Thus, the simulated cattle farm is not directly comparable to the real cattle farm and the results obtained did not intend to help the farm in its transition. Even more simplifications were made at the scale of France for cropping systems but also for soil description for instance (Chapter 5). The aim was not to be accurate on each point on the map but to highlight global trends over time by zone or region given a large diversity of climate, soils and crop rotations representing the one encountered in France. This work would therefore be useful in helping to elaborate a strategy for the development of AD rather than for agricultural advice for a specific farmer.

Our work could therefore be improved on technical aspects, such as the calibration of some species in STICS, a better chaining in STICS to be able to trigger the end of a simulation after the crop has been harvested and sow the next crop as close as possible to the harvest, or the integration of AD in livestock systems. On the data aspect, the work could also be improved by updating the cropping systems and current agricultural practices. The most recent databases mobilized dated back to 2012. Since then, the LPIS has continued to be filled out and the 2017 Agricultural Practices Survey has been released but it remained a time-consuming work to update this information in the database used for simulations.

6.3 Potential production of ECC biomass and energy

According to our work, **the amount of energy that ECCs could produce is considerable since, going from very conservative assumptions to assumptions of strong mobilization of ECCs, it would range from 17 to 115 TWh** lower heating value (LHV). This represents from 4 to 27% of the French gas consumption in 2021 (419 TWh LHV; GRDF et al. 2022). The previous studies are rather in the low range of our estimates: 8 TWh LHV (ADEME et al. 2013), 46 TWh LHV (ADEME 2018b), 30–55 TWh LHV (ADEME 2021), 36 TWh LHV (WWF France 2022), 27–43 TWh LHV (France Stratégie 2021), 6–33 TWh (Agrosolutions and EIFER 2022). In these studies, the areas of ECCs were all calculated from the 2010 Agricultural census of the Statistics and Forecasting Service of the Ministry of Agriculture and Food. With the exception of the study

from ADEME et al. (2013) which attempted to better account for the fallow period, the other studies have taken all spring crop area as potential winter ECC area and all winter crop area as potential summer ECC area. Then, assumptions about ECC adoption and crop rotation modifications altered the area of ECCs considered in each study. By calculating the area of ECC from the main rotations by zones of a few km² and by applying decision rules on the date of harvesting of the previous crop and sowing of the next one, we obtained areas close to the first ADEME study, i.e. lower than in the other studies overall. The yields considered and the biomass mobilization rates were very variable from one study to another and chosen coarsely by expertise. Our work helped refine the achievable yields and failure frequencies. Our sensitivity analysis to the assumptions of amount of biomass left in place and threshold yield clarified what the studies were implying by their "mobilization rate" which ultimately had a strong influence on the mobilizable resource.

6.4 Agro-environmental impacts of ECCs

6.4.1 SOC storage

Starting with equivalent plant biomass, the AD of ECCs did not maintain the same SOC stocks as if their biomass was left in place in the field (Chapter 3). The AD of ECCs could only increase SOC stocks if the ECCs provided significantly more C than the initial situation. These results are different from the results of Thomsen et al. (2013) and Béghin-Tanneau et al. (2019) who nonetheless used a method close to ours. With a C storage difference of 14% in favor of plant raw biomass, the first ones concluded a similar level of storage. With a difference in storage of 27% this time to the benefit of the crop-based digestate, the latter concluded to an advantage of the digestate. The study by Levavasseur et al. (2023a), which did not use incubations but models, came to the same conclusion as ours. At the France scale, the SOC storage observed by switching from the current cropping systems or MSCC extension scenario to the ECC introduction scenario could therefore be explained by the increase in biomass production (Chapter 5). The biomass produced by the cover crops was much higher in the ECC introduction and extension scenarios than in the baseline and MSCC extension scenario. This additional biomass production more than offset the C loss in biogas. On the other hand, if we abandoned the organic amendments currently used, the ECC AD and use of digestates alone would not be sufficient to maintain the SOC stocks (Chapter 4). The territory

studied, the plain of Versailles, included field crop farms using mineral fertilizers and field crop farms using livestock manure or green waste compost. The results for the farms using mineral fertilizers were in line with the results for the same farms retained in the France-wide simulations. On the other hand, on field crop farms using livestock manure or compost, the abandonment of these amendments in favor of the digestate destored C despite the insertion of ECCs. The interest of this territorial study was therefore to realize that with the changes in practices and the competition for organic resources induced by AD, some farmers could be penalized in terms of SOC in their fields. In addition, some of these organic resources are hardly used as AD substrate (ligneous green waste for example) and the combination of digestate and compost use could be recommended to maintain high SOC storage.

6.4.2 N fertilizer consumption

As expected, **ECCs increased mineral fertilizer use on average over the rotation** (Chapters 4 and 5). It is explained on the one hand because ECCs were fertilized but did not valorize all the N well, as some of the N supplied was lost (directly in the field or during digestate storage through gas emissions) and not transformed into digestate. On the other hand, increased inputs to the main crops were necessary to compensate for the preemptive competition for N. Fertilizer consumption still decreased over time as SOM mineralization increased (Chapter 4). The increase in fertilizer consumption increase the dependence on synthetic fertilizers. To limit this disservice, we can consider legume ECCs, in mixture or sole crop. They can provide a significant amount of N in the form of controllable digestate thanks to AD (Chapter 4). As legumes are less efficient to reduce N leaching, they probably should be used in mixture with a non-legume to maintain both services of N leaching reduction and green manure effect. Legume species should also be chosen carefully to be able to reach the profitable threshold to produce biogas.

6.4.3 Ammonia volatilization

The development of AD related with ECCs tended to increase ammonia volatilization responsible for the formation of fine particles in the atmosphere, and it was more related with the increase in fertilizer consumption rather than digestate spreading (Chapters 4 and 5). Although digestate was more sensitive to volatilization than mineral fertilizer (Chapters 2 and 4), it appeared that it was the increase in fertilization that was primarily responsible for

the increase in volatilization (Chapter 5). The volatilization of digestate can be reduced by choosing a period of application less favorable to volatilization (cold and rainy weather) and by injecting it directly into the soil (Chapter 2). However, the volatilization of the digestate was probably underestimated in the France-wide study (Chapter 5) compared to the study on the Versailles plain (Chapter 4). The characteristics of the digestate simulated in chapter 5 were quite different from the characteristics of the ECC-based digestate simulated in chapter 4 and from a real ECC-based digestate (Levavasseur et al. 2022b) ; it was much lower in N-NH_4^+ .

6.4.4 N_2O emissions

In the France-wide simulations (Chapter 5), **the insertion of ECCs increased N_2O emissions more than the extension of MSCCs**. Our initial hypothesis, that exporting biomass from ECCs and returning it in a more stable form would decrease N_2O emissions compared to MSCCs left in place, was not always true. It was only true if the ECC replaced a cover crop whose residues were rapidly mineralized, i.e. with a low C:N ratio (e.g. mustard; Chapters 2 and 4). In the case where the residues of the cover crop immobilized N (e.g. rye), their export had the opposite effect since the soil mineral N that would have been immobilized by residue degradation becomes available for the nitrification/denitrification chain (Chapter 2). In the long-term simulations, by considering not what happens during one fallow period but during 30 successive fallow periods, another relationship appeared: N_2O emissions were highly correlated to the increase in SON mineralization (Chapter 5). This trade-off between N_2O emissions and SOM storage at the origin of the increase in SON mineralization had already been highlighted by Guenet et al. (2020).

6.4.5 Nitrate leaching

On a territorial and French scale (Chapters 4 and 5), the insertion of ECCs tended to reduce leaching compared to the baseline and the extension of MSCCs when ECC reached the profitability threshold to be harvested frequently enough. This result is in agreement with the study by Heggenstaller et al. (2008) who found that on a succession, leaching was less important with fertilized ECCs than with bare soil. During the fallow, fertilization of ECCs had no impact on leaching because it was applied after the drainage period. However, on a large scale over a long period of time with less precise control of the application period and harvest failures, it could increase leaching compared to bare fallows or fallows sown with catch crops

if the ECCs did not export all the N surplus. **The choice of cover crop species proved to be more critical than the management of the cover crops.** We saw both in the field (Chapter 2) and in the France-wide simulations (Chapter 5), that mustard was more effective than the cereal-legume mixture (rye-faba bean) and the pure cereal (ryegrass or winter barley). The pure legume (faba bean) was less efficient than the pure cereal (winter barley) in the France-wide simulations. Contrary to our initial hypothesis, more than the biomass produced at the end of the cycle, it is the speed of implantation and the N uptake capacity that were determining factors in the reduction of leaching (Thorup-Kristensen et al. 2003; Tribouillois et al. 2016a; Riau et al. 2021). Möller and Stinner (2009) who compared ECCs with MSCCs used the same species but destroyed or harvested them before the drainage period and did not fertilize the ECCs. In this study, MSCC residues mineralized prior to the drainage period. Therefore, exporting the residues and returning it as digestate after the drainage period decreased leaching. The closest experiment to this situation in our study is the comparison of the summer ECC and summer MSCC modalities in the field experiment but the ECC was fertilized (Chapter 2). The amount of N leached was the same in both modalities. Fertilization of the ECC would have perfectly offset the export of the residues.

6.4.6 Water supply and recycling

It turns out that **ECCs deteriorated the water recycling service.** Whether looking at the scale of a field for one season (Chapter 2) or of the whole France for 30 years (Chapter 5), ECCs reduced drainage compared to bare fallow. In the field experiment (Chapter 2), summer ECCs had a greater impact than winter ECCs, whereas at the scale of France, drainage was more correlated with the frequency of insertion of winter ECCs than summer ECCs. The field experiment showed a relationship between cover crop transpiration and reduced drainage. Consequently, summer MSCCs that produced less biomass and therefore transpired less than ECCs reduced drainage less than the latter. It was similar for winter MSCCs if the drainage period dragged on into the spring when growth resumed. The cover crops reduced soil evaporation but not enough to offset the increase in transpiration. These results are consistent with the study by Tribouillois et al. (2018) which took place in the same region. They found a significant correlation between cover crop biomass production, drainage reduction, cover crop transpiration and soil evaporation. When scaling up to the whole of France (Chapter 5), the cover crop species changed between MSCCs and ECCs. Thus, another factor of variation

appeared which was the species of the cover crop. Faba bean seemed to consume much more water than winter barley. As a result, even though the frequency and biomass of cover crops was higher in the ECCs extension scenario compared to the MSCCs extension scenario, the average drainage was about the same. This result may help to understand the confounding effects obtained by Meyer et al. (2019) in their meta-analysis of the impact of cover crops on drainage in which they failed to find a significant explanatory factor. In accordance with our results, we found two studies that compared water use efficiency between different summer cover crop species in dry climate and found significant differences (Wunsch et al. 2017; Berriél et al. 2022).

Winter ECCs increased the water consumption of the following spring crops if irrigation was available (Chapter 5). Winter ECCs had a tendency to cause water stress on the following crops. In irrigated areas, irrigation increased significantly on the main crops. The extension of MSCCs also caused an increase in irrigation but to a lesser extent than the extension of ECCs. STICS also simulated water stress on rainfed corn following an ECC in the plain of Versailles (Chapter 4). We chose to limit the irrigation of summer ECCs in accordance with Arvalis recommendations and main farmer's practices. Thus, the irrigation at emergence authorized in the simulations for France (Chapter 5) had no impact on the overall irrigation water consumption.

6.4.7 Food production

The winter ECCs tended to decrease yield of the following spring crop, mostly in relation with water stress but also with nitrogen stress (Chapters 4 and 5). However, we did not observe any yield reduction of sunflower in the field experiment (Chapter 2). Some resilient species could therefore be preferred after an ECC. MSCCs did not reduce spring crop yield as much (Chapter 5). First, because the ryegrass was killed early enough to limit preemptive competition, and second, because the faba bean provided N to the next crop rather than immobilized it. Winter crops did not seem to be N-stressed in the France-wide simulations nor in the field experiment. In the France-wide simulations, it is quite particular because summer ECCs did not produce much biomass, which was very rarely exported, thus explaining the low N exports. Conversely, in the plain of Versailles (Chapter 4), the summer ECCs caused preemptive competition for N with the following wheat on the deep soil. On this soil type, ECCs immobilized N that would otherwise have been quickly available in spring for the following wheat. **It is therefore likely**

that winter ECCs and summer ECCs reduce food production in return for renewable gas production. However, total biomass production including ECCs over the entire succession was increased. In addition, the reduction in main crop production did not affect SOC storage.

6.5 GHG balance and climate change mitigation service

At the scale of the whole of France, the GHG balance was only carried out on cropping systems without livestock (Chapter 5). We took into account the SOC storage, the direct and indirect N_2O emissions (NH_3 volatilization, NO_3^- leaching) in the field, the emissions induced by the production of synthetic fertilizers used in the field, the emissions linked to the operation of the biogas plant (e.g. energy consumption, biogas leaks, digestate storage) and the emissions avoided by the substitution of fossil gas by biomethane. The emissions due to the transportation of matters and new harvesting and spreading operations were not included. They have a low weight compared to the emissions already taken into account (Malet et al. 2023).

ECCs provided an effective climate change mitigation service, since they reduced GHG balance from field crop systems by 26% to 58%, depending on the ECC development scenario. The development of MSCCs did not achieve the same reduction of GHG balance. Despite the increase in direct and indirect N_2O emissions via NH_3 volatilization, the increased use of synthetic fertilizers, and the emissions induced by the operation of the biogas plant, the AD of ECCs had a positive GHG balance compared to the baseline thanks to the very important C storage and the substitution of fossil gas, which had an even greater impact.

Comparison between the insertion of ECC, the insertion of MSCC and the bare soil concerning GHG balance and other agro-environmental impacts are summarized in Figure 25. In summary, ECCs sometimes had a negative impact on the yield of the following crops that did not exceed 17% yield loss. This impact was better controlled with MSCCs. ECCs provided the energy supply service that MSCCs did not, which improved the GHG balance compared to bare soil or MSCCs. The additional C storage also improved the GHG balance relative to MSCCs more effectively than expected. Overall, nitrate leaching was similarly reduced with both cover types, but the range was rather wide and influenced by the species chosen and the duration of the drainage period. Similarly, ECCs reduced drainage more or less than MSCCs depending on the species considered, the type of cover (summer or winter) and the duration of the drainage period.

Irrigation water use by ECCs was limited and had very little impact on overall use, but water use by following crops increased. Contrary to what was expected, MSCCs also increased water consumption of the main crops due to their late termination for some. While the fertilization of ECCs and the use of digestate did increase NH_3 volatilization, the reduction in mineral fertilization following the use of legume MSCCs decreased NH_3 emissions. This last point had not been noted in the literature review.

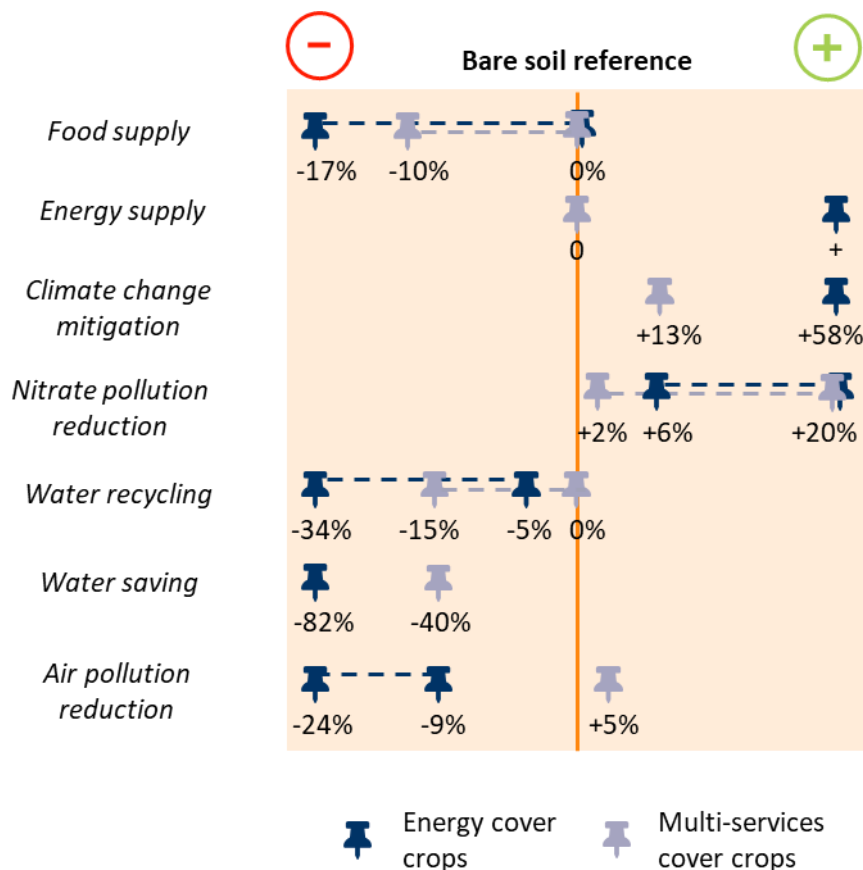


Figure 25. Synthesis of the results of the thesis on the effect of ECCs and MSCCs on different ecosystem services. The dotted lines represent the range of values for each modality. The food supply service is measured with the yield of the main crops, the energy supply with the methane produced by ECCs, the climate change mitigation with the GHG balance, the nitrate pollution reduction with the nitrate leaching, the water recycling with the drainage, the water saving with the irrigation water consumption, the air pollution reduction with the ammonia volatilization. Cover crops are compared to bare soil (placed in the middle of the scale). When the modality is placed on the right, it means that its performance is better than the bare soil, when it is placed on the left it means that its performance is worst. The most extreme result of either modality serves as the upper or lower bound against which the other modality is placed.

6.6 Interest of ECC AD in a field crop area

ECC AD had several advantages for cereal farmers compared to industrial AD which would be their main alternative in our specialized territories (Chapter 4). First, it allowed them to have the additional income from biogas and not to buy organic fertilizer. Second, it allowed them to obtain the ecosystem services provided by ECCs: SOC storage and leaching reduction. In any case, land application of digestate would emit ammonia. Food waste-based digestate was more sensitive to volatilization than ECC-based digestate. On the other hand, by digesting only their ECCs, they would not gain N and therefore would not save fertilizer. In the end, adding ECC-based digestate to the already used mineral fertilizer would increase overall volatilization, whereas food waste-based digestate would not increase NH_3 emissions as much by replacing mineral fertilizer.

6.7 Potential effect of climate change on ECC production

6.7.1 The climate change in France

All of our experiments have been performed with the current or near past climate. However, the climate is changing. It is therefore wise to ask what the validity of our results will be in the future climate, especially on the growth of ECCs.

According to the IPCC special report on the impact of 1.5°C global warming, meeting current national GHG emission reduction targets for all countries would lead to a warming of between +3 and +4°C in 2100 relative to pre-industrial temperatures (Hoegh-Guldberg et al. 2018). This is in the high range of RCP6.0 projections for 2100 (+2.0 to +3.7°C) and in the low range of RCP8.5 projections (+3.2 to +5.4°C). It is very likely that the average warming will be greater at the land surface. The increase in temperature will result in a change in the water balance, through increased evapotranspiration (Tibi et al. 2020). The consensus between models regarding future precipitation is less strong than for temperatures. In France, average precipitation is likely to decrease especially around the Mediterranean. With the RCP8.5 scenario, precipitation in France would decrease everywhere in summer and increase in the northern half of France in winter (Figure 26 ; Terray and Boé 2013). In France, if the temperature increases by 2°C and annual precipitation decreases by 5%-10%, then the flow of rivers should decrease by 10 to 40% by the middle of the century (Tibi et al. 2020). Summer low flows would be more severe and the southwest would be more affected. Irrigation capacity could be

reduced in summer. Regarding soil moisture, the 0–10 cm horizon will dry out, especially in the Mediterranean region. The country risks "drying out" (Tibi et al. 2020). Climate change will also result in more extreme events with higher intensities (e.g. intense droughts, storms, floods, heat waves). Hot extremes will be more numerous and cold extremes less numerous on daily and seasonal scales. It is very likely that heat waves will be more frequent and last longer. It is also likely that the frequency and intensity of extreme precipitation events will increase at our latitudes. The IPCC does not come to a clear conclusion on droughts, but the risk of an increase is likely in the Mediterranean basin (Tibi et al. 2020).

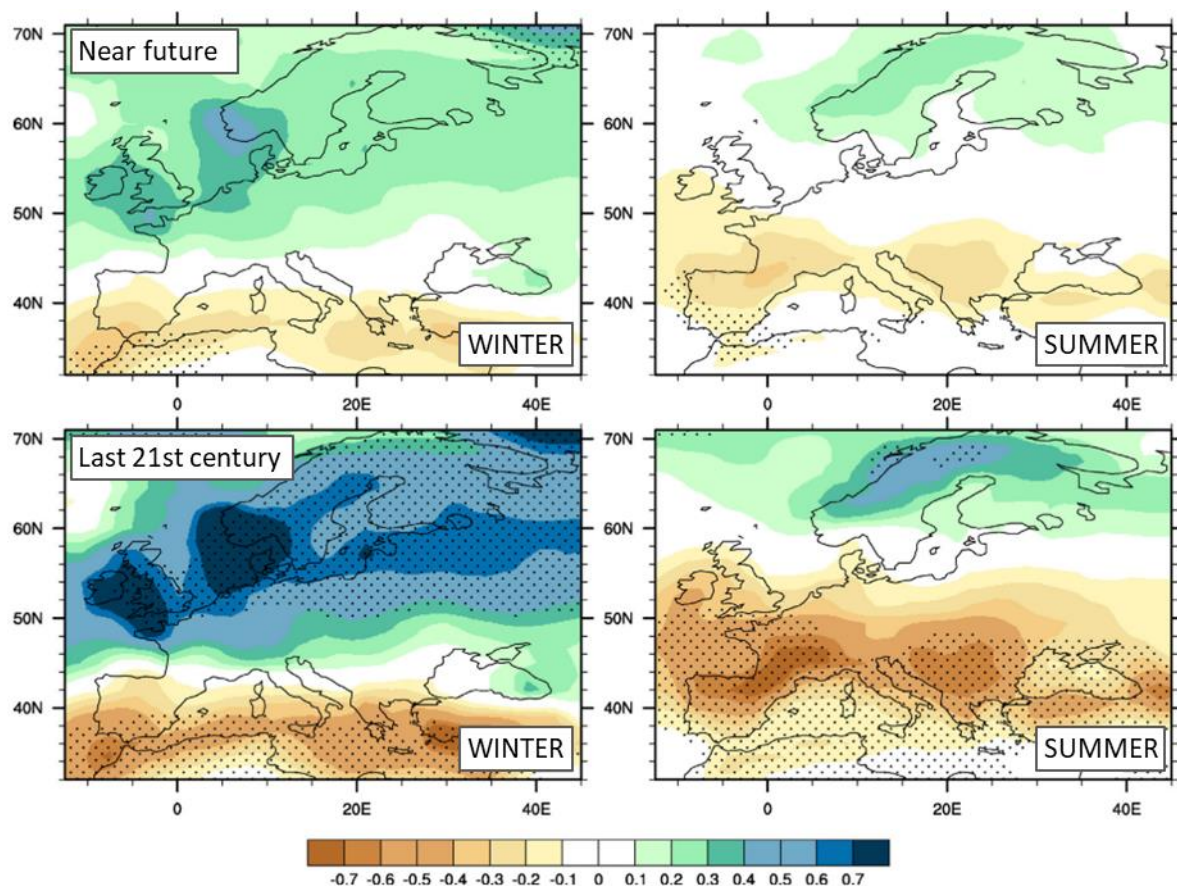


Figure 26. Seasonal precipitation changes (mm.day^{-1}) in Europe relatively to the 1900–1929 period projected in the near future (2020–2049) and in the last 21st century (2070–2099). Adapted from Terray and Boé (2013)

6.7.2 Impact of climate change on ECCs

The increase in temperature should shorten the cycle of the main crops, they will reach the amount of temperature needed to mature more quickly. The disadvantage is that they will produce less biomass because they will have less time to intercept solar radiation. The advantage is that in some situations they will be able to avoid stresses (e.g. hydric or thermal).

The other advantage is that an early harvest enables to sow summer and winter ECCs earlier. ECCs would produce more biomass and summer ECCs could possibly avoid water stress. Alonso-Ayuso et al. (2018) simulated the impact of climate change on the growth of winter cover crops sown on October 10th and killed between March 1st and April 14th in a semi-arid climate. The cover crops grew faster and stronger; biomass production at destruction nearly doubled.

Increased temperature extremes can have deleterious effects on summer ECC by causing accelerated senescence. In the study of De la Haba et al. (2014), young sunflower leaves exposed to high temperatures for several days (33°C daytime/ 29°C nighttime) compared to leaves exposed to medium temperatures (23°C daytime/ 19°C nighttime) had reduced growth and photosynthesis. High temperatures triggered early senescence through accumulation of soluble sugars and H₂O₂ and decrease in starch and antioxidant activity.

The change in precipitation patterns (less water in summer, more water in winter) would bring France closer to a Mediterranean climate. We can therefore look at studies carried out in a Mediterranean climate to get an idea of what cover crops might become in France in the future. For example, Rose et al. (2022) calculated on eight different Australian sites without irrigation that opportunities for sowing summer cover crops (i.e. seven days with more rainfall than evaporation) were rare. On half of the sites, seeding was only possible one year out of three and on the other sites seeding was never possible more than two years out of three. Moreover, even if the seeding would be successful, growth and thus biomass production would be limited by the availability of water during the lifespan of the ECC. Some flexibility could be possible by using species with better water use efficiency. For example, Japanese millet grew faster and produced more biomass while leaving more water at the end of its cycle than the legumes tested as summer cover crops by Wunsch et al. (2017).

The complement to the study of Launay et al. (2021) provides some initial answers to the question of the impact of climate change on cover crop production in France, even though it focuses on MSCCs (Constantin et al. 2020). The simulation plan was tested with the RCP8.5 climate change scenario over 2030-2060. Vetch and faba bean sown in the MSCC extension scenario were impacted differently by climate change. Without changing the sowing or destruction dates, the production of vetch in summer fallow decreased by 0.8 t DM.ha⁻¹, i.e. a

decrease of 30% compared to the current climate, whereas the production of faba bean in winter fallow increased by 0.7 t DM.ha⁻¹, i.e. an increase of almost 90%.

6.7.3 ECCs as an adaptation practice to climate change

One of the problems caused by extreme rain events is soil erosion (Kaye and Quemada 2017). These extreme rain events are expected to be more frequent in our latitudes. Cover crops that will produce sufficient biomass (>30% soil cover according to Kaye and Quemada 2017) at the time of the extreme rain event could reduce erosion. They could also reduce the leaching of nutrients carried by heavy rainfall (vertically and laterally). By providing C to the soil, cover crops improve the stability of soil aggregates, which contributes to the reduction of erosion even when the cover crop is no longer present (Kaye and Quemada 2017).

As temperatures rise, N mineralization in the fall could be increased (Kaye and Quemada 2017). Once the drainage period arrives, this surplus of mineral N would sink to the water table. ECCs could prevent this excess N from leaching by absorbing it before the drainage period. Since warming would also enable winter ECCs to be sown earlier, their efficiency would be increased because they would have time to produce more biomass before winter (Kaye and Quemada 2017; Alonso-Ayuso et al. 2018).

6.7.4 ECCs as a maladaptation to climate change

With regard to adaptation to decreased precipitation, on the one hand, living cover crops can increase water infiltration by limiting the phenomenon of soil crusting that frequently occurs in the Mediterranean zone with the alternation of heavy rainfall and dry periods (Ries and Hirt 2008). On the other hand, we have seen that the transpiration of winter and summer ECCs can decrease the amount of water going down to the water table during the drainage period. This will affect the supply of water to rivers and the amount of water available for pumping the following summer. In addition, the decrease in precipitation in the spring makes the risk of water stress on the crop following winter ECC much more likely, especially on soils with low water reserves. With MSCC, the risk is avoidable if the destruction date is early enough (at least one month before the sowing of the following crop; Alonso-Ayuso et al. 2014) but with ECCs it is difficult to consider advancing the harvest date more than a week before sowing the next crop (but if ECCs grow faster with increasing temperatures we could harvest them a little earlier). We have already observed water stress on spring crops following ECCs in chapters 2

and 3 with the current climate. Yield losses were in the range of 9-18% depending on the crop, knowing that irrigation was increased in irrigated areas to limit losses. The increase in irrigation goes against the decrease in groundwater recharge, especially in southwest France and Alsace. Regarding summer ECC, Rose et al. (2022) recommend not leaving them in place for long (one to two months) so that their transpiration does not counterbalance the reduction in evaporation and the improvement in water infiltration; but biomass production would be hindered. They highlight the risk that these cover crops penalize the establishment of the following winter crop by drying out the first soil horizon when rainfall is still scarce. However, results on this subject are still rare in the literature.

A simple perspective of our work concerning the problem of the impact of climate change would be to run the whole France simulation plan with a climate change scenario.

6.8 Future development of anaerobic digestion

6.8.1 The anaerobic digestion in the French energetic mix

AD seems unavoidable if France wants to reach the objective of C neutrality by 2050 (ADEME 2021). It is today the most mature renewable gas production technology ahead of pyrogasification and power to gas (SER et al. 2022). Even in the ADEME scenario that foresees the highest decrease in energy consumption (among all the scenarios proposed by the agency), AD will have to provide 94 TWh LHV (ADEME 2021). This is 13 times more than what it supplied in 2021 (7 TWh LHV in injection and co-generation²). Although the use of gas should decrease in the industry and building sectors, it should develop together with hydrogen in the transport sector to decarbonize heavy and long distance transport. Also, in a system where electricity production would be based mainly on intermittent sources, gas could play a key role in securing the supply of the electricity system thanks to its strong storage capacities (120 TWh LHV ; ADEME 2021). The development of the use of biomass as a source of renewable energy, among others, therefore seems inevitable (Figure 27). Agriculture's role as an energy producer will once again become central in the years to come, in addition to its central role as food producer. Coupled with the decrease in the use of fossil fuels by farmers (which represent 80%

² <https://www.statistiques.developpement-durable.gouv.fr>

of the energy consumed by French agriculture today) to respect C neutrality, the decrease in food production seems unavoidable (Harchaoui and Chatzimpiros 2019; ADEME 2021). However, food autonomy is not directly endangered since there are levers of action to continue to satisfy the demand, e.g.: reducing exports, consumption of animal products, waste (Harchaoui and Chatzimpiros 2019; ADEME 2021).

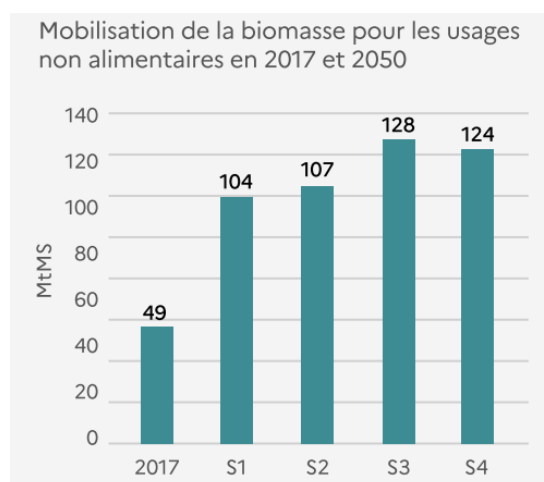


Figure 27. Prospects for the evolution of biomass use for non-food purposes between 2017 and 2050 according to the four ADEME decarbonization scenarios for France. Adapted from ADEME (2021).

The development of the sector had slowed down since Covid crisis, notably because of uncertainties about the support mechanisms for the sector. Recent developments (e.g. launch of Biogas Production Certificates, stronger encouragement from Europe towards energy autonomy following the war in Ukraine) should again encourage its development (GRDF et al. 2022).

If France wants to continue to develop AD, it has to choose the model it wishes to follow (e.g. agricultural or industrial AD, small or large units, individual or collective units). Up to now, small-scale individual biogas plants on farms are the majority (flow rate of $170 \text{ Nm}^3 \cdot \text{h}^{-1}$ on average for injection plants; SER et al. 2022). ADEME imagined two ways of developing AD (ADEME 2021). The first one, which supports scenarios of energy sobriety and territorial autonomy, relies on the generalization of small-scale collective agricultural plants (between 50 and $250 \text{ Nm}^3 \cdot \text{h}^{-1}$). A few territorial plants of a slightly larger size ($250 \text{ Nm}^3 \cdot \text{h}^{-1}$) would also develop in a short circuit logic for big cities (e.g. treatment of municipal organic waste and energy recovery in a heating network). The second pathway envisaged by ADEME concerns the scenarios of technological development and less sobriety. The plants would be fewer in number

but larger to gain in efficiency (between 150 and 350 Nm³.h⁻¹ for on-farm plants and between 350 and 400 Nm³.h⁻¹ for centralized plants). There would be no logic of short circuit or energy autonomy but a logic of economic efficiency: buying decarbonated energy abroad if it is cheaper rather than produce it locally. These two development pathways stem from different logic/ideology that needs to be arbitrated at the political level. In concrete terms, each has advantages and disadvantages. Large-scale biogas plants are more profitable than small ones (González-Arias et al. 2021) but the economies of scale that they provide (Cowley and Brorsen 2018) may lead to simplifications of systems that are dangerous for biodiversity. They are suitable for very specialized territories (Raven and Gregersen 2007; ADEME 2021), but in territories where organic resources and surfaces suitable for digestate are diverse and scattered, the distances covered reduce the energy and economic efficiency of the system (Delzeit and Kellner 2013) and logistics are more important. Moreover, social acceptability is often weaker for large plants than for small ones (e.g. strong mobilization of citizens against the Métha Herbauges project, the largest AD project in France^{3,4}). Residents mention the nuisance caused by the increased road traffic, the risk of leaks, the industrial size of the site, and the economic spin-offs that would benefit companies outside the region rather than the farmers (Anzalone and Mazaud 2021). Individual on-farm plants also have their disadvantages, such as the weight of the investment on a single person, the dimensioning of the biogas plant and the storage capacity according to the irregular supply of matter due to the fluctuations in the cover crop biomass, the difficulty of taking over the farm after the owner retires because of its cost, and the additional workload. The collective farming units put forward in the first ADEME scenarios seem to be the best compromise between the disadvantages and advantages of each plant type. Their average size provides profitability while spreading the investment over several people and companies (Raven and Gregersen 2007), the profits go back to the farmers and the plant remains close to organic resources and spreadable surfaces for digestate, the grouping enables to share the extra work or to hire labor (Raven and Gregersen 2007; Anzalone and Mazaud 2021).

³ <https://www.euractiv.fr/section/agriculture-alimentation/news/un-village-gaulois-soppose-au-plus-grand-projet-de-methanisation-de-france/>

⁴ <https://www.web-agri.fr/methanisation/article/202309/pres-de-nantes-le-plus-gros-projet-de-methaniseur-de-france-passe-mal>

Regardless of the model chosen, given the rapid pace of development that would be required, it would be better to organize this development. If the plants continue to be built in an uncoordinated way, we risk ending up with competition for biomass between several plants (Silva et al. 2017; Yalcinkaya 2020; Thrän et al. 2020). Rushing the construction of new plants will not give the local population time to accept the project, which is often a key factor in its success (Silva et al. 2017; Bourdin et al. 2019; Nevzorova and Kutcherov 2019; Tahali et al. 2021). The urgency and haste could also push manufacturers to set aside certain safety rules or the authorities to relax these rules. The governance of this development remains to be defined; we could move towards national governance by the State, which would be similar to planning, or towards local governance involving several stakeholders, including citizens (Thrän et al. 2020).

6.8.2 Transformations of agricultural systems (field crops) linked to the development of anaerobic digestion

In France, main crops cannot feed more than 15% of a biogas plant whereas there is no limit for ECCs (Article D543-292). But until recently, the definitions of main crop and cover crop were unclear. Some pointed to the risk of simplifying rotations on the German model (Lüker-Jans et al. 2017; Yang et al. 2021). The new decree no.2022-1120 of August 4, 2022 on crops used for biogas and biofuel production has clarified the definitions of "main crops" and "cover crops". Among other conditions, a main crop is now "the crop present on the plot on June 1". Cover crops are "crops grown on the territory of the European Union that are not main crops and that are sown and harvested on a plot between two main crops harvested in one or two consecutive calendar years". With these new definitions, an ECC sown before or harvested after June 1 will therefore be considered as a main crop. Generally, ECCs are sown after June 1 but the opposite can happen after a vegetable crop. Winter ECC are also generally harvested before June 1, but this threshold date limits the possibility to extend their growing period to reach more biomass production. Such late harvest date could be deleterious for the next main crop yield, such as corn or sunflower so the new regulation limits concurrence for food production. The 15% supply limit for main crops remains and no limit has been added for ECCs. This regulation therefore responds to criticism of the place of energy crops in the rotation without slowing down the introduction of ECCs. The approach could have been taken to one extreme by removing the main crops from the ration. By keeping the 15% limit, the regulation allows flexibility for energy double-cropping (Heggenstaller et al. 2008; Graß et al. 2013; Dale et al.

2016). These successions maximize biomass production over two crop cycles in one year. Since both crops are harvested before maturity, the cropping windows are more flexible. The other advantage is that the yield loss of one crop is fully compensated by the gain of the other one since the purposes of both are the same.

So far, the development of AD in France does not seem to have introduced major changes in crop rotation thanks to the regulations on energy crops (Levavasseur et al. 2023b). Most farms that have implemented AD have not changed their crop rotation. However, in order to introduce ECCs, farmers may be tempted to make changes, for example by substituting a winter crop with a spring crop to precede it with a winter ECC (Carton and Levavasseur 2022). In Chapter 5, we saw that such crop rotation changes, combined with an advance in the earliness of spring crops, could increase ECCs production by a factor of 3.5. We could not separate the effect of rotation modification from the effect of ECC insertion on the ecosystem services provided by the farming systems. Therefore, we mainly observed that the services and dis-services provided by ECCs were accentuated because of their wider implantation. For services not studied here, such as pathogen and weed control and biodiversity conservation, it is not possible to judge the impact of these changes in advance. Depending on the initial situation of the farm and the choices made, these modifications can increase or decrease the diversity of the crops. There is an interest in studying these services, which are highly desirable in organic or conservation systems. **Additional studies could be conducted on the impact on these services of the diversity of ECC species that can be used alone or in combination.**

In addition to crop rotation, the introduction of AD raises the question of the evolution of farmers' production systems. In concrete terms, does AD help farmers to make their agro-ecological transition or does it lock in their practices? Conversely, is AD chosen by conventional farmers or by organic farmers? Today, the question is not clear-cut; there are examples of conversion to organic farming with AD and examples of systems that become less virtuous (see Jeanne Cadiou's thesis). **A larger study about farmers practicing ECCs would allow us to confirm our results or, on the contrary, to nuance them if the actual practices are not the recommended ones.**

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Annexes

Annexes – Chapter 1: Supplementary Material

Table 22. Effects of using digestate as fertilizer on ammonia volatilization across different studies. +: significant increase in volatilization associated with digestate use; -: significant decrease in volatilization associated with digestate use; =: no change in volatilization associated with digestate use.

Context	Digestate substrate	Control	Application	Crop	Duration (days)	Effect on volatilization	Emission factor	Reference
Field	Food waste	Cattle + pig slurry	Surface	Wheat/ grassland	2	+	40% of total N applied	Nicholson et al. 2017
Field	Cattle slurry	Cattle slurry	Incorporation	Wheat	4	+	10% of total N applied	Möller and Stinner 2009
Field	Pig slurry	Pig slurry	Surface	Wheat	5	+	40% of N-NH ₄ ⁺ applied	Nyord et al. 2012
Laboratory	Pig slurry + food waste	Pig slurry	Surface	Bare soil	4	+	10–16% of N-NH ₄ ⁺ applied	Sommer et al. 2006
Field	Energy crops or pig slurry + energy crops	Cattle and pig slurry	Surface	Bare soil/ grassland/ wheat/corn	3	+	10–13% of N-NH ₄ ⁺ applied	Ni et al. 2012
Field	Cattle slurry + food waste	Cattle slurry	Surface/ Injected	Grassland/ bare soil	4	=	30% of N-NH ₄ ⁺ applied to surface/ 10% of N-NH ₄ ⁺ injected	Wulf et al. 2002
Field	Pig slurry	Pig slurry	Surface	Bare soil	19	=	42% of N-NH ₄ ⁺ applied	Chantigny et al. 2004
Field	Pig slurry	Pig slurry	Surface	Bare soil	10	-	18% of total N applied/ 23% of N-NH ₄ ⁺ applied	Chantigny et al. 2009
Field	Pig slurry	Pig slurry Mineral fertilizer	Surface	Grassland	8	- +	10–13% of total N applied	Chantigny et al. 2007
Field	Energy crops + pig slurry	Mineral fertilizer	Surface	Grassland/ wheat/corn	2	+	6–28% of N-NH ₄ ⁺ applied	Quakernack et al. 2012
Field	Energy crops	Mineral fertilizer	Incorporation	Corn	4	+	14% of total N applied/ 29% of N-NH ₄ ⁺ applied	Wolf et al. 2014
Field	Sewage sludge	Urea	Injected	Corn	4	=	12% of N-NH ₄ ⁺ applied	Zilio et al. 2021

Table S223. Effects of using digestate as fertilizer on nitrate leaching across different studies. +: significant increase in leaching or leaching risk associated with digestate use; -: significant decrease in leaching or leaching risk associated with digestate use; =: no difference in leaching or leaching risk associated with digestate use.

Context	Digestate substrate	Control	Application rates (using equivalents)	Agricultural practices	Assessment method	Effect on leaching	Reference
Field	Animal manure	Mineral fertilizer	N use efficiency	Applied to a grassland for 3 years	Dosage of leached ammonium and nitrate	=	Matsunaka et al. 2006
Field	Animal manure	Mineral fertilizer	Total N	Applied in spring to grain corn; repeated for 3 years	Measurement of soil mineral N in late fall or post application	-	Chantigny et al. 2008
Field	Corn/pig slurry + corn	Mineral fertilizer	Total N	Applied in spring to forage corn; repeated for 2 years	Dosage of leached ammonium and nitrate	-	Svoboda et al. 2013
Field	Animal manure + energy crops + food waste	Mineral fertilizer	Total N	Applied to a grassland for 3 years	Measurement of soil mineral N in late fall	-	Tsachidou et al. 2019
Pots	Cattle slurry	Mineral fertilizer	Total N	Applied to a grassland in two steps	Dosage of leached ammonium and nitrate	-	Walsh et al. 2012a
Field	Animal manure	Animal manure	Total N	Applied in spring to grain corn; repeated for 3 years	Measurement of soil mineral N in late fall or post application	=	Chantigny et al. 2008
Field	Animal manure	Animal manure	Total N	Applied to winter wheat; repeated for 2 years	Dosage of leached ammonium and nitrate	-	Jäkel and Mau 1999 in Svoboda et al. 2013
Laboratory	Animal manure	Animal manure	Total N	Single application to bare soil	Dosage of leached ammonium and nitrate	+	Goberna et al. 2011
Field	Animal manure	Animal manure	Fresh matter	?	?	-	Börjesson and Berglund 2007
Field	Animal manure	Animal manure	?	?	Measurement of soil mineral N in late fall	=	Pötsch 2004 in Möller 2015

Laboratory	Cattle slurry	Cattle slurry	Total N	Applied to a grassland in two steps	Dosage of leached ammonium and nitrate	=	Walsh et al. 2012a
Field	Cattle slurry	Cattle slurry	N-NH ₄ ⁺	Applied to a winter barley-winter wheat rotation for 2 years	Dosage of leached ammonium and nitrate	=	Brenner and Clemens 2005 in Svoboda et al. 2013
Field	Cattle slurry	Cattle slurry	?	Applied in spring to different rotations; repeated for 2 years	Measurement of soil mineral N in late fall or post application	=	Möller and Stinner 2009
Field	Pig slurry + corn	Pig slurry	?	Applied in spring to different rotations; repeated for 2 years	N balance	=	Sieling et al. 2013
Field	Food waste	Pig slurry	?	Applied in fall to winter wheat	Dosage of leached ammonium and nitrate	=	Nicholson et al. 2017
Field	Animal manure + energy crops + food waste	Pig slurry + mineral fertilizer	N use efficiency	Applied in spring to a forage corn-Italian ryegrass rotation; repeated for 3 years	Measurement of soil mineral N in late fall	=	Sigurnjak et al. 2017
Field	Pig slurry + corn	Animal slurry	Total N	Applied in spring to forage corn; repeated for 2 years	Dosage of leached ammonium and nitrate	=	Svoboda et al. 2013
Field	Corn	Animal slurry	Total N	Applied in spring to forage corn; repeated for 2 years	Dosage of leached ammonium and nitrate	=	Svoboda et al. 2013
Field	Animal manure (as a substitute for energy cover crops)	Catch crops	Total N	Applied in spring; 3 out of 4 years	Dosage of leached ammonium and nitrate	=	Brozyna et al. 2013
Field	Crop residues + energy cover crops	Catch crops	N-NH ₄ ⁺	Applied in spring; 1 out of 3 years	Measurement of soil mineral N in late fall	-	Gunnarsson et al. 2011
Field	Crop residues + energy cover crops	Crop residues + catch crops	Total N	Applied in spring; 3 out of 6 years	Measurement of soil mineral N in late fall or post application	-	Möller and Stinner 2009

Field	Cattle slurry + crop residues + energy cover crops	Cattle slurry + crop residues + catch crops	?	Applied in spring to different rotations; repeated for 2 years	Measurement of soil mineral N in late fall or post application	-	Möller and Stinner 2009
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Table 24. Effects of using digestate as fertilizer on nitrous oxide emissions across different studies. +: significant increase in N₂O emissions associated with digestate use; -: significant decrease in emissions associated with digestate use; =: no difference in emissions associated with digestate use.

Context	Digestate substrate	Control	Application rates (using equivalents)	Crop	Assessment method	Effect on N ₂ O emissions	Emissions factor	Reference
Field	Cattle slurry	Cattle slurry	Total N	Corn	Static chamber	+126%	1.38% of total N	Möller and Stinner 2009
Field	Cattle slurry + food waste	Cattle slurry	Total N	Grassland	Static chamber	+ ?	0.1–0.3% of total N	Wulf et al. 2002
Field	Cattle slurry + food waste	Cattle slurry	Total N	Bare soil	Static chamber	- ?	0.08–0.17% of total N	Wulf et al. 2002
Field	Pig slurry	Pig slurry	Total N	Grassland	Non-flowthrough, non-steady-state chamber	-17 to -71%	0.14–0.4% of total N	Chantigny et al. 2007
Field	Animal slurry + food waste	Animal slurry	N-NH ₄ ⁺	Barley	Static chamber	-20 to -40%	0.14–0.64% of total N	Petersen 1999
Field	Pig slurry	Pig slurry	Total N	Potato	Static chamber	-48%	0.57% of total N	Vallejo et al. 2006
Field	Animal slurry	Animal slurry	N-NH ₄ ⁺	Barley	Static chamber	-38% to -72%	0.1% of total N	Baral et al. 2017
Field	Food waste	Animal slurry	?	Grassland/ wheat	Static chamber	=	0.45% of total N	Nicholson et al. 2017
Field	Cattle slurry	Cattle slurry	Fresh matter	Grassland	Static chamber	=		Amon et al. 2006
Field	Pig slurry + corn	Animal slurry	Total N	Corn/wheat/ grassland	Static chamber	=		Herrmann et al. 2017

Field	Pig slurry	Pig slurry	Total N	Corn	Non-flowthrough, non-steady-state chamber	=		Chantigny et al. 2010
Field	Pig slurry	Pig slurry	Total N	Wheat/barley	Static chamber	=	0.53% of total N (on surface)/0.96% of total N (injected)	Thomsen et al. 2010
Field	Cattle slurry	Cattle slurry	Total N	Grassland	Static chamber	=	0.7% of total N	Clemens et al. 2006
Field	Corn	Animal slurry	Total N	Corn/wheat/grassland	Static chamber	=	0.3–1.9% of total N (minimum for grassland; maximum for corn)	Senbayram et al. 2014
Field	Cattle slurry	Cattle slurry	Fresh matter	Wheat/barley	Static chamber	=	0.1% of total N (in spring)/0.44% of total N (in fall)	Rodhe et al. 2015
Laboratory	Food waste	Cattle slurry	N-NH ₄ ⁺	Bare soil	Incubator	-63%	5.77% of total N	Köster et al. 2015
Laboratory	Pig slurry	Pig slurry	Total N	Bare soil	Microcosm	=		Severin et al. 2015
Field	Crop residues + energy cover crops + grass	Catch crops	Total N	Arable rotation	Static chamber	-38%	1% of total N	Möller and Stinner 2009
Field	?	Catch crops	?	Wheat/corn	Modeling	-25%		Szerencsits 2014

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Table 25. Summary of climate characteristics of the experimental site over the past 31 years. Mean temperature (Mean T°), minimum temperature (Min T°), and maximum temperature (Max T°) in °C, sum of temperatures (ΣT° sorghum and ΣT° rye) in degree days, sum of precipitation (Σ rain) and Penman's potential evapotranspiration (Σ etpp) in mm, sum of global radiation (Σ radiation) in J. m⁻² are calculated each year over the period between sowing and harvest of summer and winter cover crops.

YEAR	Between sowing and harvesting of sorghum							Between sowing and harvesting of rye						
	ΣT° sorghum	Mean T°	Min T°	Max T°	Σ rain	Σ radiation	Σ etpp	ΣT° rye	Mean T°	Min T°	Max T°	Σ rain	Σ radiation	Σ etpp
1991	844	21.5	6.3	36	110	12.8	292	2316	10.6	-6.6	26.7	419	21.5	335
1992	771	20.4	3.7	34.3	138	13	269	2207	10	-6.3	25	601	20	313
1993	894	22.2	5.4	36.8	154	12.3	273	2496	11.4	-2.7	30.7	332	22.6	347
1994	926	22.6	8.7	35.5	62	12.8	290	2334	10.7	-5	28	447	18.7	299
1995	844	21.5	6.3	36	110	12.8	292	2316	10.6	-6.6	26.7	419	21.5	335
1996	771	20.4	3.7	34.3	138	13	269	2207	10	-6.3	25	601	20	313
1997	894	22.2	5.4	36.8	154	12.3	273	2496	11.4	-2.7	30.7	332	22.6	347
1998	852	21.6	6.9	38.6	96	13.7	305	2170	10	-7.5	27.4	432	21.7	284
1999	923	22.6	8.6	35.1	127	14.6	336	2259	10.4	-4.5	30.8	473	22	339
2000	850	21.5	7.7	34.3	65	14.2	311	2379	10.9	-5.1	29.5	462	23.2	332
2001	805	20.9	4.4	37.4	79	14.8	314	2272	10.3	-11.2	28	412	23.7	345
2002	756	20.2	7.4	33.9	82	12.9	274	2422	11.1	-4.1	30.1	406	22.4	333
2003	1043	24.3	9.2	40.4	51	15.6	373	2269	10.4	-6.3	30.6	406	24.5	379
2004	905	22.3	5.3	38	63	14.6	326	2120	9.8	-5.2	29.2	556	21.7	285
2005	854	21.6	9.9	35.1	129	14.5	325	1995	9.2	-9.5	28.9	281	22.6	338
2006	947	22.9	8.5	37.9	116	14.4	354	2315	10.6	-5.9	27.9	292	23.6	344
2007	784	20.6	6.4	36	85	14.5	321	2199	10.1	-7.9	29.1	312	21.1	319
2008	804	20.9	6.2	35.7	107	13.6	299	2206	10.1	-3.4	26.9	509	21.7	329
2009	903	22.3	7	37.8	43	14.4	328	2218	10.2	-5.4	28.2	490	21.3	333
2010	853	21.6	7.4	38.6	64	13.8	309	1974	9	-9.3	29.5	395	22.2	349
2011	864	21.7	9.7	38.3	167	13.4	304	2499	11.4	-6.3	31.1	307	23.5	379
2012	914	22.5	7.6	36.7	91	14.6	343	2223	10	-12.1	27.9	310	21.9	349
2013	901	22.3	8.7	35.6	80	14.1	321	2182	10.1	-5.6	30.5	574	20.4	345
2014	809	21	9.3	33.5	101	13.7	291	2575	11.8	-3.2	30.5	466	22.1	375
2015	852	21.6	8.5	35.6	139	13.2	306	2338	10.7	-5.3	28.9	377	22.3	336
2016	889	22.1	9	35.4	65	14.6	336	2383	10.9	-2.6	28.2	425	21.5	325
2017	839	21.4	8.2	36.1	95	12.8	295	2226	10.2	-7.3	28.7	399	24.1	346
2018	962	23.1	8.5	35.8	117	15	333	2414	11.1	-6.4	32.8	493	21.1	337
2019	928	22.7	6.9	37.4	116	15.2	346	2384	11	-5.1	30.9	512	22.5	359
2020	972	23.3	9	37.4	35	14.9	355	2528	11.5	-2.8	29.8	570	22.1	359
2021	875	21.9	10.4	36.8	93	13.6	302	2285	10.5	-5.5	26.9	385	23.1	357

Table 26. Climate monthly data of the experimental site over the 3 civil years of experiment. Minimum temperature and maximum temperature in °C, sum of global radiation in MJ.m⁻², sum of Penman's potential evapotranspiration in mm, sum of precipitation in mm.

	Minimum temperature (°C)	Maximum temperature (°C)	Mean temperature (°C)	Σradiation (MJ.m ⁻²)	ΣPenman PET (mm)	Σrainfall (mm)
2020						
1	-2.6	18.1	8.1	181.2	13.4	72
2	-0.5	21.6	10.4	273.0	32.7	37.5
3	0.5	22	10.6	428.2	67.8	61.5
4	-0.1	24.5	15.1	524.6	105.8	80
5	9.1	30	18.5	711.8	145.7	69.5
6	10.9	32.3	19.6	655.0	141	104.5
7	11.4	36.2	23.1	750.9	175.3	6
8	10.5	37.4	23.7	625.1	154.9	29.5
9	6.3	34.5	20.1	490.5	106.9	55
10	2.3	22.7	13.5	267.6	44.3	80.5
11	-1.9	22.1	12.3	216.6	25.6	21.5
12	-2.8	16.2	7.6	106.6	25.7	152
2021						
1	-5.5	17.2	5.2	149.4	16.5	86.1
2	2.1	18.6	10.8	234.4	39.2	40
3	-0.5	21.7	10.5	446.7	74.5	24.5
4	-2.1	24.4	12.0	568.5	104.2	21
5	3.2	28	15.1	630.0	120.2	71
6	11.9	33.5	21.3	659.7	148.8	70.5
7	12.1	35.6	21.5	638.7	143.5	35
8	10.4	36.8	21.5	654.9	138.9	10.5
9	11.1	33.2	20.9	436.3	94.7	71
10	0.7	25.4	14.6	365.7	52.7	46.5
11	-2	18.4	8.6	158.9	18.2	60
12	-1.8	22	8.4	162.7	11.2	80.5
2022						
1	-4.8	16.9	4.8	192.5	8.6	79.5
2	-2.4	19.3	9.3	266.5	34.6	17
3	-0.6	21.2	11.2	371.8	73.1	75.5
4	-2.5	23.1	12.5	472.0	88.4	54
5	8.9	32.7	19.5	709.3	153.6	13
6	11	39.2	22.8	675.3	159.6	61
7	10.2	39.8	24.9	836.1	195.5	1.5
8	15.2	38.8	25.7	684.9	167.1	13.5
9	6.2	33.8	20.5	498.7	105.9	68
10	8.9	29	19.5	342.9	72.3	18.5
11	1.3	21.1	12.1	197.9	23.5	103
12	-5.5	19.1	9.1	152.6	8.4	27.5

Table 27. Date and type of measurement made on each side of the field experiment.

	Date	Measurement
"Summer"	2020/07/07	Soil water and mineral nitrogen content (0-90 cm)
	2020/07/10	Start of ammonia volatilization measurement
	2020/07/17	Soil water and mineral nitrogen content (0-30 cm)
	2020/07/30	End of ammonia volatilization measurement
	2020/07/31	Plant density
	2020/08/12	Soil water and mineral nitrogen content (0-30 cm)
	2020/09/15	Plant biomass
	2020/09/21	Soil water and mineral nitrogen content (0-90 cm)
	2020/11/23	Soil water and mineral nitrogen content (0-90 cm)
	2021/02/18	Soil water and mineral nitrogen content (0-90 cm)
	2021/03/03	Start of ammonia volatilization measurement
	2021/03/10	Soil water and mineral nitrogen content (0-30 cm)
	2021/03/23	End of ammonia volatilization measurement
	2021/03/31	Soil water and mineral nitrogen content (0-30 cm)
	2021/06/21	Plant biomass
	2021/07/08	Soil water and mineral nitrogen content (0-90 cm)
	2021/07/16	Start of ammonia volatilization measurement
	2021/07/23	Soil water and mineral nitrogen content (0-30 cm)
	2021/08/05	End of ammonia volatilization measurement
	2021/08/13	Soil water and mineral nitrogen content (0-30 cm)
	2021/08/13	Plant density
	2021/10/11	Plant biomass
	2021/10/28	Soil water and mineral nitrogen content (0-90 cm)
	2022/02/23	Soil water and mineral nitrogen content (0-90 cm)
"Winter"	2020/08/31	Soil water and mineral nitrogen content (0-90 cm)
	2020/10/09	Plant density
	2020/11/19	Plant biomass
	2020/11/23	Soil water and mineral nitrogen content (0-90 cm)
	2021/02/18	Soil water and mineral nitrogen content (0-90 cm)
	2021/03/03	Start of ammonia volatilization measurement
	2021/03/10	Soil water and mineral nitrogen content (0-30 cm)
	2021/03/23	End of ammonia volatilization measurement
	2021/03/31	Soil water and mineral nitrogen content (0-30 cm)
	2021/04/28	Plant biomass
	2021/05/04	Soil water and mineral nitrogen content (0-90 cm)
	2021/06/03	Plant density
	2021/09/06	Plant biomass
	2021/09/27	Soil water and mineral nitrogen content (0-90 cm)
	2022/01/14	Plant density
	2022/02/23	Soil water and mineral nitrogen content (0-90 cm)
	2022/03/25	Plant biomass
	2022/03/31	Start of ammonia volatilization measurement
	2022/04/07	Soil water and mineral nitrogen content (0-30 cm)
	2022/04/20	End of ammonia volatilization measurement
	2022/05/03	Plant biomass
	2022/05/20	Soil water and mineral nitrogen content (0-90 cm)

Table 28. Soil mineral N content (SMNC; kg N.ha⁻¹) and soil water content (SWC; mm.ha⁻¹) before sowing the cover crops in the field experiment.

Treatment	Half Date	"Summer"				"Winter"					
	2020/07/07	2020/07/07		2021/07/08		2020/08/31			2021/09/27		
		MSCC	ECC	MSCC	ECC	CC	MSCC	ECC	CC	MSCC	ECC
SMNC kg N.ha⁻¹											
0-30 cm		42	58	46	48	53	69	87	34	29	31
30-60 cm		45	34	19	35	31	30	45	14	15	13
60-90 cm		53	76	27	92	8	9	20	9	5	2
Σ0-90 cm		140	168	92	175	92	108	152	57	49	46
SWC mm.ha⁻¹											
0-30 cm		94	83	95	93	75	77	76	87	89	85
30-60 cm		101	93	74	71	84	86	86	70	71	67
60-90 cm		93	93	78	71	86	88	85	64	62	60
Σ0-90 cm		288	269	247	235	245	251	247	221	222	212

Table 29. Sunflower emergence density (number of plants.m⁻²) measured at the three leaf pair stage on 2021/06/03 after the different fallow management modalities. Different letters indicate a significant difference ($P < 0.1$) between treatments.

Treatment	BS	CC	MSCC	ECC
Emergence density	29 a	33 a	16 b	29 a

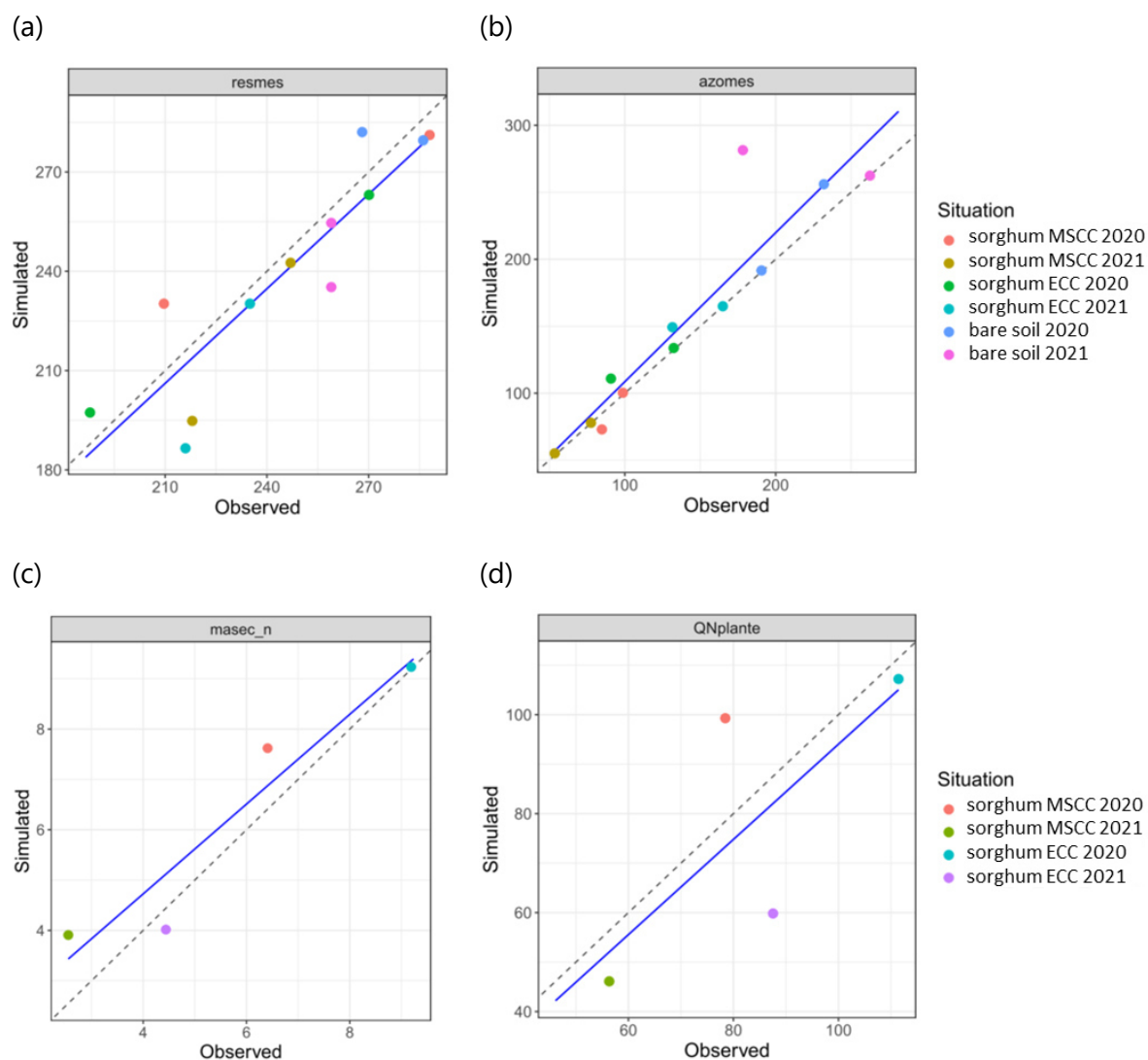


Figure 28. Simulated by STICS versus observed data from the "summer" part of the field experiment during both years of experiment used for calibrating the sorghum plant file. (a) soil water content over 0-90 cm (mm) and (b) soil nitrogen content over 0-90 cm (kg N.ha^{-1}) measured at the beginning and at the end of the summer fallow. (c) Aerial biomass (t MS.ha^{-1}) and (d) nitrogen content of the biomass (kg N.ha^{-1}) of summer cover crops.

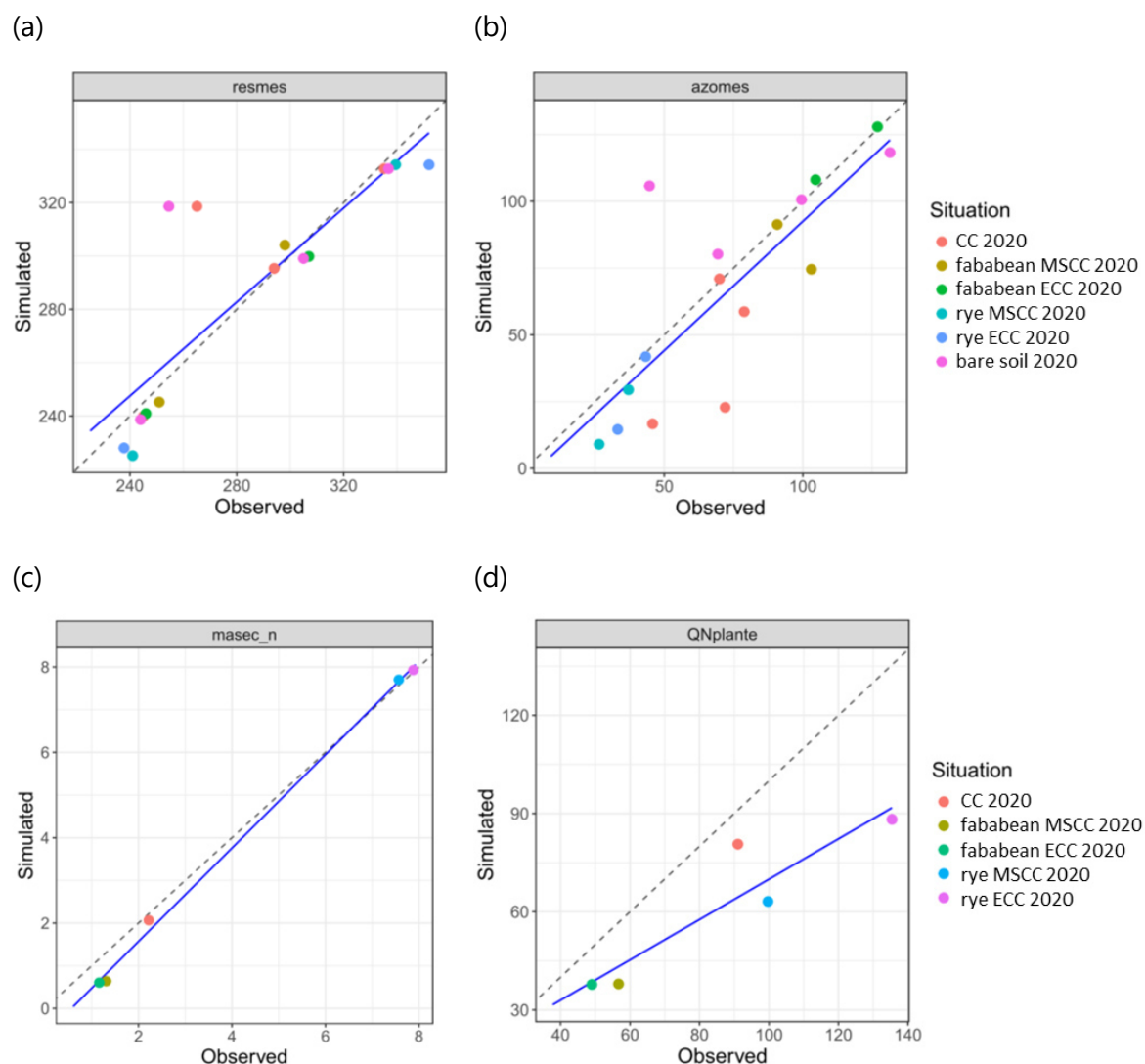
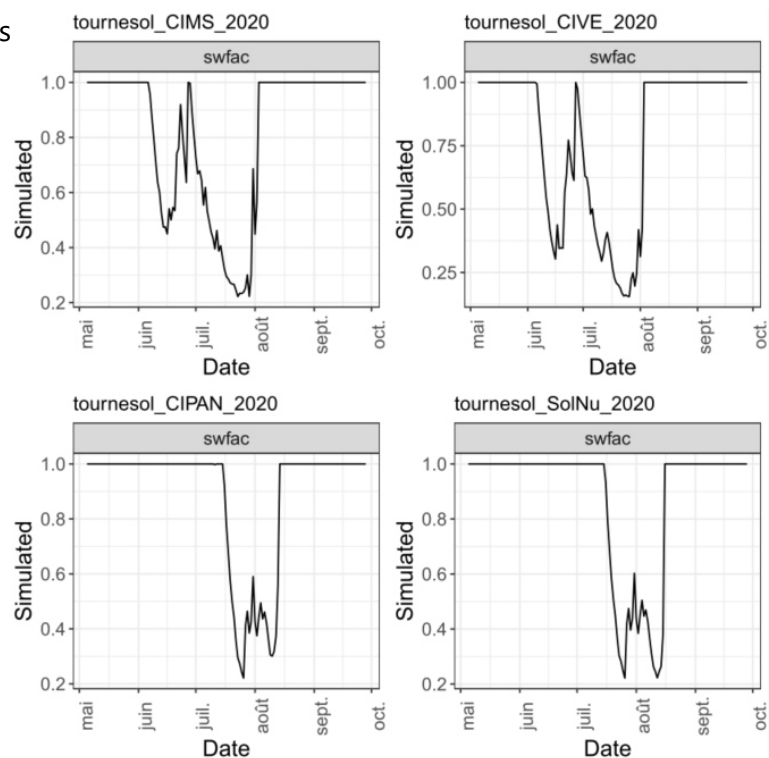


Figure 29. Simulated by STICS versus observed data from the “winter” part of the field experiment during the first year of the experiment used for calibrating the rye plant file. (a) soil water content over 0-90 cm (mm) and (b) soil nitrogen content over 0-90 cm (kg N.ha^{-1}) measured at the beginning and at the end of the winter fallow. (c) Aerial biomass (t MS.ha^{-1}) and (d) nitrogen content of the biomass (kg N.ha^{-1}) of winter cover crops.

(a) Water stress



(b) Nitrogen stress

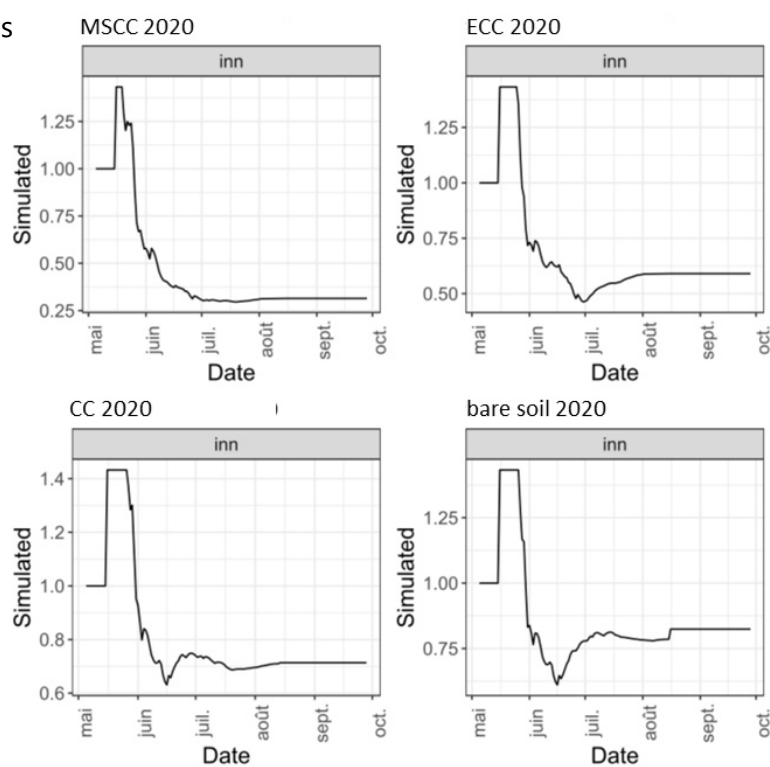


Figure 30. Water (a) and nitrogen (b) nutrition indices estimated by the STICS model for the sunflower following different fallow management modalities in 2020-21.

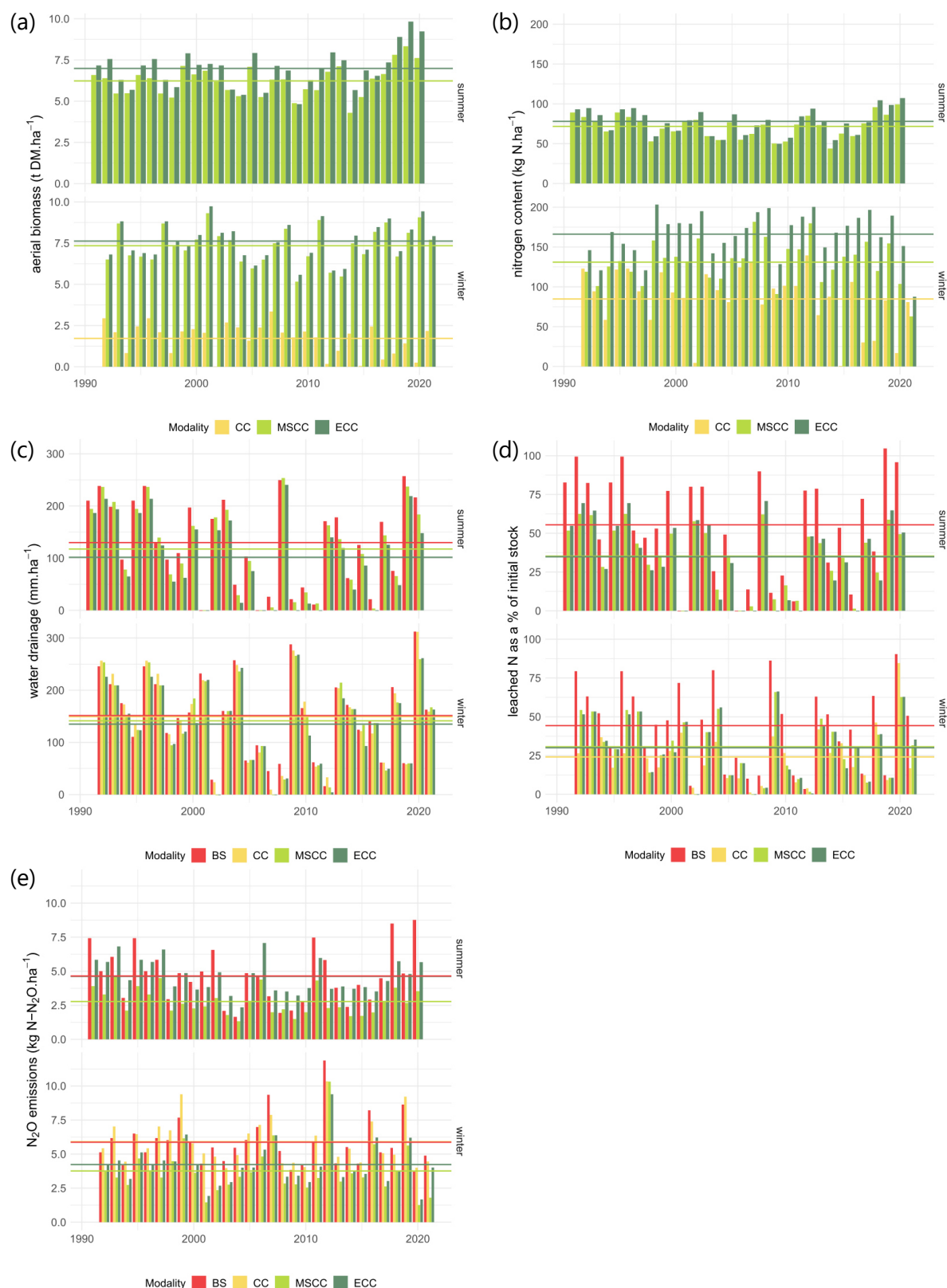


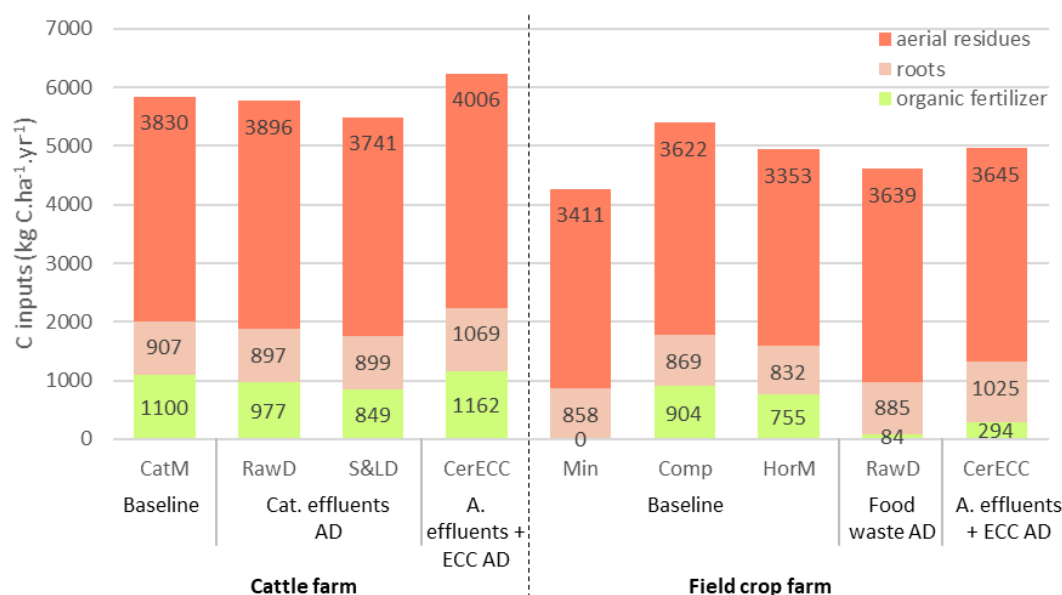
Figure 31. Climatic variability effect on (a) cover crops aerial biomass ($t MS \cdot ha^{-1}$) and (b) its nitrogen content ($kg N \cdot ha^{-1}$), (c) drainage ($mm \cdot ha^{-1} \cdot yr^{-1}$), (d) leaching (% of initial mineral content of the soil), and (e) N_2O emissions ($kg N-N_2O \cdot ha^{-1} \cdot yr^{-1}$) as simulated by STICS with initial data taken from the first year of field experiment. The horizontal line represents the 30-year mean for each modality.

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Table 30. Summary of fertilizers used, cover crops sown, and cover crop fertilization practices on each farm type for each scenario.

	Fertilization	Cover crops	Fertilization of cover crops
Baseline			
Field crop farms	Mineral, dried pig slurry, horse manure, green compost	Catch crop during winter fallow	No
Cattle farm	Mineral, cattle manure and slurry	Catch crop during winter fallow	No
Food waste AD			
Field crop farms	Mineral, digestate	Catch crop during winter fallow	No
Cattle farm	Mineral, cattle manure and slurry	Catch crop during winter fallow	No
On-farm AD			
Field crop farms	Mineral, dried pig slurry, horse manure, green compost	Catch crop during winter fallow	No
Cattle farm	Mineral, digestate	Catch crop during winter fallow	No
Territorial AD			
Field crop farms	Mineral, digestate	Summer and winter ECCs	Mineral fertilizer on the winter cereal ECC
Cattle farm	Mineral, digestate	Summer and winter ECCs	Digestate on the winter cereal ECC

(a) Clay-limestone soil



(b) Deep loam soil

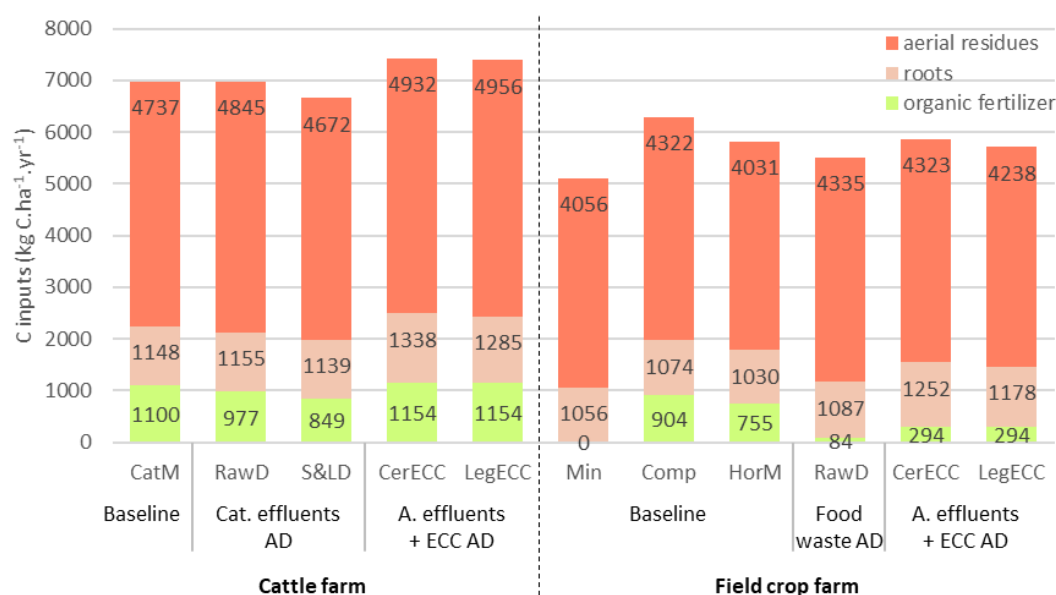


Figure 32. Average annual carbon returns to the soil (in $\text{kg C.ha}^{-1}.\text{yr}^{-1}$) over the 30-year simulation of the different scenarios in the clay-limestone soil (a) and the deep loam soil (b). Carbon inputs are differentiated according to their origin: organic fertilization, roots, aerial residues. The cattle farm uses, in the baseline, cattle manure and slurry (CatM), in the scenario of on-farm anaerobic digestion (Cat. Effluents AD), raw digestate (RawD) or liquid and slurry digestate (S&LD), in the scenario of territorial agricultural anaerobic digestion (A. effluents + ECC AD), raw digestate on a succession where either cereal ECCs (CerECC) or legume ECCs (LegECC) are introduced. The field crop farms use, in the baseline, only mineral fertilization (Min) or green compost and dried pig slurry (Comp) or horse manure (HorM), in the scenario of food waste anaerobic digestion (Food waste AD), raw digestate, in the scenario of territorial agricultural anaerobic digestion (A. effluents + ECC AD), raw digestate on a succession where either cereal ECCs (CerECC) or legume ECCs (LegECC) are introduced.

Table 31. Nitrogen inputs and outputs (t N.yr⁻¹) from the farm–biogas plant systems during the first ten years of simulation according to the different scenarios of implementation of anaerobic digestion in the territory. The scenarios are detailed for each type of farm in the territory: field crop farms using only mineral fertilizers, field crop farms using green compost and dried pig slurry, field crop farms using horse manure, and the cattle farm. Inputs contain N from mineral, organic fertilizers applications, and symbiotic fixation of leguminous crops. Outputs contain N exported from the fields, ammonia volatilization during effluents storage and organic fertilizer applications, nitrate leaching, nitrous oxide emissions through nitrification and denitrification.

Scenario	Farm type	Surface (ha)	Mineral fertilizer	Organic fertilizer	N₂ fixation	Σinputs	Exported N	Storage NH₃ volatilization	Field NH₃ volatilization	NO₃⁻ leaching	N₂O emissions	Σoutputs
Baseline	100% mineral	3360	571.2	0	0	571.2	456.2	0	45.3	21.8	5.9	529.2
	Compost	3400	412.0	282.2	0	694.2	422.6	0	38.3	29.5	6.2	496.6
	Horse manure	640	100.5	16.8	0	117.3	84.9	4	8.2	3.9	1.1	102.1
	Cattle manure	400	53.2	51.6	0	104.8	52.4	10	6.2	3.0	0.7	72.3
Food waste AD	100% mineral	3360	336.0	386.4	0	722.4	458.3	0	54.5	25.7	6.7	545.2
	Compost	3400	340.0	391.0	0	731.0	463.7	0	55.2	26.0	6.8	551.7
	Horse manure	640	64.0	73.6	0	137.6	87.3	0	10.4	4.9	1.3	103.9
	Cattle manure	400	53.2	51.6	0	104.8	52.4	10	6.2	3.0	0.7	72.3
Cattle effluents	100% mineral	3360	571.2	0	0	571.2	456.2	0	45.3	21.8	5.9	529.2

AD w/o phase sep.	Compost	3400	412.0	282.2	0	694.2	422.6	0	38.3	29.5	6.2	496.6
	Horse manure	640	100.5	16.8	0	117.3	84.9	4	8.2	3.9	1.1	102.1
	Cattle manure	400	49.15	52.4	0	101.55	54.6	2	8.1	3.3	0.8	68.8
Cattle effluents AD w/ phase sep.	100% mineral	3360	571.2	0	0	571.2	456.2	0	45.3	21.8	5.9	529.2
	Compost	3400	412.0	282.2	0	694.2	422.6	0	38.3	29.5	6.2	496.6
	Horse manure	640	100.5	16.8	0	117.3	84.9	4	8.2	3.9	1.1	102.1
	Cattle manure	400	44.1	53.2	0	97.3	52.6	2	8.3	3.3	0.8	67
Animal effluents + ECC AD	100% mineral	3360	614.3	142.8	35.4	792.5	571.9	5	74.2	3.5	5.1	659.7
	Compost	3400	621.6	144.5	35.8	801.9	578.7	5	75.1	3.6	5.2	667.6
	Horse manure	640	117.0	27.2	6.7	150.9	108.9	1	14.1	0.7	1.0	125.7
	Cattle manure	400	38.2	68.8	9.0	116.0	64.0	2	11.0	0.3	0.6	77.9

Table 32. Evolution of the nitrogen inputs and outputs ($t\ N.yr^{-1}$) from the farm–biogas plant systems during the last ten years of simulation according to the different scenarios of implementation of anaerobic digestion in the territory. The scenarios are detailed for each type of farm in the territory: field crop farms using only mineral fertilizers, field crop farms using green compost and dried pig slurry, field crop farms using horse manure, and the cattle farm. Inputs contain N from mineral, organic fertilizers applications, and symbiotic fixation of leguminous crops. Outputs contain N exported from the fields, ammonia volatilization during effluents storage and organic fertilizer applications, nitrate leaching, nitrous oxide emissions through nitrification and denitrification.

Scenario	Farm type	Surface (ha)	Mineral fertilizer	Organic fertilizer	N ₂ fixation	Σinputs	Exported N	Storage NH ₃ volatilization	Field NH ₃ volatilization	NO ₃ ⁻ leaching	N ₂ O emissions	Σoutputs
Baseline	100% mineral	3360	571.2	0	0	571.2	526.9	0	48.3	34.8	7.2	617.2
	Compost	3400	281.8	282.2	0	564.0	492.4	0	27.8	49.7	8.2	578.1
	Horse manure	640	85.4	16.8	0	102.2	93.5	4	7.6	5.7	1.2	112
	Cattle manure	400	37.4	51.6	0	89.0	61.9	10	5.4	7.0	1.0	85.3
Food waste AD	100% mineral	3360	240.2	386.4	0	626.6	533.3	0	52.3	50.8	8.5	644.9
	Compost	3400	243.1	391.0	0	634.1	539.7	0	53.0	51.4	8.6	652.7
	Horse manure	640	45.8	73.6	0	119.4	101.6	0	10.0	9.7	1.6	122.9
	Cattle manure	400	37.4	51.6	0	89.0	61.9	10	5.4	7.0	1.0	85.3
Cattle effluents	100% mineral	3360	571.2	0	0	571.2	526.9	0	48.3	50.8	8.5	634.5

AD w/o phase sep.	Compost	3400	281.8	282.2	0	564.0	492.4	0	27.8	51.4	8.6	580.2
	Horse manure	640	85.4	16.8	0	102.2	93.5	4	7.6	9.7	1.6	116.4
	Cattle manure	400	28.5	52.4	0	80.9	63.3	2	7.1	7.1	1.1	80.6
Cattle effluents AD w/ phase sep.	100% mineral	3360	571.2	0	0	571.2	526.9	0	48.3	50.8	8.5	634.5
	Compost	3400	281.8	282.2	0	564.0	492.4	0	27.8	51.4	8.6	580.2
	Horse manure	640	85.4	16.8	0	102.2	93.5	4	7.6	9.7	1.6	116.4
	Cattle manure	400	32.5	53.2	0	85.7	65.0	2	8.1	7.2	1.1	83.4
Animal effluents + ECC AD	100% mineral	3360	540.5	142.8	35.4	718.7	631.9	5	71.5	9.8	6.1	724.3
	Compost	3400	546.9	144.5	35.8	727.2	639.4	5	72.4	9.9	6.2	732.9
	Horse manure	640	103.0	27.2	6.7	136.9	120.4	1	13.6	1.9	1.2	138.1
	Cattle manure	400	30.5	68.8	9.0	108.3	81.3	2	11.0	1.6	0.9	96.8

Annexes – Chapter 5: Supplementary Material

Table 33. Correspondence between the names of the regions of Metropolitan France and their numerical code.

Region code	Region name
11	Ile-de-France
21	Champagne-Ardenne
22	Picardie
23	Haute-Normandie
24	Centre
25	Basse-Normandie
26	Bourgogne
31	Nord-Pas-de-Calais
41	Lorraine
42	Alsace
43	Franche-Comté
52	Pays de la Loire
53	Brittany
54	Poitou-Charentes
72	Aquitaine
73	Midi-Pyrénées
74	Limousin
82	Rhône-Alpes
83	Auvergne
91	Languedoc-Roussillon
93	Provence-Alpes-Côte d'Azur



Figure 33. Map of the former regions of France

Table 34. Maximum harvest dates of winter crops in current cropping systems. This is the limiting factor for the sowing of summer cover crops.

	Regions before the 07/19	Regions after the 07/19
Winter wheat	72, 73	11, 21, 22, 23, 24, 25, 26, 31, 41, 42, 43, 52, 53, 54, 74, 82, 83, 91, 93
Rapeseed	11, 22, 24, 25, 31, 42, 43, 52, 53, 54, 72, 73, 74, 82, 83, 91, 93	41, 26, 23, 21
Winter pea	All	
Spring pea	52, 53, 54, 72, 73, 82, 93	11, 21, 22, 23, 24, 25, 26, 31, 41

Table 35. Sowing date of winter crops in the current cropping systems. This is the limiting factor for the harvest of summer energy cover crops.

	In the earliest region	In the latest region
Winter wheat	September 29 th	November 5 th
Rapeseed	August 22 nd	September 15 th
Winter pea	October 26 th	December 9 th

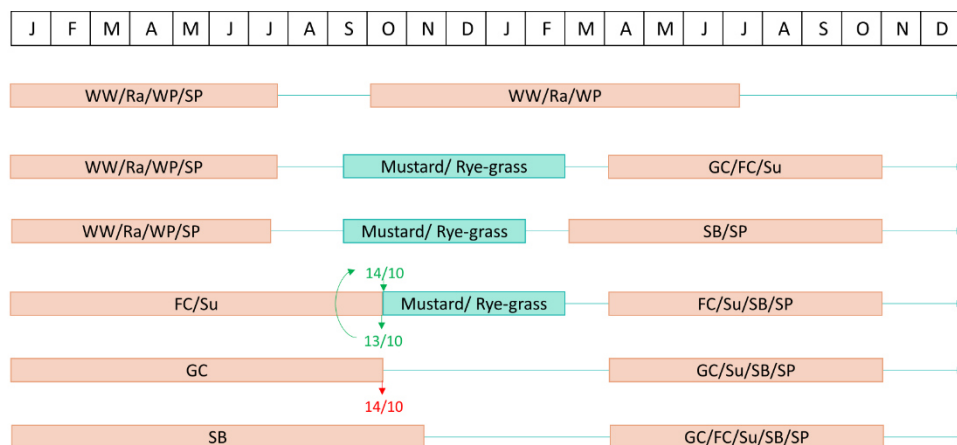
Table 36. Maximum harvest dates of spring crops in current cropping systems. This is the limiting factor for the sowing of winter energy cover crops.

	Regions before the 10/14	Regions after the 10/14
Grain corn		All
Forage corn	All	
Sunflower	All	
Sugar beet	21	11, 22, 23, 24, 25, 31, 42, 83

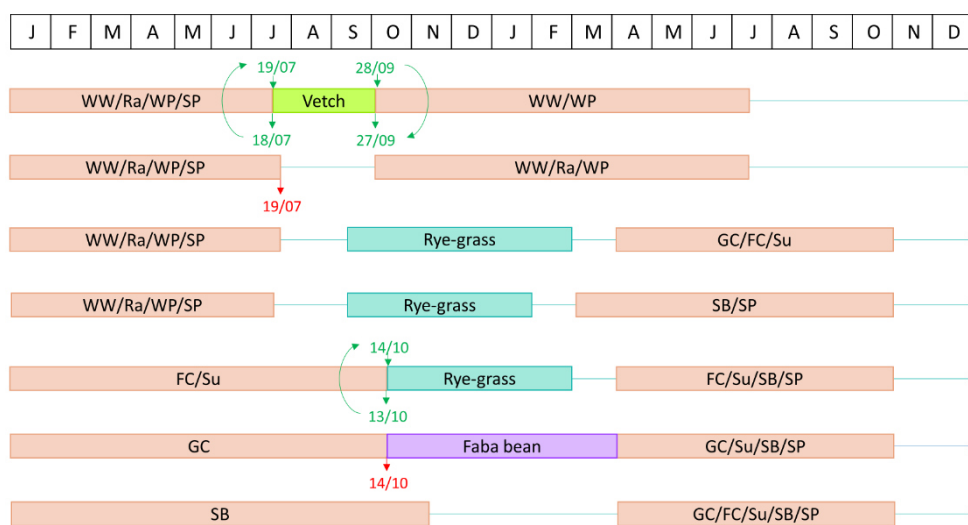
Table 37. Sowing date of spring crops in the current cropping systems. This is the limiting factor for the harvest of winter energy cover crops.

	In the earliest region	In the latest region
Grain corn	April 12 th	April 30 th
Forage corn	April 19 th	May 12 th
Sunflower	April 9 th	April 21 st
Spring pea	December 10 th	March 15 th
Sugar beet	March 12 th	March 30 th

(a) Baseline



(b) MSCC extension



(c) ECC insertion and extension

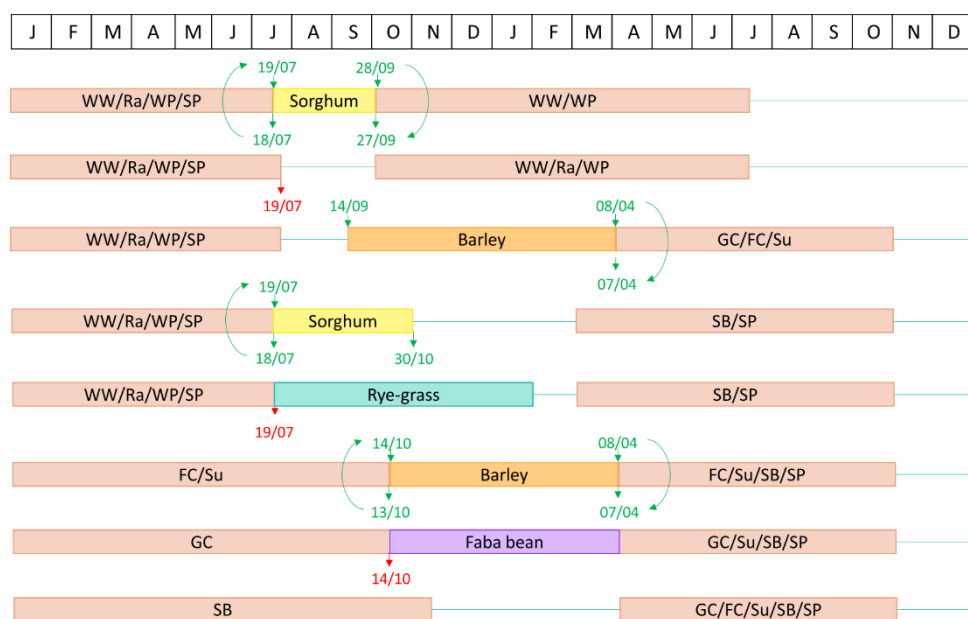


Figure 34. Insertion of cover crops in cropping sequences depending on the scenarios. FC: forage corn, GC: grain corn, Ra: rapeseed, SB: sugar beet, SP: spring pea, Su: sunflower, WP: winter pea, WW: winter wheat.

Table 38. Definition of the variables used in the correlation matrices.

Variable name	Variable definition
aerial_Fe_t_ha_mean.y	Aerial biomass produced by faba bean in the cover crops scenarios
aerial_Or_t_ha_mean.y	Aerial biomass produced by winter barley in the cover crops scenarios
aerial_Rg_t_ha_mean.y	Aerial biomass produced by rye-grass in the cover crops scenarios
aerial_So_t_ha_mean.y	Aerial biomass produced by sorghum in the cover crops scenarios
aerial_Ve_t_ha_mean.y	Aerial biomass produced by vetch in the cover crops scenarios
arg.y	Clay content
cprecip_mean_an.y	Sum of annual precipitation and irrigation
ddelta_SOC_annuel_absolu	SOC annual storage difference between the cover crops scenarios and the baseline
delta_bilan_GES	GHG balance difference between the cover crops scenarios and the baseline
delta_biomass_CC	Cover crops cumulative biomass difference between the cover crops scenarios and the baseline
delta_export_tMS	Biomass exportation (of main crops and cover crops) difference between the cover crops scenarios and the baseline
delta_drain	Drainage difference between the cover crops scenarios and the baseline
delta_ferti_tot	Mineral fertilization difference between the cover crops scenarios and the baseline
delta_freq_CC	Cover crop frequency difference between the cover crops scenarios and the baseline
delta_freq_Co	Rapeseed frequency difference between the cover crops scenarios and the baseline
delta_freq_MG	Gain corn difference between the cover crops scenarios and the baseline
delta_irrig_tot	Irrigation difference between the cover crops scenarios and the baseline
delta_N_lix	N leaching difference between the cover crops scenarios and the baseline
delta_N_N2O	N ₂ O emissions difference between the cover crops scenarios and the baseline
delta_N_volat	NH ₃ volatilization difference between the cover crops scenarios and the baseline
delta_QCintot	C inputs to the soil difference between the cover crops scenarios and the baseline
delta_Qmin	SON mineralization difference between the cover crops scenarios and the baseline

delta_QNplantCC	N uptake by cover crops difference between the cover crops scenarios and the baseline
delta_tp_summer_CC	Summer cover crops transpiration difference between the cover crops scenarios and the baseline
delta_tp_winter_CC	Winter cover crops transpiration difference between the cover crops scenarios and the baseline
energie_primaire_produite_an_injection.y	Energy produced from AD of ECCs
export_cive_an.y	Exportation frequency of ECCs
flux_Ntot_dig_poststockage_mean_an.y	Annual N input through digestate
pH.y	Soil pH
profondeur.y	Soil depth
taux_export_Or.y	Exportation frequency of winter barley
taux_export_So.y	Exportation frequency of sorghum
taux_Fe_an.y	Frequency of faba bean in the cover crops scenarios
taux_Mo_an.x	Frequency of mustard in the baseline
taux_Or_an.y	Frequency of winter barley in the cover crops scenarios
taux_Rg_an.y	Frequency of rye-grass in the cover crops scenarios
taux_So_an.y	Frequency of sorghum in the cover crops scenarios
taux_Ve_an.y	Frequency of vetch in the cover crops scenarios
stock_C_initial.x	Initial SOC stock

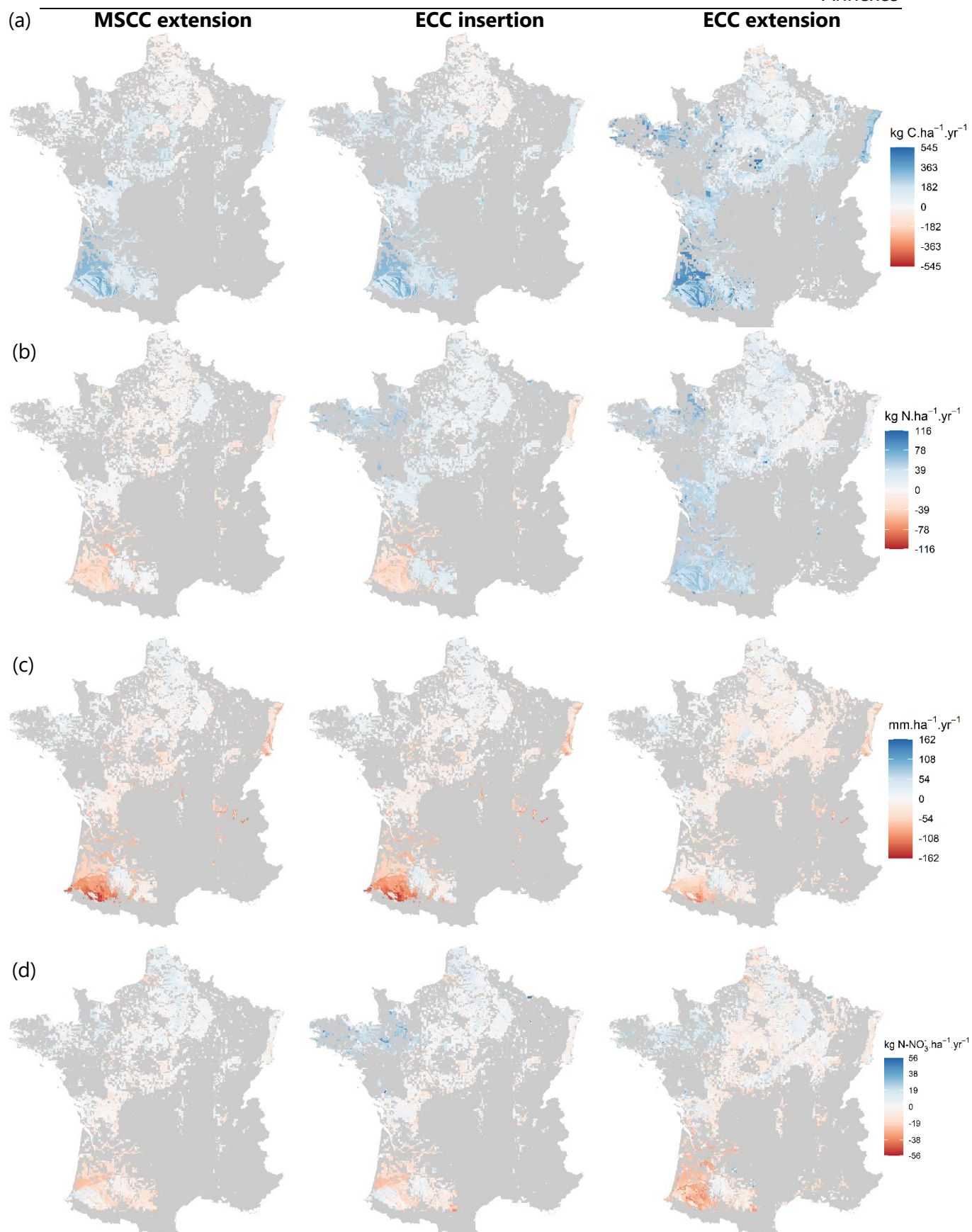


Figure 35. Maps of the environmental impacts of each scenario compared to the baseline a) on SOC storage, b) on mineral N fertilizer consumption, c) on water drainage, d) on nitrate leaching.

Annexes – Résumé en français

Insertion de cultures intermédiaires à vocation énergétique dans les systèmes de cultures en France : évaluation multi-échelles du potentiel de production et des impacts eau – azote – carbone.

Introduction

La méthanisation agricole, filière de production de biogaz, est aujourd'hui en pleine expansion en Europe grâce à l'appui des politiques publiques. La méthanisation est la dégradation de la matière organique par des micro-organismes dans des conditions contrôlées et en l'absence d'oxygène. Cette dégradation produit du gaz, un mélange de CO₂ et de CH₄, qui peut être purifié avant d'être injecté dans le réseau de gaz. La matière organique restante est appelée digestat et est épandue sur les sols agricoles comme engrais organique. La méthanisation a connu un regain d'intérêt en France au début des années 2000. Les éleveurs se sont d'abord intéressés à ce procédé pour traiter leurs effluents d'élevage en co-digestion avec des matières organiques issues des industries agro-alimentaires ou des municipalités (ex : déchets verts) (Béline et al. 2012). Cependant, les effluents d'élevage sont peu méthanogènes. Les cultures énergétiques produisent plus de biogaz, c'est pourquoi elles ont commencé à être utilisées comme co-substrat (Béline et al. 2012).

Les cultures intermédiaires, implantées entre deux cultures principales, pourraient fournir une quantité considérable de biomasse pour la méthanisation, en particulier dans les exploitations sans élevage. Pour parvenir à une sortie du gaz fossile d'ici 2050, la méthanisation des cultures intermédiaires à vocation énergétique (CIVEs) apparaît comme centrale dans les études prospectives. Selon ses différents scénarios (ADEME 2018a, 2021), l'ADEME prévoit que la méthanisation fournisse d'un tiers à deux tiers de l'approvisionnement en gaz en France (de 95 à 140 TWh_{PCS}) et qu'un tiers à la moitié de ce gaz produit par méthanisation serait fourni par les CIVEs (51-61 TWh_{PCS}).

Jusqu'à présent, les cultures intermédiaires n'étaient ni fertilisées ni récoltées, rendues parfois obligatoires car piégeant le nitrate pour limiter la contamination des nappes (cultures intermédiaires pièges à nitrate, CIPAN). Elles peuvent aussi rendre d'autres services écosystémiques comme la protection du sol face à l'érosion ou le stockage de carbone dans les sols (cultures intermédiaires multi-services, CIMS). Avec le développement de la méthanisation,

la conduite des cultures intermédiaires change pour produire plus de biomasse. Elles sont fertilisées, potentiellement avec le digestat de méthanisation, et récoltées avant maturité. C'est pourquoi on parle alors de cultures intermédiaires à vocation énergétique (CIVEs). La transition énergétique vers l'utilisation d'énergie moins carbonée ne doit pas faire oublier les autres crises environnementales que nous devons régler comme la 6ème extinction de masse des espèces ou encore les diverses pollutions de notre environnement mettant en jeu notre santé. D'autres critères que les émissions de CO₂ évitées doivent être pris en compte, tels que les émissions d'autres gaz à effet de serre (N₂O et CH₄), les impacts sur la biodiversité, la pollution de l'air et de l'eau, la production alimentaire, etc. Face à cette problématique complexe, notre ambition est d'évaluer si et dans quelle mesure les CIVEs peuvent répondre aux enjeux environnementaux et énergétiques d'aujourd'hui sans compromettre la production alimentaire.

Les objectifs de la thèse sont i) de consolider les connaissances scientifiques sur les impacts des systèmes de culture intégrant des CIVEs et la fertilisation par digestat sur les cycles de l'eau, de l'azote et du carbone, ii) d'analyser les inconvénients d'une production accrue de cultures intermédiaires par rapport à la concurrence potentielle avec la production alimentaire ou à l'utilisation accrue d'intrants (i.e. engrais azotés et irrigation), iii) de proposer une nouvelle estimation de la production potentielle de biomasse des CIVEs et de la production d'énergie associée en utilisant une approche à haute résolution et en tenant compte de la diversité du climat, du sol et des systèmes de culture en France. Ces objectifs sont décomposés en plusieurs questions de recherche plus spécifiques qui seront étudiées à trois échelles différentes :

- Les systèmes de culture incluant des CIVEs et des digestats atténuent-ils le changement climatique ? C'est-à-dire stockent-ils du carbone dans les sols, réduisent-ils les émissions directes et indirectes de N₂O et les émissions induites, substituent-ils suffisamment de gaz fossile pour compenser leurs émissions ?
- L'introduction de CIVEs fertilisées dans un système de culture augmente-t-elle les apports et les pertes d'azote (lixiviation des nitrates, émissions de N₂O, volatilisation de NH₃) ?
- Quelle est la consommation d'eau des CIVEs ? Quel est son impact sur le cycle de l'eau ?

- Dans quelle mesure les CIVEs augmentent-elles la production de biomasse agricole ? Est-ce au détriment de la biomasse alimentaire ? Pourraient-ils produire de la biomasse partout en France ?
- La prise en compte des flux à l'échelle de la parcelle et de l'unité de méthanisation modifie-t-elle les conclusions sur les questions précédentes ?

Etat de l'art

Une revue de la littérature sur les connaissances et les lacunes sur les CIVEs et leurs impacts possibles est nécessaire dans un premier temps pour affiner les questions de recherche et formuler des hypothèses de travail pour la suite de l'étude. L'état de l'art sur les avantages et les impacts environnementaux de l'introduction de CIVEs dans les systèmes agricoles est présenté dans le Chapitre 1. Après avoir passé en revue la littérature grise et scientifique concernant les impacts des CIMS, des CIVEs et de la fertilisation par le digestat, nous avons émis les hypothèses suivantes :

- La CIVE avec retour à la parcelle du digestat stockerait autant de carbone dans le sol que la CIMS laissée sur place. Avec la production supplémentaire de gaz qui couvrirait une bonne partie des besoins en gaz d'une France décarbonée, les CIVEs amélioreraient le bilan GES de l'agriculture française.
- L'insertion de CIVEs fertilisées devrait augmenter la consommation globale d'azote et la volatilisation de NH_3 associée. L'utilisation de digestat devrait générer une volatilisation supplémentaire de NH_3 .
- L'exportation de la biomasse des CIVEs et sa restitution sous une forme plus stable réduirait les émissions de N_2O par rapport aux CIMS laissées sur place.
- Même si elles sont fertilisées, les CIVEs réduiraient la lixiviation des nitrates par rapport au sol nu et aux CIMS en raison de la production plus importante de biomasse et de l'exportation de résidus.
- Les CIVEs devraient réduire le drainage de la même manière que les CIMS par rapport au sol nu, mais l'irrigation des CIVEs pourrait accroître la pression sur les ressources en eau.

- Les CIVEs sont plus susceptibles de réduire le rendement de la culture suivante que les CIMS en raison de leur récolte tardive, mais surtout en raison des changements de pratiques des agriculteurs qui sèmeraient leurs cultures plus tard avec des variétés plus précoces.

Démarche

Pour tester ces hypothèses, nous avons mis en œuvre une approche combinant expérimentation et modélisation. L'objectif de l'expérimentation était de quantifier en laboratoire et sur le terrain les services et les dys-services rapportés dans la synthèse bibliographique afin de la consolider. La modélisation a permis d'accéder à des flux non mesurés dans les expérimentations et d'élargir considérablement l'échelle spatiale et temporelle de l'étude.

Après le chapitre 1, la thèse est divisée en quatre chapitres correspondant aux quatre différentes échelles spatiales auxquelles les impacts ont été étudiés : le chapitre 2 commence à l'échelle de la parcelle, le chapitre 3 descend à l'échelle du laboratoire, le chapitre 4 monte à l'échelle d'une petite région et le chapitre 5 à l'échelle de la France entière. Les méthodes utilisées et les objectifs/questions scientifiques sont donc complémentaires dans chaque chapitre.

Le Chapitre 2 présente une expérimentation au champ de deux ans dans laquelle les CIVEs sont comparés à d'autres modes de gestion de l'interculture : sol nu et cultures intermédiaires non exportées. L'expérimentation est couplée au modèle de culture STICS (Brisson et al. 2009). Ce dispositif expérimental répond à différents objectifs :

- Déterminer l'impact des CIVEs et de la restitution du digestat sur les cycles de l'eau et de l'azote à l'échelle du champ à court terme.
- Analyser l'impact des CIVEs sur le rendement et les niveaux de stress de la culture suivante.
- Comparer ces effets à ceux des cultures intermédiaires non exportées terminées à l'automne ou au printemps et à ceux d'un sol nu pendant la période d'interculture.
- Calibrer le modèle de culture STICS sur certaines espèces de CIVEs.

Le Chapitre 3 est complémentaire du chapitre 2 car il est basé sur la même expérimentation au champ. Une partie de la biomasse récoltée au champ est digérée dans un réacteur pilote et la minéralisation du carbone et de l'azote du digestat produit et de la biomasse végétale initiale est suivie dans des incubations en laboratoire. L'objectif principal de ce chapitre est donc complémentaire à ceux du chapitre précédent :

- Déterminer l'impact de la méthanisation des CIVEs sur le cycle du carbone.

Dans le Chapitre 4, nous utilisons le modèle STICS pour représenter les systèmes de culture d'une petite région agricole pendant 30 ans. Couplé à un modèle de méthanisation SYS-Metha (Bareha et al. 2021b), il nous permet de simuler différents scénarios d'implantation d'unités de méthanisation sur le territoire, dont un scénario de développement des CIVEs. A cette échelle d'étude, nous prenons en compte les contraintes sur les systèmes de culture et les pratiques des agriculteurs de la zone, les ressources organiques disponibles, le volume total de digestat produit et les surfaces disponibles pour l'épandage de ce digestat. Les objectifs de ce chapitre sont très différents de ceux de l'échelle parcellaire :

- Analyser les impacts des CIVEs sur les échanges d'azote et de carbone avec une unité de méthanisation.
- Comparer les conséquences du développement de différents scénarios de développement de la méthanisation sur les entrées et sorties d'azote de la région, et sur le stockage de C dans la région.

Enfin, dans le Chapitre 5, nous mettons en place une chaîne de simulation initialement créée par Launay et al. (2021a) réutilisant des résultats ou des points méthodologiques des chapitres précédents. Nous l'utilisons pour simuler différents scénarios de développement de cultures intermédiaires pour la méthanisation ou non à l'échelle de la France entière sur une grille à la résolution de quelques km² pendant 30 ans. Cette échelle permet de remplir les objectifs suivants :

- Quantifier le potentiel de production de biomasse des CIVEs en France et les impacts environnementaux associés.
- Analyser les impacts à long terme de l'insertion des CIVEs dans une large gamme de conditions pédoclimatiques et de successions de cultures sans élevage.

Nous concluons dans le Chapitre 6 par une discussion générale dans laquelle nous synthétisons et mettons en cohérence les points saillants des résultats.

Résultats

Production d'énergie

Selon nos travaux, la quantité d'énergie que les CIVEs pourraient produire est considérable puisque, en passant d'hypothèses très conservatrices à des hypothèses de forte mobilisation des CIVEs, elle serait comprise entre 17 et 115 TWh_{PCI}. Cela représente de 4 à 27 % de la consommation française de gaz en 2021 (419 TWh_{PCI} ; GRDF et al. 2022). Les études précédentes se situent plutôt dans la fourchette basse de nos estimations : 8 TWh_{PCI} (ADEME et al. 2013), 46 TWh_{PCI} (ADEME 2018b), 30-55 TWh_{PCI} (ADEME 2021), 36 TWh_{PCI} (WWF France 2022), 27-43 TWh_{PCI} (France Stratégie 2021), 6-33 TWh_{PCI} (Agrosolutions et EIFER 2022). Dans ces études, les surfaces des CIVEs ont toutes été calculées à partir du Recensement Agricole 2010 du Service de la Statistique et de la Prospective du Ministère de l'Agriculture et de l'Alimentation. À l'exception de l'étude de l'ADEME et al. (2013), qui a tenté de mieux prendre en compte la période d'interculture, les autres études ont considéré toutes les surfaces de cultures de printemps comme des surfaces potentielles d'implantation de CIVEs d'hiver et toutes les surfaces de cultures d'hiver comme des surfaces potentielles de CIVEs d'été. Ensuite, les hypothèses relatives à l'adoption des CIVEs et aux modifications de l'assolement ont modifié la superficie des CIVEs prise en compte dans chaque étude. En calculant par zones de quelques km² la surface de CIVEs dans les principales rotations et en appliquant des règles de décision sur la date de récolte de la culture précédente et de semis de la suivante, nous avons obtenu des surfaces proches de la première étude de l'ADEME, c'est-à-dire inférieures à l'ensemble des autres études. Les rendements considérés et les taux de mobilisation de la biomasse étaient très variables d'une étude à l'autre et choisis grossièrement par expertise. Notre travail a permis d'affiner les rendements réalisables et les fréquences d'échec. Notre analyse de sensibilité aux hypothèses de quantité de biomasse laissée sur place et de seuil de rentabilité a permis de préciser ce que les études mettaient derrière leur "taux de mobilisation" qui a finalement une forte influence sur la ressource mobilisable.

Stockage de carbone

En partant d'une même quantité de biomasse végétale, la méthanisation des CIVEs n'a pas permis de maintenir le même stock de carbone organique dans le sol que si leur biomasse avait été laissée sur place dans le champ (chapitre 3). La méthanisation des CIVEs ne pouvait augmenter les stocks de carbone organique que si les CIVEs fournissaient significativement plus de carbone que la situation initiale. Ces résultats sont différents de ceux de Thomsen et al. (2013) et Béghin-Tanneau et al. (2019) qui ont pourtant utilisé une méthode proche de la nôtre. Avec une différence de stockage de carbone de 14% en faveur de la biomasse brute végétale, les premiers ont conclu à un niveau de stockage similaire. Avec une différence de stockage de 27% cette fois au profit du digestat issu des cultures, les seconds concluent à un avantage de la méthanisation. L'étude de Levavasseur et al. (2023a), qui n'a pas utilisé d'incubations mais des simulations sur des modèles, aboutit à la même conclusion que la nôtre. A l'échelle de la France, le stockage de carbone organique observé en passant des systèmes de culture actuels ou du scénario d'extension des CIMS au scénario d'introduction des CIVEs pourrait donc s'expliquer par l'augmentation de la production de biomasse en période d'interculture (chapitre 5). La biomasse produite par les cultures intermédiaires était beaucoup plus importante dans les scénarios d'introduction et d'extension des CIVEs que dans le scénario de référence et le scénario d'extension des CIMS. Cette production supplémentaire de biomasse a plus que compensé la perte de carbone dans le biogaz. En revanche, si l'on abandonne les amendements organiques actuellement utilisés, la méthanisation des CIVEs et l'utilisation des digestats seuls ne suffiraient pas à maintenir les stocks de carbone organique des sols (chapitre 4). Le territoire étudié, la plaine de Versailles, comprend des exploitations de grandes cultures utilisant des engrais minéraux et des exploitations de grandes cultures utilisant des effluents d'élevage ou des composts de déchets verts. Les résultats pour les exploitations utilisant des engrais minéraux sont conformes aux résultats pour les mêmes exploitations retenues dans les simulations à l'échelle de la France (chapitre 5). En revanche, sur les exploitations de grandes cultures utilisant des effluents d'élevage ou du compost, l'abandon de ces amendements au profit du digestat a déstocké du carbone malgré l'insertion de CIVEs. L'intérêt de cette étude territoriale a donc été de constater qu'avec les changements de pratiques et la compétition pour les ressources organiques induits par la méthanisation, certains agriculteurs pouvaient être pénalisés en termes de stocks de carbone organique dans leurs champs. Etant donné que certaines de ces ressources organiques sont difficilement utilisables comme substrat pour la méthanisation (déchets verts ligneux par exemple), la

combinaison de l'utilisation du digestat et du compost pourrait être recommandée pour maintenir un stockage élevé de carbone dans les sols.

Consommation d'engrais minéraux

Comme prévu, les CIVEs ont augmenté l'utilisation d'engrais minéraux en moyenne au cours de la rotation (chapitres 4 et 5). Cela s'explique d'une part par le fait que les CIVEs ont été fertilisées mais n'ont pas bien valorisé tout l'azote, car une partie de l'azote fourni a été perdue (directement dans le champ ou pendant le stockage du digestat par le biais d'émissions gazeuses) et n'a pas été transformée en digestat. D'autre part, l'augmentation des apports azotés aux cultures principales a été nécessaire pour compenser la concurrence préemptive des CIVEs pour l'azote. La consommation d'engrais a tout de même diminué au fil du temps grâce à la minéralisation de la matière organique du sol qui augmentait (chapitre 4). L'augmentation de la consommation d'engrais accroît la dépendance à l'égard des engrais synthétiques. Pour limiter cet inconvénient, nous pouvons envisager d'utiliser des légumineuses en tant que CIVEs, en mélange ou en culture pure. Elles peuvent fournir une quantité significative d'azote, pilotable sous forme de digestat grâce à la méthanisation (chapitre 4). Comme les légumineuses sont moins efficaces pour réduire la lixiviation de l'azote, elles devraient probablement être utilisées en mélange avec une non-légumineuse pour maintenir les deux services de réduction de la lixiviation de l'azote et l'effet d'engrais vert. Les espèces de légumineuses doivent également être choisies avec soin afin d'atteindre le seuil de biomasse rentable pour la production de biogaz.

Volatilisation d'ammoniac

Le développement de la méthanisation des CIVEs a eu tendance à augmenter la volatilisation de l'ammoniac responsable de la formation de particules fines dans l'atmosphère, et cela était plus la conséquence de l'augmentation de la consommation d'engrais que de l'épandage du digestat (chapitres 4 et 5). Bien que le digestat soit plus sensible à la volatilisation que l'engrais minéral (chapitres 2 et 4), il semble que ce soit l'augmentation de la fertilisation qui soit principalement responsable de l'augmentation de la volatilisation (chapitre 5). La volatilisation du digestat peut être réduite en choisissant une période d'application moins favorable à la volatilisation (temps froid et pluvieux) et en l'injectant directement dans le sol (chapitre 2). Cependant, la volatilisation du digestat a probablement été sous-estimée dans l'étude à l'échelle de la France (chapitre 5) par rapport à l'étude sur la plaine de Versailles (chapitre 4).

Les caractéristiques du digestat simulé dans le chapitre 5 étaient très différentes des caractéristiques du digestat à base de CIVEs simulé dans le chapitre 4 et d'un digestat réel à base de CIVEs (Levavasseur et al. 2022b) ; il était beaucoup moins riche en azote minéral.

Emissions de N₂O

Dans les simulations à l'échelle de la France (chapitre 5), l'insertion de CIVEs a augmenté les émissions de N₂O davantage que l'extension des CIMS. Notre hypothèse initiale, selon laquelle l'exportation de la biomasse des CIVEs et son retour sous une forme plus stable diminuerait les émissions de N₂O par rapport aux CIMS laissées en place, ne s'est pas toujours vérifiée. Elle ne s'est vérifiée que si la CIVE remplaçait une culture intermédiaire dont les résidus étaient rapidement minéralisés, c'est-à-dire avec un faible rapport C:N (par exemple, la moutarde ; chapitres 2 et 4). Dans le cas où les résidus de la culture intermédiaire immobilisent l'azote (par exemple, le seigle), leur exportation a l'effet inverse puisque l'azote minéral du sol qui aurait été immobilisé par la dégradation des résidus devient disponible pour la chaîne de nitrification/dénitrification (chapitre 2). Dans les simulations à long terme, en considérant non pas ce qui se passe pendant une période d'interculture mais pendant 30 périodes d'interculture successives, une autre relation est apparue : les émissions de N₂O étaient fortement corrélées à l'augmentation de la minéralisation de l'azote organique du sol (chapitre 5). Ce compromis entre les émissions de N₂O et le stockage de matière organique dans le sol à l'origine de l'augmentation de la minéralisation de l'azote organique avait déjà été mis en évidence par Guenet et al. (2020).

Lixiviation du nitrate

A l'échelle du territoire et de la France (Chapitres 4 et 5), l'insertion de CIVEs a eu tendance à réduire la lixiviation par rapport à la situation actuelle et à l'extension des CIMS lorsque les CIVEs atteignaient suffisamment le seuil de rentabilité pour être récoltées. Ce résultat est en accord avec l'étude de Heggenstaller et al. (2008) qui a trouvé que sur une succession, la lixiviation était moins importante avec des CIVEs fertilisées qu'avec un sol nu. Pendant l'interculture, la fertilisation des CIVEs n'a pas eu d'impact sur la lixiviation parce qu'elle était appliquée après la période de drainage. Cependant, à grande échelle et sur une longue période, avec un contrôle moins précis de la période d'application et des échecs de récolte, elle pourrait augmenter la lixiviation par rapport aux intercultures laissées nues ou aux cultures intermédiaires si les CIVEs n'exportaient pas tout l'excédent d'azote. Le choix des espèces de

cultures intermédiaire s'est avéré plus critique que la gestion des cultures intermédiaire. Nous avons constaté, tant sur le terrain (chapitre 2) que dans les simulations à l'échelle de la France (chapitre 5), que la moutarde était plus efficace que le mélange céréale-légumineuse (seigle-féverole) et la céréale pure (ray-grass ou orge d'hiver). La légumineuse pure (féverole) était moins efficace que la céréale pure (orge d'hiver) dans les simulations à l'échelle de la France. Contrairement à notre hypothèse initiale, plus que la biomasse produite en fin de cycle, c'est la vitesse d'implantation et la capacité d'absorption de l'azote qui ont été déterminantes dans la réduction de la lixiviation (Thorup-Kristensen et al. 2003 ; Tribouillois et al. 2016a ; Riau et al. 2021). Möller et Stinner (2009) qui ont comparé les CIVEs avec les CIMS ont utilisé les mêmes espèces mais les ont détruites ou récoltées avant la période de drainage et n'ont pas fertilisé les CIVEs. Dans cette étude, les résidus de CIMS se sont minéralisés avant la période de drainage. Par conséquent, l'exportation des résidus et leur retour sous forme de digestat après la période de drainage ont diminué la lixiviation. L'expérience la plus proche de cette situation dans notre étude est la comparaison des modalités CIVE d'été et CIMS d'été dans l'expérimentation au champ, mais la CIVE était fertilisée (chapitre 2). La quantité d'azote lixiviée était la même dans les deux modalités. La fertilisation de la CIVE aurait parfaitement compensé l'exportation des résidus.

Consommation d'eau

Il s'avère que les CIVEs ont détérioré le service de recyclage de l'eau. Que ce soit à l'échelle d'un champ pendant une saison (chapitre 2) ou de la France entière pendant 30 ans (chapitre 5), les CIVEs ont réduit le drainage par rapport à l'interculture nue. Dans l'expérimentation au champ (chapitre 2), les CIVEs d'été ont eu un impact plus important que les CIVEs d'hiver, tandis qu'à l'échelle de la France, le drainage était davantage corrélé à la fréquence d'insertion des CIVEs d'hiver qu'à celle des CIVEs d'été. L'expérience au champ a montré une relation entre la transpiration des cultures intermédiaire et la réduction du drainage. Par conséquent, les CIMS d'été qui produisaient moins de biomasse et transpiraient donc moins que les CIVEs réduisaient moins le drainage que ces dernières. Il en allait de même pour les CIMS d'hiver si la période de drainage se prolongeait jusqu'au printemps, lorsque la croissance des cultures reprenait. Les cultures intermédiaires ont réduit l'évaporation du sol, mais pas suffisamment pour compenser l'augmentation de la transpiration. Ces résultats sont cohérents avec l'étude de Tribouillois et al. (2018) qui s'est déroulée dans la même région. Ils ont trouvé une corrélation

significative entre la production de biomasse des cultures intermédiaires, la réduction du drainage, la transpiration des cultures intermédiaires et l'évaporation du sol. Lors du passage à l'échelle de la France (chapitre 5), les espèces de cultures intermédiaires ont changé entre les CIMS et les CIVEs. Ainsi, un autre facteur de variation est apparu, à savoir l'espèce de la culture intermédiaire. La féverole semblait consommer beaucoup plus d'eau que l'orge d'hiver. Par conséquent, même si la fréquence et la biomasse des cultures intermédiaires étaient plus élevées dans le scénario d'extension des CIVEs que dans le scénario d'extension des CIMS, le drainage moyen était à peu près le même. Ce résultat peut aider à comprendre les effets confondants obtenus par Meyer et al. (2019) dans leur méta-analyse de l'impact des cultures intermédiaires sur le drainage, dans laquelle ils n'ont pas réussi à trouver un facteur explicatif significatif. Conformément à nos résultats, nous avons trouvé deux études qui ont comparé l'efficacité de l'utilisation de l'eau entre différentes espèces de cultures intermédiaires d'été dans un climat sec et ont trouvé des différences significatives (Wunsch et al. 2017 ; Berriel et al. 2022).

Les CIVEs d'hiver ont augmenté la consommation d'eau des cultures de printemps suivantes si l'irrigation sans contrainte était possible (chapitre 5). Les CIVEs d'hiver ont eu tendance à provoquer un stress hydrique sur les cultures suivantes. En conséquence, dans les zones irriguées, l'irrigation a augmenté de manière significative pour les cultures principales. L'extension des CIMS a également entraîné une augmentation de l'irrigation, mais dans une moindre mesure que l'extension des CIVEs. STICS a également simulé un stress hydrique sur du maïs pluvial suivant une CIVE d'hiver dans la plaine de Versailles (chapitre 4). Nous avons choisi de limiter l'irrigation des CIVEs d'été conformément aux recommandations d'Arvalis et aux principales pratiques des agriculteurs. Ainsi, l'irrigation à la levée autorisée dans les simulations pour la France (chapitre 5) n'a pas eu d'impact visible sur la consommation globale d'eau d'irrigation.

Production alimentaire

Les CIVEs d'hiver ont eu tendance à diminuer le rendement de la culture de printemps suivante, principalement à cause d'un stress hydrique mais aussi à cause d'un stress azoté (chapitres 4 et 5). Cependant, nous n'avons pas observé de réduction du rendement du tournesol dans l'expérimentation au champ (chapitre 2). Certaines espèces résistantes pourraient donc être préférées après une CIVE. Les CIMS n'ont pas autant réduit le rendement des cultures de

printemps (chapitre 5). D'une part, parce que le ray-grass a été détruit suffisamment tôt pour limiter la compétition préemptive et, d'autre part, parce que la féverole a fourni de l'azote à la culture suivante au lieu de l'immobiliser. Les cultures d'hiver n'ont pas semblé subir de stress azoté dans les simulations à l'échelle de la France ni dans l'expérimentation au champ. Dans les simulations à l'échelle de la France, la situation est assez particulière parce que les CIVEs d'été n'ont pas produit beaucoup de biomasse, qui a été très rarement exportée, expliquant ainsi les faibles exportations d'azote. A l'inverse, dans la plaine de Versailles (chapitre 4), les CIVEs d'été ont provoqué une compétition préemptive pour l'azote avec le blé suivant sur sol profond. Sur ce type de sol, les CIVEs ont immobilisé de l'azote qui aurait été rapidement disponible au printemps pour le blé suivant. Il est donc probable que les CIVEs d'hiver et les CIVEs d'été réduisent la production alimentaire en échange de la production de gaz renouvelable. Toutefois, la production totale de biomasse, en incluant les CIVEs, a augmenté sur l'ensemble de la succession. En outre, la réduction de la production des cultures principales n'a pas affecté le stockage du carbone organique du sol.

Bilan gaz à effet de serre

A l'échelle France entière, le bilan GES n'a été calculé que sur les systèmes de culture sans élevage (chapitre 5). Nous avons pris en compte le stockage de carbone organique dans le sol, les émissions directes et indirectes de N_2O (volatilisation de NH_3 , lixiviation de NO_3) au champ, les émissions induites par la production des engrais de synthèse utilisés au champ, les émissions liées au fonctionnement de l'unité de méthanisation (e.g. consommation d'énergie, fuites de biogaz, stockage du digestat) et les émissions évitées par la substitution du gaz fossile par du biométhane. Les émissions dues au transport des matières et aux nouvelles opérations de récolte et d'épandage n'ont pas été prises en compte. Elles ont un poids faible par rapport aux émissions déjà prises en compte (Malet et al. 2023).

Les CIVEs ont permis d'atténuer efficacement le changement climatique, puisqu'elles ont réduit les émissions de GES des systèmes de grandes cultures de 26 % à 58 %, selon le scénario de développement des CIVEs. Le développement des CIMS n'a pas permis d'obtenir la même amélioration du bilan GES. Malgré l'augmentation des émissions directes et indirectes de N_2O , l'utilisation accrue d'engrais synthétiques, et les émissions induites par le fonctionnement du méthaniseur, la méthanisation des CIVEs a eu un bilan GES positif par rapport au scénario de

référence grâce au stockage très important de carbone et à la substitution du gaz fossile, qui a eu un impact encore plus important.

Bilan sur les services et dys-services rendus

En résumé, les CIVEs ont parfois eu un impact négatif sur le rendement des cultures suivantes qui n'a pas dépassé 17% de perte de rendement. Cet impact a été mieux maîtrisé avec les CIMS. Les CIVEs ont fourni un service de fourniture d'énergie que les CIMS n'ont pas fourni, ce qui a amélioré le bilan GES par rapport au sol nu ou aux CIMS. Le stockage supplémentaire de carbone a également amélioré le bilan GES par rapport aux CIMS de manière plus efficace que prévu. Dans l'ensemble, la lixiviation du nitrate a été réduite de manière similaire avec les deux types de couverts, mais la fourchette était assez large et influencée par les espèces choisies et la durée de la période de drainage. De même, les CIVEs ont plus ou moins réduit le drainage par rapport aux CIMS en fonction des espèces considérées, du type de couverture (estivale ou hivernale) et de la durée de la période de drainage. L'irrigation sur les CIVEs a été limitée et a eu très peu d'impact sur l'utilisation d'eau globale, mais l'utilisation de l'eau par les cultures suivantes a augmenté. Contrairement à ce qui était attendu, les CIMS ont également augmenté la consommation d'eau des cultures principales en raison de la fin tardive de certaines d'entre elles. Si la fertilisation des CIVEs et l'utilisation du digestat ont augmenté la volatilisation du NH_3 , la réduction de la fertilisation minérale suite à l'utilisation des CIMS légumineuses a diminué les émissions de NH_3 . Ce dernier point n'avait pas été relevé dans la revue de la littérature.

Intérêt de la méthanisation de CIVEs dans une zone de grandes cultures

La méthanisation des CIVEs présentait plusieurs avantages pour les céréaliers par rapport à la méthanisation industrielle qui serait leur principale alternative dans nos territoires spécialisés (chapitre 4). Tout d'abord, elle leur permettait d'avoir un revenu supplémentaire provenant du biogaz et de ne pas avoir à acheter d'engrais organique. Deuxièmement, elle leur permettait d'obtenir les services écosystémiques fournis par les CIVEs : stockage de carbone et réduction de la lixiviation. Dans tous les cas, l'épandage de digestat émettait de l'ammoniac, mais le digestat à base de déchets alimentaires était plus sensible à la volatilisation que le digestat à base de CIVEs. Par contre, en ne digérant que leurs CIVEs, ils ne gagneraient pas d'azote et n'économiseraient donc pas d'engrais. Au final, l'ajout de digestat à base de CIVEs à l'engrais minéral déjà utilisé augmenterait la volatilisation globale, tandis que le digestat à base de

déchets alimentaires n'augmenterait pas autant les émissions de NH_3 en remplaçant l'engrais minéral.

Hypothèses sur l'impact du changement climatique

L'augmentation de la température devrait raccourcir le cycle des cultures principales, qui atteindront plus rapidement la somme de température nécessaire à leur maturation. L'inconvénient est qu'elles produiront moins de biomasse car elles auront moins de temps pour intercepter le rayonnement solaire. L'avantage est que, dans certaines situations, elles pourront éviter les stress (hydrique ou thermique). L'autre avantage est qu'une récolte précoce permettrait de semer plus tôt les CIVEs d'été et d'hiver. Les CIVEs produiraient plus de biomasse et les CIVEs d'été pourraient éventuellement éviter le stress hydrique. La possibilité de faire pousser des CIVEs d'été paraît tout de même remise en question par le changement climatique. L'augmentation des températures extrêmes peut avoir des effets délétères sur les CIVEs d'été en provoquant une sénescence accélérée. L'augmentation de la fréquence des sécheresses en été diminue aussi les chances de succès du semis des CIVEs d'été et la fourniture d'eau pendant la durée de vie de la CIVE. Une certaine flexibilité pourrait être possible en utilisant des espèces ayant une meilleure efficacité d'utilisation de l'eau.

Le complément à l'étude de Launay et al. (2021a) fournit quelques réponses à la question de l'impact du changement climatique sur la production des cultures intermédiaires en France, même s'il se concentre sur les CIMS (Constantin et al. 2020). Le plan de simulation a été testé avec le scénario de changement climatique RCP8.5 sur la période 2030-2060. La vesce et la féverole semées dans le scénario d'extension des CIMS ont été impactées différemment par le changement climatique. Sans changer les dates de semis et de destruction, la production de vesce en interculture d'été a diminué de 0.8 t MS.ha^{-1} , soit une diminution de 30% comparé au climat actuel, tandis que la production de féverole en interculture d'hiver a augmenté de 0.7 t MS.ha^{-1} , soit une augmentation de presque 90%.

Annexes – Post-print of the paper published in Agronomy for Sustainable Development

Incorporating energy cover crops for biogas production into agricultural systems: benefits and environmental impacts. A review.

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Abstract: Some European countries are exploring the idea of replacing dedicated crops with energy cover crops for biogas production. Indeed, energy cover crops can generate consequential biomass without competing with food crops for land use. However, the potential benefits and impacts of this choice are not fully understood. Here, we review what is known about the consequences of energy cover crop usage by examining management regimes and digestate use, including impacts on the environment and cropping system performance. First, compared to cover crops, energy cover crops are intensively managed to produce more biomass (< 5 t DM/ha vs. up to 16 t DM/ha). Second, nitrogen is conserved during anaerobic digestion and is more readily available to crops in digestate than in cover crops residues. However, ammonia is lost via volatilization, which could reduce nitrogen use efficiency, depending on the storage conditions and application method. Third, 43–80% of the crops' initial carbon is transformed into biogas. That said, levels of soil carbon storage may nonetheless resemble those obtained with cover crops left behind because carbon is stabilized during anaerobic digestion and the energy cover crops' roots and stubble are left behind in the soil. Fourth, energy cover crops can act as multiservice cover crops, reducing nitrate leaching, improving soil microbial activity, and enhancing soil physical properties during the fallow period. Fifth, energy cover crop usage can have certain disservices, such as soil compaction, the need for additional inputs (e.g., irrigation, fertilization, pesticides), reduced groundwater recharge, and reduced following crop yield. In summary, expanding the usage of energy cover crops for biogas production does not seem to be an environmental threat. However, care must be taken to avoid the intensification of irrigation and lengthening growing periods to boost biomass, which could reduce food production.

33 **Keywords:** energy cover crop, catch crop, ecosystem services, digestate, carbon storage, leaching,
34 volatilization, drainage, microbial activity

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68

1. Introduction

69 After rapidly developing in the late 2000s, anaerobic digestion-based biogas production in Europe has
70 stagnated since 2017 because of new measures and regulations in the key countries (e.g., Germany, the
71 United Kingdom), which have reduced economic incentives and limited the use of cash crops as source
72 materials (EurObserv'ER 2020). Cash crops can be included in the digester ration, as "dedicated crops"
73 or "energy crops", given that they do not surpass a certain threshold, which varies among countries (e.g.,
74 2021: 15% in France vs. 44% in Germany; Thrän et al. 2020). The use of food crops to produce energy
75 (e.g., biogas, biofuels) is a subject of debate because land use competition between food and energy
76 crops must be avoided. Consequently, Europe has introduced sustainability criteria to apply when
77 producing biomass for energy. Described in the European Union Renewable Energy Directive
78 2018/2001, these criteria specify that energy crops should not be grown in areas with high biodiversity
79 nor in soils containing high levels of stored carbon. Moreover, the use of biofuels and biogas should
80 prevent a certain proportion of greenhouse gas emissions defined by the directive. Biogas production
81 will need to meet the above criteria to be categorized as a renewable energy source (EurObserv'ER
82 2020). The sustainability of anaerobic digestion is also a key concern (WWF France 2020). It is
83 recommended that (i) all stakeholders be mobilized when a new project is launched (ii) agroecological
84 practices be implemented at the farm level; and (iii) further steps be taken to sustainably manage
85 biomass, promote a positive carbon balance, reduce greenhouse gas emissions, and preserve biodiversity
86 (WWF France 2020).

87 France is the European country in which primary biogas production has shown the most growth (+11%
88 between 2018 and 2019) (EurObserv'ER 2020). Initially employed to treat livestock farming and food
89 processing waste, anaerobic digestion is now being promoted to produce 6 to 8% of the gas consumption
90 by 2028 (Ministère de la transition écologique et solidaire 2020). To meet high demands for biomass,
91 France is employing cover crops, particularly in grain-growing areas without appreciable livestock
92 farming. According to a futures study, 30% of the country's gas needs should be met by anaerobic
93 digestion by 2050. Energy cover crops could provide one-third of the necessary biomass and serve as
94 the main source of agricultural biomass (ADEME 2018). Energy cover crops are seeded and harvested
95 between two cash crops (i.e., in systems using double cropping or growing three crops in two years). By

definition, energy cover crops are not cash crops and do not compete with food crops because they develop over a period that is usually too short to grow food crops. However, this period is already being increasingly used to produce supplementary fodder (Binder et al. 2020; Andersen et al. 2020) or even food crops due to climate change (Meza et al. 2008; Sandler et al. 2015).. Cover crops without immediate monetary return are already in use because they provide many ecosystem services; their deployment is also mandatory in zones susceptible to nitrate leaching, where they limit groundwater pollution during the long fallow period (European Union Nitrate Directive 91/676/EEC). In addition to reducing nitrate leaching (Constantin et al. 2010), cover crops also protect soils from erosion (Blanco-Canqui et al. 2015), provide nitrogen to the subsequent crop (Tonitto et al. 2006), increase carbon storage in soils (Poeplau and Don 2015), provide habitat and resources for wildlife and microorganisms (Ellis and Barbercheck 2015; Finney et al. 2017; Wilcoxon et al. 2018; Carmona et al. 2021), and, under certain conditions, can limit diseases and weeds (Blanco-Canqui et al. 2015). For this reason, they are also referred to as multiservice cover crops (Couëdel et al. 2019).

Countries other than France are also interested in exploiting energy cover crops, such as the BiogasdonerightTM initiative in Italy (Dale et al. 2016) and the Syn-Energy research project in Austria (Szerencsits 2014). However, there are some key concerns. Certain levels of biomass production are required for the process to be economically sustainable; energy cover crops need more intensive management than do conventional cover crops; and the use of inputs such as water or fertilizer is generally recommended (Marsac et al. 2019). Few studies have looked at the environmental impacts of energy cover crops, and questions remain with regards to the sustainability of anaerobic digestion, if we apply the criteria of the European Union or WWF France. For example, more information must be gathered about the relationship with land use, greenhouse gas emissions, soil organic matter storage, and biodiversity. Stated more succinctly, would energy cover crops retain their status as multiservice cover crops if utilized to produce biogas? We sought to answer this question by reviewing what is known about energy cover crops and, more specifically, by attempting to formulate predictions based on current knowledge about cover crops and digestate-based fertilizers. After providing a detailed description of energy cover crops and how they are affected by anaerobic digestion, we reviewed the effects of growing energy cover crops and using their digestates as fertilizers, examining how both interact to influence different fluxes and processes. Finally, we examined the potential impacts of energy cover crop use on cropping systems.



Figure 1. Energy cover crop experiment in Auzeville-Tolosane, France: (a) digestate obtained from energy cover crops is spread before the summer energy cover crop, sorghum, is seeded (in July) and (b) the sorghum prior to harvest (in September). Photographs by the authors.

2. Material and Methods

A literature review was carried out using the Web of Science and Google Scholar databases (accessed between January 2020 and September 2021) complemented with scientific reports that the authors were aware of. To find publications in the scope of our study, we combined different groups of keywords: “ecosystem services”, “sustainability”, “soil quality”, “ecological footprint”, “environmental impact”, “environmental assessment” for environmental impacts and benefits; “water balance”, “drainage”, “soil water content”, “water deficit”, “water stress” for water-related impacts; “nitrogen balance”, “nutrient cycling”, “nitrous oxide emissions”, “nutrient limitation”, “volatilization”, “leaching” for nitrogen-related impacts; “carbon balance”, “carbon storage”, “carbon sequestration”, “soil organic carbon” for organic matter-related impacts; “energy crop”, “green manure”, “catch crop”, “cover crop”, “double crop*” for cover crops; “fertilization” or “digestate” for the use of digestate; “anaerobic digestion”, “biogas production” to include anaerobic digestion. To complete and broaden the search, we also checked the references in the collected papers. We started the investigation with reviews and meta-

analysis of the benefits and drawbacks of cover crops on the one hand and digestates (all origins) on the other. We then restricted the reading as much as possible to papers dealing with cover crops used for biogas production or digestate derived, at least in part, from plant biomass.

3. Differences between multiservice cover crops and energy cover crops

a. Multiservice cover crops

As noted in several literature reviews, cover crops furnish numerous additional environmental benefits, which has given rise to the term multiservice cover crops. Part of the benefits is provided through soil cover and part is provided through the return of residues to the soil. Cover crops improve water quality in several ways. They reduce drainage, thus limiting pesticide contamination of groundwater (Giuliano et al. 2021). They mobilize nitrates before the drainage period, helping to preserve groundwater quality by preventing nitrate leaching (Constantin et al. 2010; Tribouillois et al. 2016). Cover crops serve the same function for sulfates (Couëdel et al. 2018b). Furthermore, the roots of cover crops increase soil porosity, both promoting water infiltration and reducing surface runoff (Blanco-Canqui et al. 2015). As a result of the latter, dissolved phosphorus is retained in the soil for the following crop, and water pollution is limited (Daryanto et al. 2018). Cover crops act as physical barriers against water and wind erosion (Blanco-Canqui et al. 2015). They can also help directly mitigate climate change because, in temperate regions, they have higher albedo than do soils; in Europe, the mitigation potential is 3.16 Mt CO₂-eq per year (Carrer et al. 2018). Cover crops suppress weeds by competing for space and resources (i.e., light, water, nutrients) (Schipanski et al. 2014; Blanco-Canqui et al. 2015; Smith et al. 2020). Finally, cover crops provide habitat and food resources for birds, insects, and microorganisms, thus promoting biodiversity (Blanco-Canqui et al. 2015). Upon destruction, cover crops residues continue to provide other services. Residue mineralization supplies nitrogen and sulfur to the following crop (Thorup-Kristensen et al. 2003; Tribouillois et al. 2016; Couëdel et al. 2018a). Leguminous cover crops fix atmospheric dinitrogen, boosting the supply available for the next crop, even if they have a lower “catch crop” effect on nitrates (Tonitto et al. 2006). When cover crop biomass (roots and shoots) is incorporated into the soil, carbon storage levels can reach 320 kg C/ha per year (Poeplau and Don 2015) or even 560 kg C/ha per year (Jian et al. 2020), further contributing to climate change mitigation. In fact, expanding cover crop use in France could reduce greenhouse gas emissions by 515 kg CO₂-eq/ha per year (Launay et al. 2021) despite a slight increase in nitrous oxide emissions also demonstrated in other studies after the incorporation of cover crops (Blanco-Canqui et al. 2015; Guenet et al. 2020; Abalos et al. 2022). If cover crop roots and shoots are left in place, they can also control pathogens and weeds via allelopathy (Snapp et al. 2005; Matthiessen and Kirkegaard 2006; Blanco-Canqui et al. 2015). Cover crops can also have deleterious effects (i.e., disservices). For example, they may engage in pre-emptive competition with cash crops for nutrients and water. Their water and nitrogen consumption, as well as cases of nitrogen immobilization after residue incorporation can lead to yield losses (Krueger et

al. 2011; Alonso-Ayuso et al. 2014; Alvarez et al. 2017; Meyer 2020). Such effects can be limited by destroying the cover crops early or increasing soil nitrogen mineralization over time by accumulating organic matter (Constantin et al. 2011; Krueger et al. 2011; Alonso-Ayuso et al. 2014; Acharya et al. 2017). Cover crops can also host pathogens, allowing their populations to persist and multiply during the fallow period, which can put subsequent crops at greater risk of diseases (i.e., the “green bridge” phenomenon). Such dynamics have led to yield losses in several cash crop-cover crop (Acharya et al. 2017). To eliminate potential green bridge effects, care should be taken to select a cover crop species that is non-host for the next crop. Finally, cover crop water consumption in the winter can also reduce drainage and, consequently, groundwater recharge and the water supply available for other uses (Meyer et al. 2019).

Depending on the specific combination of services and disservices, cover crops can have positive or negative impacts on the following crop’s yield. For example, green manure effects, pathogen control, and weed control often boost yield (Matthiessen and Kirkegaard 2006; Tonitto et al. 2006; Schipanski et al. 2014; Bergtold et al. 2017). In general, yield tends to be positively influenced, particularly over the long term (Constantin et al. 2011; Blanco-Canqui et al. 2015), and becomes less vulnerable to climatic hazards (Snapp et al. 2005; Bergtold et al. 2017).

The selection and management of cover crop species must take into consideration local climatic conditions, soils, cropping systems, and desired services. For example, if the objective is to prevent nitrate leaching, a fast-growing, non-leguminous species should be chosen, and the cover crop should be seeded early, so it can take up as much nitrogen as possible before the drainage period. To increase soil organic matter content, it is better to select a species with a long establishment period that is capable of producing large quantities of biomass. Sometimes trade-offs are observed. For example, there may be a trade-off in water quality and the supply of blue water under specific soil-climate conditions where certain agricultural practices are used (Obiang Ndong et al. 2021). In such contexts, species mixtures could be useful (Tosti et al. 2014; Tribouillois et al. 2016; White et al. 2017; Couëdel et al. 2019). Finally, incorporating cover crops into the soil early on (i.e., more than 15 days before the next crop is seeded) generally limits the risk of deleterious effects (Justes et al. 2012; Acharya et al. 2017).

b. Energy cover crops

In France, the definition of an energy cover crop is provided in Decree n°2016-929 for the Application of Article L.541-39 of the Environmental Code: it is a crop grown between two cash crops, and its biomass is harvested and anaerobically digested to produce biogas. In western Europe, energy cover crops are planted either during the summer fallow period (June to October) or during the winter fallow period (September to May). They can function as multiservice cover crops.

The fact that energy cover crops are harvested for biogas production rather than being incorporated into the soil is what differentiates them from conventional multiservice cover crops. Indeed, the function of

energy cover crops more closely resembles that of cover crops transformed into livestock feed, which are also known as “double crops.” Due to the short growing period, energy cover crops are harvested before they reach maturity. Because methanogenic potential varies little among crop species during vegetative stage, the amount of biomass harvested is the key factor determining levels of biogas production (Graß et al. 2013; Marsac et al. 2019). Aboveground biomass production is higher for species with high growth levels, such as sorghum [*Sorghum bicolor* (L.) Moench], corn [*Zea mays* L.], and sunflower [*Helianthus annuus* L.] in the summer and triticale [*x Triticosecale rimpaui* Wittm.], rye [*Secale cereale* L.], barley [*Hordeum vulgare* L.], and oats [*Avena sativa* L.] in the winter. Inputs (e.g., fertilizer, irrigation water, pesticides) can be provided to further boost biomass accumulation. During the summer fallow period, solar radiation levels are high, but seeding conditions can be challenging, particularly in the dry regions of southern Europe, which may face water scarcity during this time of year. Farmers should favor short-cycle species that are resistant to water stress, such as sunflower and sorghum, and recognize that irrigation must sometimes be employed to ensure cover crop emergence and biomass production. During the winter fallow period, biomass production mostly takes place in spring. Farmers should thus target species with explosive growth during the early spring, such as grasses. Because energy cover crops are good at accumulating biomass, they take up large amounts of nitrogen from the soil and require a moderate supply of fertilizer (40–80 kg N/ha), mainly in the form of digestate; using this approach can ensure sufficient yields without impairing the growth of the following crop (Szerencsits 2014; Marsac et al. 2019).

Much research has been dedicated to management strategies for maximizing biomass production by energy cover crops (Heggenstaller et al. 2008; Graß et al. 2013; Molinuevo-Salces et al. 2013, 2014; Negri et al. 2014; Szerencsits 2014; Igos et al. 2016; Marsac et al. 2019; Wannasek et al. 2019). Best practices for summer energy cover crops include early seeding, the use of drought-resistant species, and a sufficient supply of water at seedling emergence (Marsac et al. 2019). In the case of winter energy cover crops, biomass is largely harvested from April onwards, so a good approach is to delay harvesting as much as possible, without overly shortening the subsequent crop’s growing period (Szerencsits 2014; Marsac et al. 2019). To produce sufficient energy cover crop biomass, attention must be paid to seeding and harvesting dates, which could require farmers to redesign crop rotations. It may be enough to employ early varieties of cash crops and harvest them a few days or weeks in advance, in the case of summer energy cover crops; for winter energy cover crops, cash crops can be seeded with a slight delay. To increase the total biomass production of energy cover crops, the crop cycle can be modified either by removing winter cash crops—allowing the addition of winter energy cover crops—or by introducing winter cash crops that can be harvested before July—allowing the addition of summer energy cover crops.

With regards to environmental impacts, energy cover crops should display the same services and disservices as conventional cover crops during the soil cover period. These include reducing drainage;

protecting groundwater quality; structuring the soil; mitigating climate change thanks to higher albedo levels and enhanced carbon storage in soils (via belowground biomass); maintaining biodiversity; and controlling weeds. Questions remain with respect to service intensity. Because energy cover crops are harvested, they do not supply any benefits associated with aboveground biomass incorporation into the soil (i.e., allelopathy, green manure effects, enhanced carbon storage via aboveground biomass). It is also unclear whether the use of energy cover crops could negatively affect food production because of (i) the resulting changes to crop rotations and (ii) preemptive competition for water and nitrogen between crop types.

4. From cover crop biomass production to digestate storage

a. Cover crop biomass and nutrient absorption

Cover crop biomass production can vary markedly depending on species, management regime, soil characteristics, and climatic conditions. In the summer, it is hard to obtain dense and homogenous cover if plants are seeded in dried-out soil and water is scarce, given that cover crops are rarely irrigated. In the fall, the important limiting factors are the total number of growing degree days and levels of global radiation. Furthermore, conventional cover crops are not usually fertilized and are destroyed early on, by either frost or the farmer. In Europe, aboveground biomass rarely exceeds 5 tons of dry matter (DM) per hectare (ARVALIS - Institut du végétal et al. 2011; Justes et al. 2012; Hansen et al. 2021). In general, energy cover crop biomass is higher because of the longer growing season, the species chosen and the use of inputs (fertilizer, irrigation water, pesticides). That said, production remains highly variable for summer and winter energy cover crops (3–15 t DM/ha and 2–16 t DM/ha, respectively), depending on species, variety, pedoclimatic conditions, and management regime (Szerencsits 2014; Marsac et al. 2019). Initial results have shown that cereal-legume mixtures where the proportion of legumes does not exceed 40% did not impact the cereal yield (Marsac et al. 2019). Nitrogen levels in the aboveground biomass of cover crops differ based on species: they range from 13.6 to 52 g N/kg of dry matter for brassicas and grasses (Justes et al. 2009; Bareha et al. 2018; Hansen et al. 2021). They are higher for legumes: between 43 and 84 g N/kg of dry matter (Bareha et al. 2018; Hansen et al. 2021). Across trials conducted in France and Denmark, the total nitrogen absorbed by unfertilized cover crops ranged from 10 to 171 kg N/ha for legumes and 9 to 89 kg N/ha for non-legumes (ARVALIS - Institut du végétal et al. 2011; Hansen et al. 2021). Other studies conducted in temperate regions have found that nitrogen uptake by legumes can exceed 300 kg N/ha since the plants are not limited by levels of soil nitrogen (Thorup-Kristensen et al. 2003; Tonitto et al. 2006). Brassicas also have a high nitrogen uptake capacity, up to 300 kg N/ha (Constantin et al. 2015); furthermore, their nitrogen acquisition rates per growing degree day are higher than those of legumes when nitrogen is not limiting (Tribouillois et al. 2015). The C:N ratios of cover crops generally vary from 9 to 40 (Justes et al. 2012; Hansen et al. 2021). They are lower for legumes and/or cover crops that have experienced a short growing period or conditions of high

nitrogen availability (Justes et al. 2012). There were ranges of values for other nutrients: 2–8.2 g/kg of dry matter for phosphorus; 15–52.8 g/kg of dry matter for potassium; 0.9–4 g/kg of dry matter for magnesium; and 1–9 g/kg of dry matter for sulfur (MERCİ tool, French National Reference Database; Chambre Régionale d’Agriculture Nouvelle-Aquitaine 2020; Hansen et al. 2021). The carbon content of plant dry matter can be quite consistent (40–50%; Bertrand et al. 2019). Root:shoot ratios of carbon and nitrogen may differ greatly among cover crop species. Constantin et al. (2011) found that the root:shoot ratio for carbon was 20% for mustard [*Sinapis alba* L.] versus 72% for ryegrass [*Lolium multiflorum* Lam.]; for nitrogen, it was 6% for mustard versus 37% for ryegrass. Bispecific mixtures, especially when including legumes, tend to have intermediate values between those obtained by both species in sole crop. Tribouillois et al. (2016) and Couëdel et al. (2018b) observed nitrogen uptake values intermediate or equal to the best species for cereal-legume and crucifer-legume mixtures. The same observation was made by Couëdel et al. (2018a) for sulfur uptake. C:N and C:S ratios are always halfway between both species (Tribouillois et al. 2016; Couëdel et al. 2018b, a). A recent meta-analysis showed that it was very rare for a mixture to perform better than the best of the species alone on various criteria including biomass production and nitrogen uptake (Florence and McGuire 2020).

b. Ensiling energy cover crops

After being harvesting, fresh energy cover crop biomass is preserved as silage until it is anaerobically digested. Feedstock quality greatly affects the success of the storage process: the feedstock should have high dry matter content, high accessible carbohydrate content, and low buffering capacity (Teixeira Franco et al. 2016). Farmers should aim for dry matter content of 25–30% to limit fermentation, loss of matter and energy, and leachate formation (Teixeira Franco et al. 2016). However, this threshold is difficult to achieve when energy cover crops have a short growing period. The harvest date can be delayed to allow further declines in moisture content, but increased lodging risks may result because of poor weather conditions in the spring and fall (for winter and summer energy cover crops, respectively) (Marsac et al. 2019). That said, it remains unclear whether dry matter content affects biochemical methane potential even if it affects fermentation during ensiling (Teixeira Franco et al. 2016). In addition, silage juices could be recovered to cofeed the digester. It is known that biochemical methane potential can be affected by air exposure within the silo. To avoid any potential losses, it is important to use good management practices, such as rapid silo closure and high levels of biomass compaction (Teixeira Franco et al. 2016).

c. Fate of carbon during anaerobic digestion

During anaerobic digestion, between 20 and 95% of the substrate’s carbon content is transformed into biogas (CH₄ and CO₂), depending on substrate type (Möller and Müller 2012). The percentage is higher for plants that have undergone little to no transformation: 64% or 80% for a 100% corn substrate (Thomsen et al. 2013; Béghin-Tanneau et al. 2019) versus 46% for corn previously digested by animals

(assuming that ruminal degradation of original carbon content is 70%) (Thomsen et al. 2013). For energy cover crops, the amount of carbon converted into biogas during anaerobic digestion represents 43–74% of initial carbon content, compared to 36–41% in the case of livestock manure (Bareha et al. 2018). These figures are maxima given that the degree of degradation depends on material residence time and substrate preparation (Bareha 2018). During anaerobic digestion, microorganisms preferentially degrade the labile fraction of the organic substrates—avoiding recalcitrant molecules (e.g., lignin)—and produce stabilized metabolites (Coban et al. 2015; Möller 2015). Past research has found differences in the degree of degradation of the different organic matter fractions in energy cover crops: it is between 17 and 30% for the most recalcitrant fraction (lignin); between 32 and 72% for the intermediate fraction (cellulose + hemicellulose); and between 10 and 75% for the soluble fraction (Bareha et al. 2019).

d. Fate of nutrients during anaerobic digestion

During anaerobic digestion, the nitrogen in crop residues is largely conserved in the digestate (Möller and Müller 2012), which contrasts with the fate of carbon. Moreover, the nitrogen can change form: depending on the proportion of mineral nitrogen in the substrate, further mineralization of organic nitrogen can occur (Bareha et al. 2018). In fresh crop residues with N-NH_4^+ :total nitrogen ratios of around zero, an average of 57% of the organic nitrogen is mineralized during anaerobic digestion; this percentage is closer to 33% if residues are transformed beforehand (e.g., via ensiling or animal consumption; Bareha et al. 2018). Consequently, the digestate has a higher N-NH_4^+ content than the substrate of origin, a difference that is further accentuated for crop residue digestate versus livestock manure digestate (Möller and Müller 2012). The percentage of N-NH_4^+ varies greatly, from 4 to 82% of total nitrogen, depending on the substrate (Möller and Müller 2012; Nkoa 2014; Bareha et al. 2018; Guilayn 2018). Even for fresh cover crop residues, the final digestate displays marked variability: from 1.2 g N-NH_4^+ /kg of fresh matter for barley (31% of total nitrogen) to 6.0 g N-NH_4^+ /kg of fresh matter for vetch (71% of total nitrogen) (Bareha et al. 2018). This range of values results from differences in substrate organic nitrogen content and in organic nitrogen biodegradability. Other nutrients are similarly retained during anaerobic digestion. Levels of phosphorus range from 0.2 to 31.5 g/kg of dry matter, and levels of potassium range from 0.6 to 95 g/kg of dry matter (Möller and Müller 2012; Nkoa 2014; Guilayn 2018). Although phosphates, sulfates, and micronutrients (e.g., Fe, Mg, Ca) are mineralized, they are not necessarily more available to plants for several reasons: (i) they precipitate as phosphates, sulfide, carbonate, and hydroxides due to increases in pH; (ii) they experience sorption in the digestate's solid phase; and (iii) they undergo complexation with other compounds in solution (Möller and Müller 2012). Sulfur might be an issue for anaerobic digestion because at high doses it reduces the efficiency of the digestion and produces a corrosive gas, H_2S (Yang et al. 2016). The production of H_2S can be predicted by the C:S ratio of the substrates. Peu et al. (2012) found that above a C:S of 40, the amount of H_2S produced is treated efficiently. However, the C:S of cover crops varies between 63 and 319 for a large number of species (Peu et al. 2012; Couëdel et al. 2018a; Hansen et al. 2021), above the C:S of

pig manure at 44-51. Consequently, the use of cover crops produces H₂S but in acceptable quantities to be treated efficiently by the equipment already in place on the installations.

e. Digestate storage

Fermentation continues during digestate storage, and part of the carbon in the digestate is transformed into methane and carbon dioxide. Approximately 8% of the carbon in the raw digestate is transformed, with figures of 15% for the liquid phases and 34% for the solid phases (Bareha et al. 2021). These emissions are significant since they represent 1.43 to 10.36% of methane production for a given biogas unit (Liebetrau et al. 2010). If the digestate is not covered, this biogas is lost and contributes to greenhouse gas emissions (Balsari et al. 2013). Digestate degradation during the storage period also results in the release of nitrous oxide and ammonia, which can be limited by covering the digestate (Möller 2015; Holly et al. 2017). According to different studies, nitrogen loss ranges from 9% for uncovered raw cattle manure digestate to 6% for liquid digestate (Holly et al. 2017) and to 30% of nitrogen for uncovered raw pig slurry digestate (Sommer 1997). Based on these figures, a recent review by Walling and Vaneeckhaute (2020) found that emissions from livestock manure digestate under storage conditions ranged from less than 0.01 to 0.13 kg CO₂-eq/kg N per day. For the moment, we have not identified any figures from the digestion of cover crops.

5. How the anaerobic digestion of energy cover crops affects nitrogen fluxes

a. Nitrogen availability for crops

Upon cover crop destruction, soil levels of mineral nitrogen are often 50% lower than those associated with bare soil because cover crops absorb nitrogen, a phenomenon that is particularly pronounced in dry climates (Tribouillois et al. 2016; Alvarez et al. 2017; Meyer 2020). The level of mineral nitrogen available to the following crop is strongly correlated with cover crop termination date and winter drainage intensity. Later destruction dates result in greater differences relative to what is seen on bare soil because the absorption period is longer and growth is faster in the spring. Lower levels of drainage have the same effect because mineral nitrogen remains in the bare soil while it is absorbed by the cover crop. When levels of soil organic matter and cover crop mineralization are insufficient in the early spring, the low quantities of mineral nitrogen may induce preemptive competition, resulting in nitrogen stress when the next crop begins growing (Thorup-Kristensen et al. 2003; Marcillo and Miguez 2017). However, cover crops may furnish nitrogen to the subsequent crop via green manure effects. Green manure effects result because (i) the growing cover crop takes up mineral nitrogen that would otherwise leach away during winter and (ii) mineralization releases this nitrogen after the cover crops are incorporated into the soil, making it available to the next crop. The degree of these effects depends on the cover crop's nitrogen uptake efficiency; residue mineralization dynamics; and leaching risks during the drainage period. When legumes are used as cover crops (by themselves or in combination with non-legumes), the fixation of atmospheric nitrogen can boost green manure effects. Furthermore, the C:N

ratio can shape the level and timing of the nitrogen available for the following crop (Jensen et al. 2005; Justes et al. 2009). Residues with a ratio of less than 13 resulted in immediate net positive nitrogen mineralization; in contrast, residues with a ratio of more than 26 resulted in net nitrogen immobilization over the five-month incubation period. For residues with intermediate ratios, temporary immobilization occurred during the first few weeks but was then followed by net mineralization (Justes et al. 2009). It is difficult to reliably determine the range of nitrogen made available to subsequent crops by cover crops because of all the aforementioned factors. Justes et al. (2012) tried to establish potential orders of magnitude based on the research to date. They found available nitrogen ranges of -20 to $+10\%$, -10 to $+30\%$, and $+10$ to $+50\%$ of absorbed N from grasses, crucifers, and legumes, respectively.

When digestate is applied to fields, the nitrogen made available to crops is the sum of the mineral nitrogen present in the digestate and the relative amount of organic nitrogen that mineralizes in the months following application. This figure corresponds almost entirely to the amount of N-NH_4^+ , since only 10–20% of the organic nitrogen is mineralized within six months (based on studies of manure digestates) (Möller and Müller 2012). Since organic nitrogen is mineralized during anaerobic digestion, nitrogen in energy cover crop digestate should be more readily available than nitrogen in cover crop residues (Möller and Müller 2012; Bareha et al. 2018). For example, it has been found that a mean energy cover crop digestate input of 30 t/ha should provide 37–179 kg N-NH_4^+ /ha (Bareha et al. 2018). Moreover, digestate application could be timed to better correspond to crop demands for nitrogen.

Energy cover crops are particularly likely to provoke preemptive competition since they are harvested late and are likely to deplete the mineral nitrogen in the soil as they grow. On the other hand, avoiding the incorporation of cover crops with high C:N ratios can limit or prevent nitrogen immobilization. Producing an energy cover crop yield of 4.5 t dry matter/ha requires the uptake of 60–100 kg of nitrogen, which could cause nitrogen stress for the next crop. By providing a source of nitrogen, such as digestate fertilizer, it may be possible to both meet the needs of cover crops and reduce the risk of nitrogen stress for the subsequent cash crop (Szerencsits 2014). While the latter sometimes occurs after winter energy cover crops (Szerencsits 2014; Marsac et al. 2019), it is less common after summer energy cover crops. Indeed, the initial nitrogen requirements of winter cash crops are often low enough that nitrogen uptake by cover crops does not have a lasting impact. For example, Szerencsits (2014) did not observe any nitrogen stress after the use of a summer energy cover crop. However, in simulations comparing crop cycles with and without energy cover crops, it was necessary to fertilize winter wheat at higher levels when the cash crop followed an energy cover crop ($+30$ – 80 kg N/ha) to maintain yields at control levels (as defined in Launay et al. 2020). Thus, the risk of nitrogen stress could be lowered by providing mineral or organic fertilizer to the cash crop. Another solution could be utilizing an energy cover crop mixture containing legumes (Valkama et al. 2015).

In conclusion, anaerobic digestion can help promote the green manure effects of energy cover crops as long as the digestate contains levels of available nitrogen that can compensate for nitrogen losses during storage and digestate application. At the very least, removing residues from fields can prevent the nitrogen immobilization that can result from energy cover crops with high C:N ratios (Brozyna et al. 2013). Moreover, digestate application can be optimally timed to better respond to the nitrogen needs of cash crops (Möller and Stinner 2009).

b. Ammonia volatilization

429 **Table 1. Effect of anaerobic digestion on the intensity of ammonia volatilization after application of digestate as fertilizer across field studies. +:**
430 **significant increase in volatilization associated with digestate use; -: significant decrease in volatilization associated with digestate use; =: no change in**
431 **volatilization associated with digestate use. ¹Laboratory experiment. ²Calcium ammonium nitrate, calcium nitrate, ammonium nitrate, urea.**

Digestate substrate	Control	Application	Crop/soil applied to	Duration (days)	Effect on volatilization	Emission factor	Reference
Animal slurry and/or food waste	Animal slurry	Surface	Wheat; grassland; bare soil	2-5	+	10-40% of total N or N-NH ₄ ⁺ applied	Sommer et al. 2006 ¹ ; Möller and Stinner 2009; Nyord et al. 2012; Nicholson et al. 2017
Animal slurry with or without food waste	Animal slurry	Surface; Injected	Grassland; bare soil	4-19	=	30-42% of N-NH ₄ ⁺ applied on surface; 10% of N-NH ₄ ⁺ injected	Wulf et al. 2002; Chantigny et al. 2004
Pig slurry	Pig slurry	Surface	Bare soil; Grassland	8-10	-	10-18% of total N applied; 23% of N-NH ₄ ⁺ applied	Chantigny et al. (2007, 2009)
Energy crops with or without pig slurry	Animal slurry	Surface	Wheat; grassland; bare soil; corn	3	+	10-13% of N-NH ₄ ⁺ applied	Ni et al. (2012)
Energy crops or pig slurry or sewage sludge	Mineral fertilizer ²	Surface; incorporated; injected	Grassland; wheat/corn	2-8	+ or = if injected	10-14% of total N applied on surface; 6-29% of N-NH ₄ ⁺ applied on surface; 12% of N-NH ₄ ⁺ injected	Chantigny et al. 2007; Quakernack et al. 2012; Wolf et al. 2014; Zilio et al. 2021

432

When nitrogen fertilizer is applied, the degree of ammonia volatilization depends on fertilizer characteristics (pH, NH_4^+ content, dry matter content), soil characteristics (pH), the application method (surface vs. injection), and climatic conditions (temperature, wind speed, rainfall) (Ni et al. 2012; Möller 2015). Higher pH and NH_4^+ content both enhance volatilization (Möller 2015). Temperature and wind speed are positively correlated with the degree of volatilization because they promote ammonia's transition from its liquid phase to its gaseous phase; the occurrence of rainfall immediately after application strongly reduces volatilization (Ni et al. 2012). Since volatilization occurs at the interface with the atmosphere, injecting the fertilizer limits its exposure, reducing volatilization (Webb et al. 2010; Maris et al. 2021). Similarly, low levels of dry matter content allow the fertilizer to better infiltrate the soil, also reducing volatilization. Consequently, the identity of the crop in place when fertilizer application occurs can have an indirect effect on volatilization by slowing down infiltration; for example, the presence of corn will result in less volatilization than the presence of grasslands or wheat because the soil is largely bare under corn (Ni et al. 2012; Quakernack et al. 2012). The same thing for the presence of crop residues (Maris et al. 2021). Studies have found that volatilization is generally higher for digestate than for slurry (Table 1, S1). Increases in pH and NH_4^+ levels during anaerobic digestion should have a marked effect on volatilization. However, some work has observed a decrease in volatilization for digestate versus untreated slurry when the former is more fluid than the latter (Chantigny et al. 2004, 2007, 2009). Very little research has considered anaerobic digestion in areas with arable crops but no livestock, which means that there have been few examinations of digestate serving as a substitute for mineral fertilizer. In their meta-analysis, Pan et al. (2016) found that, on average, mineral fertilizers lose 18% of applied nitrogen to volatilization, recognizing that urea releases the most emissions and that substituting in non-urea-based fertilizers can reduce volatilization by 75%. Chantigny et al. (2007) found that digestate emitted three times more ammonia than did mineral fertilizer, while Wolf et al. (2014) and Quakernack et al. (2012) found much larger differences since emissions from the mineral fertilizer control occurred at levels deemed to be negligible. On the contrary, Zilio et al. (2021) found that with good spreading practices, i.e. injecting the digestate directly, no difference was visible with urea.

Most studies to date have looked at digestates produced from monofermented slurry. Digestates resulting from the codigestion of crop residues are more viscous than are pure slurry digestates (Plöchl et al., 2009 in Quakernack et al., 2012). Thus, infiltration-mediated reductions in volatilization are probably less pronounced for crop residue digestates. The few studies examining digestates from monofermented or cofermented crops have found that 6 to 29% of the N-NH_4^+ supplied is released via volatilization (Ni et al. 2012; Quakernack et al. 2012; Wolf et al. 2014), which falls within the value range for digestates of all origins (6–42% of the N-NH_4^+ or total nitrogen supplied). In the small number of studies where the digestate was injected, ammonia emissions drop to 4–12% of the N-NH_4^+ supplied (Wulf et al. 2002; Zilio et al. 2021; Maris et al. 2021). In addition, it is important to recognize that the use of nitrogen

469 fertilizers will increase if energy cover crops are added to rotations, a move that will enhance
470 volatilization overall. If the release of ammonia is not limited, there could be negative impacts on air
471 quality and greenhouse gas emissions.

472 c. Nitrate leaching

473 **Table 2. Effect of anaerobic digestion on the intensity of nitrate leaching across field studies. +: significant increase in leaching or leaching risk**
474 **associated with digestate use; -: significant decrease in leaching or leaching risk associated with digestate use; =: no difference in leaching or leaching**
475 **risk associated with digestate use. ¹Ammonium sulfate, ammonium nitrate, potassium nitrate. ²Pot experiment.**

Digestate substrate	Control	Application rates (using equivalents)	Agricultural practices	Effect on leaching	Reference
Animal manure and/or energy crops or food waste	Mineral fertilizer ¹	Total N or N use efficiency for one study	Applied in spring or fall to a grassland or in spring to corn	= when same N use efficiency and – when same total N applied	Matsunaka et al. 2006; Chantigny et al. 2008; Walsh et al. 2012a ² ; Svoboda et al. 2013; Tsachidou et al. 2019
Animal manure and/or energy crop and/or food waste	Animal manure	Total N or others	Applied in spring or in fall to grassland or different rotations	= for 9 studies out of 12	Jäkel and Mau 1999 in Svoboda et al. 2013; Pötsch 2004 in Möller 2015; Brenner and Clemens 2005 in Svoboda et al. 2013; Börjesson and Berglund 2007; Chantigny et al. 2008; Möller and Stinner 2009; Goberna et al. 2011; Walsh et al. 2012a; Sieling et al. 2013; Svoboda et al. 2013; Nicholson et al. 2017; Sigurnjak et al. 2017
Animal manure and/or crop residues + energy cover crops	Cover crops and crop residues left behind	Total N or others	Applied in spring; 3 out of 4 years to 1 out of 3 years	= for the highest frequency of application and – for others	Möller and Stinner 2009; Gunnarsson et al. 2011; Brozyna et al. 2013

476

477 Nitrate leaching occurs during the drainage period. The best way to minimize leaching is to keep soil
478 mineral nitrogen as low as possible before and during this time. One way to achieve this outcome is by
479 planting a fall cover crop, which will take up mineral nitrogen from the soil. On bare soils, such an
480 approach efficiently limits nitrate leaching, mainly by reducing soil mineral nitrogen but also by
481 decreasing drainage (Justes et al. 2012; Tribouillois et al. 2016; Meyer et al. 2019). The degree of
482 efficacy is species dependent: on average, non-legumes versus legumes reduce leaching by 70% and
483 40%, respectively (Tonitto et al. 2006; Tribouillois et al. 2016).

484 Fertilizer use may increase leaching risks if application takes place in the fall just before the drainage
485 period and/or if too much is employed. The European Union's Nitrate Directive (91/676/EEC) prohibits
486 applying mineral nitrogen fertilizers, as well as some organic nitrogen fertilizers (with C:N ratios < 8),
487 including digestates, to winter crops seeded in the fall. It similarly prohibits the use of all fertilizer types
488 on bare soil from early summer to February, before spring crops are planted, if no cover crops are seeded.
489 Medium- and long-term leaching risks can also be increased by higher levels of soil organic matter,
490 which can arise from repeated cover crop use or the application of organic amendments. In such cases,
491 levels of mineralized nitrogen will climb (Constantin et al. 2011) if fertilizer quantities are not
492 concomitantly reduced (Schröder et al. 2007; Constantin et al. 2012; Girault et al. 2019). In the short
493 term, using digestate as fertilizer does not increase leaching risks if fertilizer levels are calculated based
494 on the nitrogen use efficiency for the digestate and if the digestate is applied when nitrate leaching risks
495 are low, such as after the planting of a crop with high nitrogen uptake (Matsunaka et al. 2006) (Table 2,
496 S2).

497 During the winter fallow period, leaching risks can be reduced by growing energy cover crops without
498 fertilizer; the effect is similar to that obtained with cover crops. When Riau et al. (2021) tested the
499 efficacy of three energy cover crop species, they found that black oat reduced leaching more than did
500 ryegrass or forage rapeseed because the former had faster, more uniform development. When black oat
501 was grown without fertilizer as an energy cover crop and harvested in the spring, it was more effective
502 than the same species terminated early and left in the field as a cover crop (Möller and Stinner 2009;
503 Gunnarsson et al. 2011). Indeed, when residues undergo mineralization in the late fall and early spring,
504 it creates the opportunity for the nitrogen absorbed by the cover crop to leach (Tribouillois et al. 2016).
505 In contrast, the digestate created from the harvested energy cover crops is applied at a time when
506 leaching risks are lower (Möller and Stinner 2009; Gunnarsson et al. 2011). Applying fertilizer to energy
507 cover crops does not appear to diminish their ability to reduce nitrate leaching. First, Heggenstaller et
508 al. (2008) observed lower levels of leaching in systems with energy cover crops versus in systems with
509 bare soil despite the higher levels of nitrogen fertilizer usage across the crop succession. Second,
510 modeling research showed that, when identical species were used, spring-fertilized, harvested energy
511 cover crops reduced leaching more than unfertilized cover crops that were destroyed a couple of weeks
512 early (Szerencsits 2014; Malone et al. 2018). Malone et al. (2018) found that fertilized, harvested rye

reduced leaching by 18% compared to unfertilized, unharvested rye and by 54% compared to what was seen on bare soil. Similarly, Szerencsits (2014) found that, in multiyear experiments, fertilized winter energy cover crops reduced leaching by 20% compared to the same species when destroyed 15 days earlier and by 25% compared to what was seen on bare soil. This result could be explained by greater biomass production leading to a larger reduction in drainage (Szerencsits 2014) or the decrease in nitrogen mineralization due to residue removal (Malone et al. 2018). Summer energy cover crops also seem to be effective in reducing leaching during the following winter compared to what is seen on bare soil in summer (Szerencsits 2014; Girault et al. 2019).

Clearly, cover crops have a demonstrated ability to reduce leaching. Initial studies of energy cover crops suggest that they display this function, which is sometimes even enhanced. Managing energy cover crops in specific ways can affect nitrate leaching dynamics: (i) leaching can be reduced by producing more biomass and avoiding asynchrony between residue mineralization and nitrogen uptake by the following crop, and (ii) leaching may be increased in the medium to long term if nitrogen fertilizer is used. It is important to underscore that long-term research in this area remains scarce, and it is necessary to further explore the effects of crop cycle management when rotations include energy cover crops.

d. Nitrous oxide emissions

530 **Table 3. Effect of anaerobic digestion on the intensity of nitrous oxide emissions across field studies. +: significant increase in N₂O emissions associated**
531 **with digestate use; -: significant decrease in emissions associated with digestate use; =: no difference in emissions associated with digestate use.**
532 ¹Modeling.

Digestate substrate	Control	Application	Effect on N ₂ O emissions	Emissions factor	Reference
Animal slurry with or without food waste and energy crops	Animal slurry	On surface; incorporated; injected	−72% to +126%	0.08-1.9% of total N	Petersen 1999; Wulf et al. 2002; Amon et al. 2006; Clemens et al. 2006; Vallejo et al. 2006; Chantigny et al. 2007; Möller and Stinner 2009; Chantigny et al. 2010; Thomsen et al. 2010; Senbayram et al. 2014; Rodhe et al. 2015; Baral et al. 2017; Herrmann et al. 2017; Nicholson et al. 2017
Crop residues and energy cover crops and grass	Cover crops left behind		−25 to −38%	1% of total N	Möller and Stinner 2009; Szerencsits 2014 ¹

533

Nitrous oxide (N_2O) is emitted mainly during denitrification, i.e. the transformation of NO_3^- into N_2 , as an intermediate product under anaerobic conditions. A small portion of N_2O is also emitted as a co-product during nitrification, i.e. the transformation of NH_4^+ into NO_2^- then into NO_3^- under aerobic conditions (Hénault et al. 2012). Both reactions are influenced by the availability of their substrate (NH_4^+ for nitrification and NO_3^- for denitrification) (Hénault et al. 2012; Nicholson et al. 2017) and organic carbon can boost the activity of denitrifying bacteria if it is easily mobilized/degradable (Möller and Stinner 2009). Beyond that, N_2O emissions are mostly influenced by climatic conditions: temperature and moisture (Petersen 1999; Hénault et al. 2012). Soil moisture above a certain threshold promotes denitrification by creating anoxic conditions (Möller and Stinner 2009).

Within crop cycles, there is no clear consensus on the effect of cover crops on N_2O emissions (Blanco-Canqui et al. 2015; Kaye and Quemada 2017; Abdalla et al. 2019; Guenet et al. 2020). During their growth, they reduce the amount of N available to microorganisms and the amount of nitrate leached, thus reducing the risk of direct and indirect N_2O emissions. On the other hand, the decomposition of their residues after their destruction releases N_2O which tends to offset the previous effect (Viard et al. 2013; Blanco-Canqui et al. 2015; Guenet et al. 2020; Abalos et al. 2022). The magnitude of N_2O emissions depends on the C:N ratio of residues, their rate of decomposition and their incorporation or not into soil (Guenet et al. 2020; Abalos et al. 2022). For example, several studies have found an overall increase in N_2O emissions with the insertion of legume cover crops due to their low C:N (Tribouillois et al. 2018; Abdalla et al. 2019; Guenet et al. 2020).

Since digestates are richer in mineral nitrogen but poorer in labile carbon than their substrates of origin, their use as fertilizers could have contrasting impacts on nitrous oxide emissions. No consistent pattern has been seen in past research comparing the effects of digestates with their substrates of origin (Table 3, S3). When the soil is rich in carbon, either because it is covered by grassland or because of its crop history, labile carbon is no longer limiting denitrification, and digestate use is no longer advantageous (Vallejo et al. 2006; Pelster et al. 2012; Corré and Conijn 2016). Under dry conditions, nitrous oxide emissions largely arise from nitrification, whose rate outstrips that of denitrification. In this case, the supply of NH_4^+ determines the level of nitrous oxide emissions (Möller and Stinner 2009). Reviewing available studies, we found an average field emissions factor for digestates of 0.52% (0.08–1.9%) of the total nitrogen applied. This figure is lower than the reference emissions factor provided by the IPCC (1%). The digestate application method influences the emissions factor. For example, injection reduces volatilization and increases denitrification (Wulf et al. 2002; Thomsen et al. 2010).

To date, only two studies have compared the effects of using crop residue digestates to leaving cover crop residues in the field; one was a field study, and the other was a modeling study (Möller and Stinner 2009; Szerencsits 2014). They reached the same conclusion: compared to terminating and incorporating cover crops into the soil, removing energy cover crop biomass to later return it as digestate seems to

reduce nitrous oxide emissions. This difference can be explained by the lower levels of labile carbon in the digestate versus in the incorporated cover crop (Möller and Stinner 2009).

e. Synthesis of the nitrogen balance

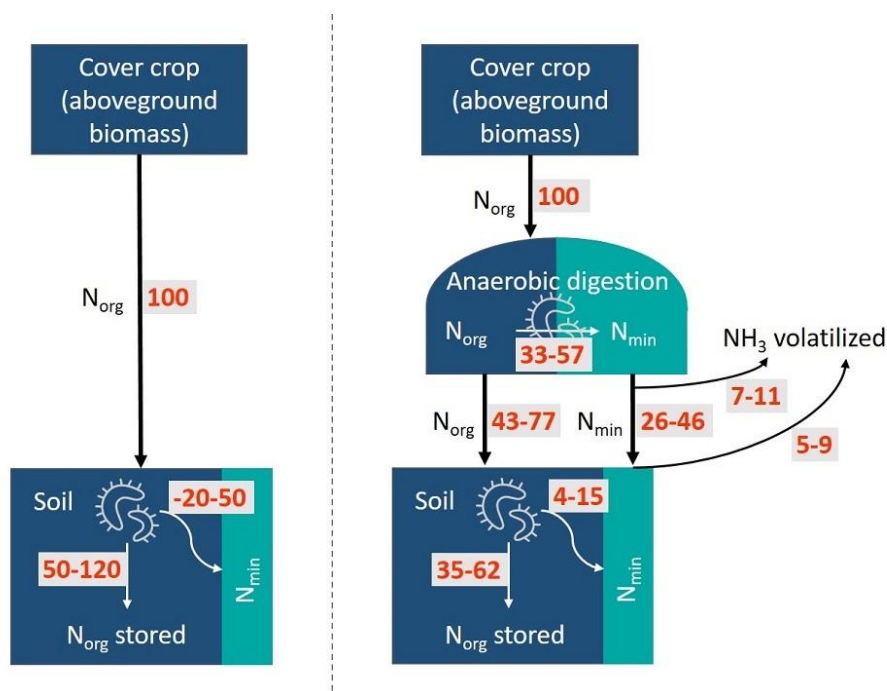


Figure 2. Schematic representation of the nitrogen balance after the incorporation of aboveground cover crop biomass into the soil (on the left) or the application of digestate obtained from anaerobically digested aboveground cover crop biomass (on the right). In both scenarios, initial cover crop nitrogen content is the same (100 units) and the mineralization period is short (5–6 months). Each number is a percentage of the original value. The cover crops' nitrogen mineralization figures were obtained from Justes et al. (2012); the nitrogen mineralization figures associated with anaerobic digestion were obtained from Bareha et al. (2018); and the digestate mineralization figures were obtained from Möller and Müller (2012). In both scenarios, we consider that ammonia volatilization took place during digestate storage and application. For the storage period, we considered that, on average, 20% of the $N-NH_4^+$ volatilized (Sommer 1997; Holly et al. 2017). For the application period, we assumed the same average level of volatilization (20%), based on figures for crop residues digestates (Table 1). The nitrogen balance represented here focuses on the mineralization of the nitrogen in cover crop residues in the soil. It does not represent nitrous oxide emissions or nitrate leaching. N_{org} : organic nitrogen; N_{min} : mineral nitrogen.

To summarize, we compared the nitrogen balance of a cropping system using multiservice cover crops and a cropping system using digestate produced via the anaerobic digestion of energy cover crops (Figure 2). When the systems start with the same initial amount of nitrogen, the nitrogen potentially available to the plants across the crop rotation depends on the amount of available nitrogen added and lost (i.e., nitrogen is preserved during anaerobic digestion). Nitrogen mineralization during anaerobic digestion and the transformation of green manures into a controllable fertilizer increased nitrogen availability. With regards to the nitrogen lost, we saw no increase in nitrous oxide emissions, an increase in volatilization after digestate creation, and a potential decrease in leaching. Overall, digestate use

seemed to slightly improve nitrogen balance, but the issue should be explored further as few studies are available for energy cover crops.

6. How the anaerobic digestion of energy cover crops affects carbon dynamics

By fixing atmospheric carbon dioxide, cover crops can increase the amount of carbon stored in the soil (Blanco-Canqui et al. 2015; Poeplau and Don 2015; Kaye and Quemada 2017; Tribouillois et al. 2018; Jian et al. 2020). Incorporating the above- and belowground biomass of cover crops could result in the storage of 320 kg C/ha per year, based on a meta-analysis by Poeplau and Don (2015), or even 560 kg C/ha per year, according to a meta-analysis by Jian et al. (2020). In France, the proportion in the cover crops of potential remaining carbon after application to soil is 28% on average, similar in magnitude to that of grain straw (Justes et al. 2012). The main factors driving carbon storage levels are the frequency of cover crop inclusion and cover crop biomass (Launay et al. 2021). Species identity also has an effect, given differences in C:N ratios. The meta-analysis by Jian et al. (2020) showed that cover crop C:N ratios tend to negatively correlate with the amount of carbon stored in the soil. Residues with high C:N ratios are hardly stabilized due to the lower carbon use efficiency of decomposers (Sinsabaugh et al. 2016) arising from stoichiometric constraints in organic matter decomposition; microorganisms C:N ratios vary generally between 6 and 11 (Bertrand et al. 2019). Furthermore, it is important to look at the distribution of carbon in aboveground versus belowground biomass. Indeed, the belowground sources of organic carbon (i.e., roots and rhizodeposition) contribute more to soil carbon levels than do aboveground sources (Chenu et al. 2019). Due to their physical and chemical nature and incorporation depths, belowground carbon sources are more effectively stabilized by adsorption or physical protection (Chenu et al. 2019).

Anaerobic digestion increases substrate stability (i.e., Stumpe et al. 2012; Wentzel et al. 2015; Coban et al. 2015; Möller 2015). The organic carbon remaining in digestate is at least 50% more stable than it is in the initial substrate (Chen et al. 2012; Thomsen et al. 2013; Béghin-Tanneau et al. 2019). Consequently, carbon sequestration is equivalent in the initial crop biomass and its various byproducts (animal digested and/or biogas plant digested) and corresponds to 12–14% of the carbon present at the start, according to Thomsen et al. (2013). Other work has found that the digestion of corn results in a sequestration level of 23% of the carbon initially present in corn; in contrast, direct incorporation of corn residues does not result in carbon sequestration but rather in the release of 4% of the initial carbon (Béghin-Tanneau et al. 2019). This result primarily arises because the fresh biomass triggered a significant priming effect on soil organic matter mineralization. A temporary inhibition of certain microbial activities after digestate application has been observed, which contrasts with the boost in response to slurry (Abubaker et al. 2015). Chen et al. (2012) had already demonstrated that digestate use had less of a priming effect on soil organic matter degradation than did crop residue incorporation.

Marsac et al. (2019) identified some factors affecting how energy cover crops can influence carbon storage. Using above- and belowground biomass data in AMG model, they observed that harvested energy cover crops could result in as much carbon storage as incorporated cover crops grown over shorter periods without fertilizer (and that thus produced less biomass). Indeed, energy cover crops grown over longer periods with fertilizer would leave behind, post harvest, quantities of stubble (1–2 t dry matter/ha depending on cutting height) and roots (~20% of total biomass) equivalent to quantities of cover crop residues. Choosing a cover crop species with a high root:shoot ratio, such as grass (Constantin et al. 2011), could (i) enhance carbon returns via belowground biomass and (ii) provide carbon more effectively stabilized than aboveground biomass (Chenu et al. 2019). Marsac et al. (2019) found that, if the resulting digestate was applied as fertilizer, the levels of stored carbon would exceed those associated with incorporated cover crops. Subsequently, Szerencsits (2014) assessed the humic balance using the above findings in conjunction with the method described in Kolbe (2007). It was found that applying the digestate derived from the aboveground biomass more efficiently stored carbon than leaving the biomass in place as residues.

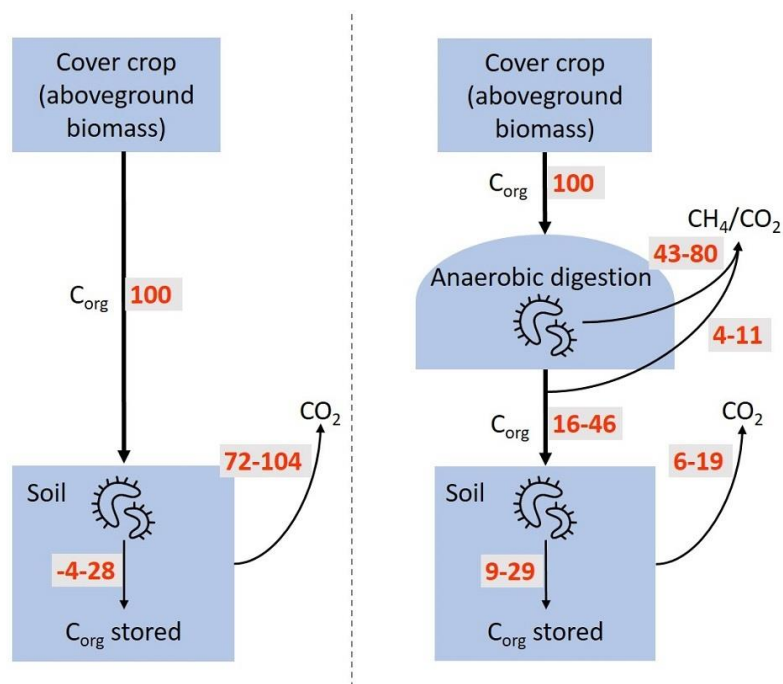


Figure 3. Schematic representation of the carbon balance after the incorporation of aboveground cover crop biomass into the soil (on the left) or the application of digestate obtained from anaerobically digested aboveground cover crop biomass (on the right). In both scenarios, initial cover crop carbon content is the same (100 units). Each number is a percentage of the original value. The figures for the carbon mineralization of cover crop residues are based on the decomposition of corn (i.e., serving as a summer energy cover crop) in Thomsen et al. (2013) and Béghin-Tanneau et al. (2019) and on the decomposition of winter cover crops in Justes et al. (2012). The figures for the cover crop decomposition via anaerobic digestion were taken from Thomsen et al. (2013), Bareha et al. (2018), and Béghin-Tanneau et al. (2019). We assumed that, on average, 20% of the digestate's carbon was lost during storage (Bareha et al. 2021). The figures for the decomposition of cover crop digestate were taken from Thomsen et al. (2013) and Béghin-Tanneau et al. (2019). C_{org} : organic carbon.

In conclusion, our initial results suggest that the use of energy cover crops can have rather positive impacts on carbon storage, when the results are compared to those for cover crops incorporated into the soil (Figure 3). Although some carbon is lost during anaerobic digestion, net levels of soil organic carbon are seemingly unaltered because (i) biomass production increases, increasing the amount of carbon returned below ground and (ii) the remaining carbon is stabilized during anaerobic digestion. However, this assessment is based on a handful of studies. Some results are still being discussed and investigated, such as the extent of carbon stabilization during anaerobic digestion compared to carbon losses in biogas or the amount of above-ground and root biomass left in the field by energy cover crops compared to traditional cover crops.

7. How the anaerobic digestion of energy cover crops affects soil biological activity

a. Microbial activity

During their growth, cover crops increase microbial abundance and activity via their inputs of carbon from root exudates and root turnover (Elfstrand et al. 2007; Blanco-Canqui et al. 2015; Finney et al. 2017). A recent meta-analysis from Muhammad et al. (2021) found that cover crops significantly increase microbial biomass compared to a bare soil from 24 to 51% depending on the indicator. On average, they increase the abundance of bacteria by 15% and the abundance of fungi by 19%, thus increasing the fungi/bacteria ratio. In general, non-legumes increase the abundance of microorganisms slightly more than legumes due to higher C substrate supply through higher biomass production. In addition, bacteria and fungi respond differently to these two groups of species. Non-legumes favor fungi because they are specialized in the decomposition of high C:N residues, whereas bacteria specialized in low C:N residues are favored by legumes. Among fungi, arbuscular mycorrhizal fungi (AMF) are particularly important for crop production because they improve nutrient uptake and provide resistance to drought and soil pathogens (Soti et al. 2016). In addition, they can have a stabilizing effect on the soil by entangling soil particles with their mycelium or by sticking them together with glomalin, which is a glycoprotein produced by AMF that acts as a glue (García-González et al. 2018). Because they live in symbiosis with their host plants, fallow periods are particularly detrimental to mycorrhizal fungi (Soti et al. 2016). On average, cover crops increase AMF abundance, AMF root colonization and AMF spore density by 26%, 13%, and 47%. Legumes had slightly less effect than non-legumes because increased N returns may be deleterious to AMF root colonization (Muhammad et al. 2021). Schipanski et al. (2014) calculated that after a winter cover crop, the roots of the following crop were colonized at 100% of their potential by AMF against 85% if the soil was left bare during winter. Cover crops residue management has an impact on microbial community abundance and structure. Exporting residues as well as leaving them on the surface reduces the abundance of bacteria (+10% relative to bare soil) compared to incorporating them (+25% relative to bare soil). The abundance of fungi is not impacted, in all cases it is increased. But on AMF in particular, exporting residues improves root colonization less

(+5%) than incorporating residues (+50%) and seems to have a little less effect than residues left on the surface (+10%) (Muhammad et al. 2021). Finally, cover crops also tend to increase the size of earthworm populations, resulting in increased water infiltration and soil aggregate stability (Blanco-Canqui et al. 2015).

In the short term, digestate use tends to increase soil microbial activity, compared to the use of mineral fertilizers or no fertilizers, although the boost is less than that provided by undigested substrates (based on measurements of induced respiration; Fuchs et al. 2008; Abubaker et al. 2015; Möller 2015; Gómez-Brandón et al. 2016; Risberg et al. 2017). This climb in microbial activity is not due to the digestate adding microorganisms to the soil since such microorganisms do not persist in the soil (Fuchs et al. 2008; Stumpe et al. 2012; Coelho et al. 2020). Nor is it correlated with the quantity of carbon supplied (Abubaker et al. 2015). Instead, it is associated with the quality of carbon supplied (Stumpe et al. 2012; Wentzel and Joergensen 2016). DNA analysis and the quantification of taxon-specific growth rates have revealed that a shift may occur in microbial communities due to the lack of readily degradable organic matter (Chen et al. 2012; Abubaker et al. 2013). Fast-growing microorganisms (r-strategists) that preferentially degrade labile organic matter disappear; they are replaced by slow-growing microorganisms (K-strategists) that more efficiently degrade recalcitrant organic matter. This change induces a modification in the ratio of fungi to bacteria (Chen et al. 2012). Differences between treatments tend to fade a few months or years into digestate use (Walsh et al. 2012b; Abubaker et al. 2013; Möller 2015; Gómez-Brandón et al. 2016; Sadet-Bourgeteau et al. 2018). Consequently, a single dose of exogenous organic matter has a temporary effect on microbial communities, depending on dose size. In the case of repeated applications, the effects on microbial communities can be long lasting and associated with changes in soil chemical characteristics such as pH, cation exchange capacity, and soil organic carbon (Sadet-Bourgeteau et al. 2018). Several studies have shown that soil type also has a significant impact: clay soils are more resilient than sandy soils (Walsh et al. 2012b; Abubaker et al. 2013; Wentzel et al. 2015). If a cover crop is in place when fertilizer is applied, microbial population size is not directly affected by the fertilizer's physicochemical characteristics, but is rather indirectly affected by the characteristics' impact on plant growth (Terhoeven-Urselmans et al. 2009; Walsh et al. 2012b; Abubaker et al. 2013).

b. Earthworms

Rollett et al. (2020) observed a positive correlation between the amount of organic matter supplied and the increase in earthworm population size. Sizmur et al. (2017) has shown that it is the quantity of energy provided by the organic matter that matters most; it is therefore organic matter quality that strongly affects earthworms. Digestate is a source of food for earthworms, particularly anecic earthworms. In the short term, digestate use increases earthworm abundance (Clements et al. 2012) and biomass, as seen in field and microcosm studies (Ernst et al. 2008; Koblenz et al. 2015; Sizmur et al. 2017). Endogeic

earthworms are not able to directly consume organic matter from digestate (Ernst et al. 2008), but they can still benefit from the input of energy by consuming the waste generated by anecic earthworms (Koblenz et al. 2015). In some cases, short-term mortality has resulted from the high quantity of ammonium introduced by larger doses of digestate or slurry ($> 170 \text{ kg N/ha}$) (Johansen et al. 2015; Tigrini et al. 2016; Renaud et al. 2017; Rollett et al. 2020). Sizmur et al. (2017) showed that, when equivalent levels of carbon were used, straw increased the biomass of an anecic earthworm, *Lumbricus terrestris*, significantly more than did plant digestate because of the higher energy input. Similarly, in a field study, Frøseth et al. (2014) observed that the immediate incorporation of green manure increased the size of the earthworm population compared to the use of plant digestate. In the long term, such differences seem to disappear (Johansen et al. 2015; Koblenz et al. 2015; Rollett et al. 2020). However, there are no long-term studies on the impacts of directly incorporating cover crop biomass into the soil versus returning later in the form of digestate.

8. The impact of energy cover crops and their digestate on water dynamics

Cover crops can have complicated effects on groundwater recharge. First, by covering the soil, they can increase transpiration and reduce evaporation (Qi and Helmers 2010; Nielsen et al. 2015; Tribouillois et al. 2016). Second, they can increase water infiltration and reduce runoff (Snapp et al. 2005; Blanco-Canqui et al. 2015; Yu et al. 2016). A recent meta-analysis by Meyer et al. (2019) found that, in most studies, cover crops decreased drainage, although the results were highly variable (-110 to $+40 \text{ mm}$). Depending on climatic conditions, this reduction in drainage may represent a small or a large percentage of annual water drainage, which could have major implications for water recharge in dry regions. Cover crop biomass seems to be one of the main determinant factors, with seeding date close behind (Meyer et al. 2020; Tribouillois et al. 2018). Tribouillois et al. (2018) observed that increases in cover crop biomass were strongly correlated with increases in evapotranspiration and decreases in drainage. However, at a certain threshold of biomass ($< 2.5 \text{ t dry matter/ha}$) or leaf area index values, evapotranspiration showed no further increases (Meyer et al. 2020). Based on this work, advancing seeding by one month can result in a threefold difference in the degree of drainage reduction; the termination date does not affect drainage but does affect soil water levels for the next crop. Based on these findings, the large quantities of biomass produced by energy cover crops should not significantly reduce drainage, compared to what is seen for multiservice cover crops. However, the seeding date should be chosen so as to trade off between biomass production and groundwater recharge. In any case, the broader-scale use of cover crops (whether multiservice or energy) could create challenges for groundwater recharge, an issue should be assessed.

The above increase in evapotranspiration could result in a water deficit for the following summer crop. The depletion of water reserves in surface has often been seen in association with multiservice cover crops or energy cover crops terminated/harvested in the spring (Krueger et al. 2011; Alonso-Ayuso et

al. 2014; Blanco-Canqui et al. 2015; Marsac et al. 2019; Meyer et al. 2020). During the first months after cover crop seeding, water profiles are generally the same for fields with cover crops and fields with bare soil because of heavy rainfall (Alonso-Ayuso et al. 2014). However, in the spring, the profiles begin to differ as the cover crop grows, notably if rainfall levels do not compensate for evapotranspiration levels (Alonso-Ayuso et al. 2014; Meyer et al. 2020). The factor with the greatest impact is termination date (Krueger et al. 2011; Alonso-Ayuso et al. 2014; Meyer 2020), followed by crop species, and amount of precipitation (Meyer 2020). Thus, the next crop is likely to face water stress if termination takes place later; if the cover crop produces large quantities of biomass; if climatic conditions are dry; and if water storage capacity is low. Apart from this latter situation, water stress appears to be minimal in the temperate zone, even if termination occurs at a later date, because the soil (particularly the first centimeters) has time to recharge before the next crop is established (Szerencsits 2014; Blanco-Canqui et al. 2015; Marsac et al. 2019; Meyer 2020). These findings suggest there is a risk of water stress for the subsequent crop when cultivating energy cover crops under specific circumstances (i.e., late destruction and high levels of biomass production), an issue that should be studied further. Mean quantities of digestate (30 m³/ha) contain less than a millimeter of water, which does not at all compensate for water depletion by cover crops. With regards to summer energy cover crops, it is theoretically possible for them to reduce soil temperatures, thus reducing evaporation and leading to greater water reserves than what is seen on bare soil (Blanco-Canqui et al. 2015). While such a result was observed for a multiservice cover crop at one site during a dry year (Blanco-Canqui et al. 2015), it was not observed for energy cover crops at an experimental site in southwestern France (Marsac et al. 2019). More studies are needed on this topic.

9. The impact of energy cover crops and their digestate on soil physical properties

Because they provide cover during periods when the soil would usually be left bare, cover crops reduce wind and water erosion. They are particularly effective at protecting sensitive soils, such as sandy soils (Snapp et al. 2005). Using a meta-analysis, Daryanto et al. (2018) found that, on average, cover crops reduce the amount of soil lost by 75%, compared to situations in which the soil is left bare over the winter. In their study, Du et al. (2022) even found an average reduction of 90% at different points of the globe. The determinant factors were the degree of cover and cover duration (Snapp et al. 2005; Blanco-Canqui et al. 2015). By reducing erosion, cover crops also reduce the loss of dissolved nutrients (e.g., phosphorus and nitrate) via runoff (Blanco-Canqui et al. 2015). In addition to directly protecting the soil from the disturbance caused by rain and wind, cover crops improve soil structural stability (Blanco-Canqui et al. 2015; Daryanto et al. 2018). Cover crops rapidly increase the stability of aggregates (< 3 years) by protecting them against the impacts of raindrops; by providing root-mediated carbon inputs; and by boosting microbial activity (Blanco-Canqui et al. 2015). Increasing aggregate stability subsequently increases water retention, carbon storage, macroporosity, and root growth; it decreases the soil's susceptibility to compaction (Blanco-Canqui et al. 2015).

Few studies have looked at the impact of digestate use on the physical properties of soils (Möller 2015). Albuquerque et al. (2012) performed a two-year experiment but found no effect of digestate use on structural stability when compared to other treatments (i.e., no amendment, mineral fertilizer, or cattle manure). Some studies cited in Möller (2015) found a positive effect of digestate use on bulk density, hydraulic conductivity, water retention capacity, and aggregate stability, compared to what was seen for unamended soil. Béghin-Tanneau (2020) also observed an increase in aggregate stability following digestate application over periods of 12 to 265 days. However, the digestate had a significantly weaker effect than its substrate of origin (corn). Similarly, Sarker et al. (2018) found that while digestate use increased aggregate stability, the effect was less pronounced than that seen for alfalfa residues. These results were attributed to a correlation between the decomposability of the organic residues and both soil microbial activity and aggregate stability.

Consequently, introducing energy cover crops into crop cycles and utilizing the resulting digestate as fertilizer should help reduce erosion and promote aggregate stability. As these services are furnished during the growing period, they will be unaffected by the fact that energy cover crops are harvested rather than being left in place. On the contrary, service quality should be better than that provided by conventional multiservice cover crops because their magnitude is positively correlated with biomass. That said, energy cover crops have one drawback compared to multiservice cover crops: when three crops are cultivated in two years instead of two, field traffic climbs, increasing the risk of soil compaction (Peters et al. 2016; Quennesson and Decaux 2020). Ensiling the energy cover crops and applying the digestate (Duttmann et al. 2014; Lantz and Börjesson 2014) requires the use of heavier machinery, sometimes under sensitive conditions during the early spring or fall. The risk of soil compaction is particularly high on clay soils and can lead to yield losses (Lantz and Börjesson 2014). However, this risk can be reduced by using tank-free spreading systems (Lantz and Börjesson 2014) or controlled traffic farming systems for silage operations (i.e., the equipment always follows the same path) (Duttmann et al. 2014). Moreover, commonly used energy cover crop species are rarely taproot species, which are able to loosen the soil (Chen and Weil 2010; Blanco-Canqui et al. 2015).

10. Impacts on cropping systems and farms

a. Food/feed production

In the previous sections, we noted that energy cover crops can reduce the yield of subsequent crops because of preemptive competition for water and nitrogen and because of increased soil compaction risks. However, the greatest potential deleterious effect of energy cover crops on subsequent crops is associated with the delay in seeding and the use of early varieties (Szerencsits 2014; Marsac et al. 2019). Szerencsits (2014) observed that the yield of spring crops declined by an average of 10% if seeding was delayed by more than 7 days, and Marsac et al. (2019) observed a 7% loss in yield if the delay attained 10–15 days. When the delay was even longer (one month or more), the next cash crop could not reach

maturity before harvest. Thus, the cash crop can no longer feed humans but can be used to feed animals or can undergo anaerobic digestion (Graß et al. 2013; Peters et al. 2016; Quennesson and Decaux 2020). In such systems, the objective is to optimize the production of both crops in tandem, and both crops are harvested before maturity. In such cases, there is no longer a clear distinction between the cash crop and the cover crop. These systems do not align with the intended purpose of energy cover crops, which is to produce biomass for energy purposes without replacing food crops. The widespread use of such systems in areas where food crops are grown could end up reducing overall food production (Kemp and Lyutse 2011; WWF France 2020).

b. Nitrogen balance at the farm level

A survey program in France contacted farmers with anaerobic digesters and obtained data to calculate the nitrogen balance on their farms. Unfortunately, most were livestock farmers, and only a small number (9 out of 46) had introduced energy cover crops to their crop rotations. None of these nine farms increased their mineral fertilizer consumption following the introduction of anaerobic digestion and energy cover crops. Four of them even reduced their mineral nitrogen fertilizer purchases (ADEME and Solagro 2018). However, this survey does not allow us to isolate the impact of energy cover crops. Anaerobic digestion is often accompanied by other changes in farm practices and, above all, by exchanges of materials with neighboring farms, industries and collectivities. In addition, these farms still have insufficient hindsight on their new production system to observe long-term effects on soil fertility. According to field trials or simulations, introducing a third fertilized crop within a two-year rotation would likely mean an increased need for nitrogen (Heggenstaller et al. 2008; Igos et al. 2016; Berti et al. 2017; Girault et al. 2019). Additional nitrogen would be required to meet the energy cover crop's needs and, possibly, to compensate the following crop for nitrogen lost between the ensiling of the energy cover crop and the application of the resulting digestate; there could also be preemptive competition for nitrogen. The use of synthetic fertilizers could be reduced by codigesting farm-derived biomass with externally derived biomass or by using legumes alone or in mixture as energy cover crops.

c. Life cycle assessment

A recently released Life Cycle Assessment (LCA) found that an anaerobic digestion scenario with 50% energy cover crops in the feedstock supply performed better than a non-biogas scenario with multiservice cover crops on indicators of energetic resource depletion, climate change and ozone depletion (Esnouf et al. 2021). This study considered the production of heat energy through the combustion of methane injected into the network, the management of livestock effluents and soil fertilization. The finality of the biogas produced had a strong impact on these indicators. In LCAs studying biogas transformed by cogeneration, the poor valorization of heat completely degrades the environmental balance of the anaerobic digestion (Bacenetti et al. 2016; Hijazi et al. 2016). If we look at the greenhouse gas balance in more detail, the studies agree that double cropping increased i) nitrogen

fertilizer use and therefore N₂O emissions in the field or CO₂ emissions upstream and ii) field operations and thus CO₂ emissions from fuel combustion (Igos et al. 2016; Berti et al. 2017; Maier et al. 2017; Esnouf et al. 2021). However, the soil C storage and above all the substitution of fossil gas largely compensated these side-effects in the study of Esnouf et al. (2021) where the anaerobic digestion scenario reduced by 75% the greenhouse gas emissions. The indicators of fine particle emissions, environmental acidification and terrestrial eutrophication that were also measured in this study depend to a very large extent on ammonia emissions during storage and spreading of the effluent and digestate. In this case, adopting anaerobic digestion with good storage and spreading practices improved the performance on these indicators compared to the reference scenario. Conversely, not covering the digestate could increase greenhouse gas emissions by 80% (Bacenetti et al. 2016; Esnouf et al. 2021). Finally, still in the same study, for indicators related to electricity consumption and fertilizer consumption, the performance was worse with anaerobic digestion but the introduction of legumes in energy cover crops and the optimization of digestate spreading equipment could reduce these impacts by 10 to 50%. The use of legumes had already been noted to reduce the greenhouse gas balance related to the reduction of the use of synthetic fertilizers and the reduction of N₂O emissions (Stinner 2015). Other LCAs exist in the literature but they rather study energy crops whose impact on land use change strongly influences the performance on the climate change indicator (Bacenetti et al. 2016; Igos et al. 2016; Hijazi et al. 2016). Styles et al. (2015) compared different energy production systems, including one in which corn serves as a summer energy cover crop or as a simple energy crop. They found that the greenhouse gas balance of the first system was rather neutral compared to the baseline system. In contrast, anaerobically digesting dedicated crops increased emissions, notably because additional land was needed to compensate for the loss in food production.

11. Conclusions

This review reveals that the use of energy cover crops and their digestates has several advantages. In addition to allowing the production of renewable energy, the crops can provide several ecosystem services, including improved water quality, climate change mitigation, reduced soil erosion, and increased microbial activity. Thus, to answer the question raised in the introduction, we can still consider them as multiservice cover crops. However, they could also have some disservices, such as reduced groundwater recharge and the need for increased nitrogen inputs. Furthermore, energy cover crops are not always used as intended, leading to competition with food crops. Energy cover crops can compete with food crops for water and nutrients. This competition can be limited if there are sufficient levels of spring precipitation and if fertilizer is used. Additionally, incorporating energy cover crops into rotations induces changes in cropping systems that can lead to certain excesses, where energy cover crop production is favored to the detriment of food crop production. Cropping systems should be designed so as to maximize the non-energy-related services provided by energy cover crops. Alternatively, trade-offs should be identified, such that energy cover crops can be treated more as multiservice cover crops

than as cash crops. In this way, farmers would view energy cover crops not only as a new income source, but also as a way to improve their fields over the long term (e.g., via increased soil organic matter, improved soil structural stability, decreased pest pressure). Some research is still needed to expect widespread adoption of energy cover crops by farmers. We can suggest a few leads. At the varietal selection level, improvements are possible to adapt forage species to double cropping. In terms of technical management, the problem of summer cover crops establishment need a solution; seeding under cover could be an opportunity to explore. Finally, at the academic level, we have a great deal of knowledge about cover crops and digestates that allows us to speculate on the impacts of energy cover crops. They remained to be confronted with the field in a wide variety of situations.

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Insertion of energy cover crops in cropping systems in France: multi-scale assessment of production potential and water - nitrogen - carbon impacts.

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Agricultural anaerobic digestion (AD), a biogas production process, is currently expanding rapidly in Europe thanks to the support of public policies. Cover crops, i.e. crops grown between two main crops, could provide a considerable amount of biomass for anaerobic digestion, especially on farms without livestock. Until now, cover crops were neither fertilized nor harvested, and were sometimes mandatory in order to limit groundwater contamination because they capture nitrate (catch crops, CCs). They can also provide other ecosystem services (multi-services cover crops, MSCCs). With the development of anaerobic digestion, the management of cover crops is changing to produce more biomass. They are fertilized, potentially with AD digestate, and harvested before maturity. They are then referred to as energy cover crops (ECCs).

There are two questions concerning ECCs: (i) on the amount of energy that ECCs could produce on a national scale, a question addressed via the evaluation of potential biomass production on a French scale, (ii) on the impacts associated with their intensification and the return of digestates to the soil in comparison with MSCCs and CCs. We studied the impacts studied concerning main crop production, greenhouse gas emissions, nitrate leaching, ammonia volatilization, changes in soil organic matter stocks, and water consumption.

To this end, we set up an approach combining field experimentation and modeling, and then deployed a modeling chain on a territorial scale over the long-term and subsequently throughout France. The STICS model was used to simulate crop production and the impact of cropping systems on environmental variables. It was coupled to the ALFAM2 model for ammonia volatilization after organic fertilizer application and to the SYS-Metha model for the AD process of ECCs.

We found that ECCs could be a significant source of energy in France. Depending on the assumptions made about the level of deployment of ECCs, the harvesting threshold and the cutting height, the resource could be multiplied by seven (from 17 to 115 TWh) and significantly exceed previous estimates (from 6 to 55 TWh). Regarding environmental impacts, ECCs are indeed a lever for climate change mitigation despite their fertilization and the associated increase in N_2O emissions. In addition to substituting fossil gas with biomethane, they store more C in soils than MSCCs due to their larger biomass. They also reduce nitrate pollution but to a lesser extent than CCs because the species chosen are less efficient. It is very likely that ECCs will reduce food production by reducing the yield of the spring crops that follow them. We observed both nitrogen stress that could be compensated for by increased fertilization and water stress compensated for by increased irrigation. MSCCs can also have a negative impact on the yield of the following crop, but their flexible management offers several levers of avoidance (e.g. choice of species, destruction date). Increased fertilization in crop sequences integrating ECCs decreases the mitigation of fossil fuel dependence and increases ammonia volatilization and thus air pollution. Finally, both MSCCs and ECCs increase water consumption, which decreases potential groundwater recharge and water availability for the next crop. The species of cover crop used is the first factor of variation, followed by the quantity of biomass produced.