Incorporating energy cover crops for biogas production into agricultural systems

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REVIEW ARTICLE

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.

Keywords Energy cover crop · Catch crop · Ecosystem services · Digestate · Carbon storage · Leaching · Volatilization · Drainage · Microbial activity

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Contents

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 UK), 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 2018). 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; Wilcoxen 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 (Fig. 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.

3 Differences between multiservice cover crops and energy cover crops

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 CO_2-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 CO_2-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

Fig. 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.

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).

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 *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

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 (MERCI 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. 2018a, 2018b). 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).

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).

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).

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-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 H_2S but in acceptable quantities to be treated efficiently by the equipment already in place on the installations.

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 CO_2-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

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. 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).

5.2 Ammonia volatilization

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 fertilizers will increase if

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. Laboratory experiment. [2]Calcium 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

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.

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. 2016; 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. 2016).

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, S2).

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

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. [1]Ammonium sulfate, ammonium nitrate, potassium nitrate. [2]Pot 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

leach (Tribouillois et al. 2016). 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.

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).

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).

Table 3 Effect of anaerobic digestion on the intensity of nitrous oxide emissions across field studies. +: significant increase in N_2O emissions associated with digestate use; −: significant decrease in emissions associated with digestate use; =: no difference in emissions associated with digestate use. [1]Modeling.

Digestate substrate	Control	Application	Effect on N_2O 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]

5.5 Synthesis of the nitrogen balance

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 (Fig. 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

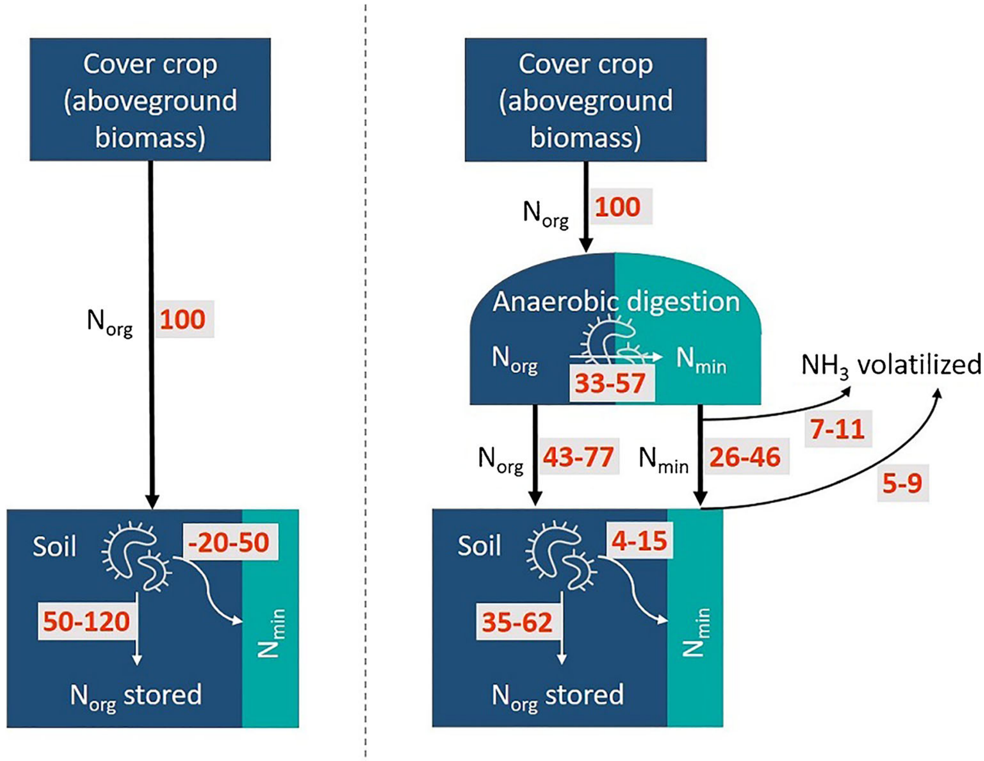

Fig. 2 Schematic representation of the nitrogen balance after the incorporation of aboveground cover crop biomass into the soil (on the left) or the application of digestate obtained from anaerobically digested aboveground cover crop biomass (on the right). In both scenarios, initial cover crop nitrogen 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.

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.

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 (Fig. 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

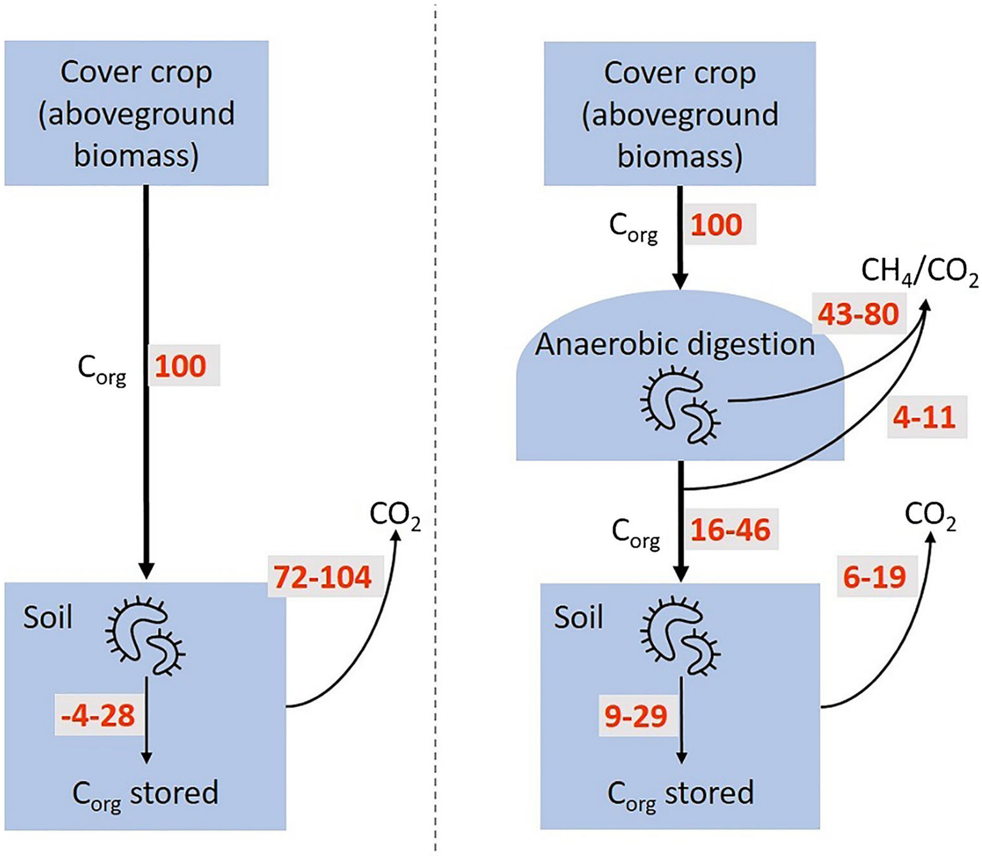

Fig. 3 Schematic representation of the carbon balance after the incorporation of aboveground cover crop biomass into the soil (on the left) or the application of digestate obtained from anaerobically digested aboveground cover crop biomass (on the right). In both scenarios, initial cover crop carbon 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.

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

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).

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 kg N/ha) (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.

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 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 m^3/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). Alburquerque 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

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).

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.

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.

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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References

Abalos D, Rittl TF, Recous S, Thiébeau P, Topp CFE, van Groenigen KJ, Butterbach-Bahl K, Thorman RE, Smith KE, Ahuja I, Olesen JE, Bleken MA, Rees RM, Hansen S (2022) Predicting field N2O emissions from crop residues based on their biochemical composition: A meta-analytical approach. Sci Total Environ 812:152532. https://doi.org/10.1016/J.SCITOTENV.2021.152532

Abdalla M, Hastings A, Cheng K, Yue Q, Chadwick D, Espenberg M, Truu J, Rees RM, Smith P (2019) A critical review of the impacts of cover crops on nitrogen leaching, net greenhouse gas balance and crop productivity. Glob Chang Biol 25:2530–2543. https://doi.org/10.1111/gcb.14644

Abubaker J, Cederlund H, Arthurson V, Pell M (2013) Bacterial community structure and microbial activity in different soils amended with biogas residues and cattle slurry. Appl Soil Ecol 72:171–180. https://doi.org/10.1016/j.apsoil.2013.07.002

Abubaker J, Risberg K, Jönsson E, Dahlin AS, Cederlund H, Pell M (2015) Short-term effects of biogas digestates and pig slurry application on soil microbial activity. Appl Environ Soil Sci 2015:1–15. https://doi.org/10.1155/2015/658542

Acharya J, Bakker MG, Moorman TB, Kaspar TC, Lenssen AW, Robertson AE (2017) Time interval between cover crop termination and planting influences corn seedling disease, plant growth, and yield. Plant Dis 101:591–600. https://doi.org/10.1094/PDIS-07-16-0975-RE

ADEME (2018) Un mix de gaz 100 % renouvelable en 2050 ? Rapport. https://librairie.ademe.fr/energies-renouvelables-reseaux-et-stockage/1548-mix-de-gaz-100-renouvelable-en-2050%2D%2D9791029710476.html?search_query=gaz&results=488. Accessed 27 Apr 2021

ADEME, Solagro (2018) La méthanisation, levier de l'agroécologie, Synthèse des résultats du programme MéthaLAE. https://solagro.org/travaux-et-productions/references/methalae-comment-la-methanisation-peut-etre-un-levier-pour-lagroecologie. Accessed 27 Apr 2021

Alburquerque JA, de la Fuente C, Campoy M, Carrasco L, Nájera I, Baixauli C, Caravaca F, Roldán A, Cegarra J, Bernal MP (2012) Agricultural use of digestate for horticultural crop production and improvement of soil properties. Eur J Agron 43:119–128. https://doi.org/10.1016/j.eja.2012.06.001

Alonso-Ayuso M, Gabriel JL, Quemada M (2014) The kill date as a management tool for cover cropping success. PLoS ONE 9: e109587. https://doi.org/10.1371/journal.pone.0109587

Alvarez R, Steinbach HS, De Paepe JL (2017) Cover crop effects on soils and subsequent crops in the pampas: A meta-analysis. Soil Tillage Res 170:53–65. https://doi.org/10.1016/j.still.2017.03.005

Amon B, Kryvoruchko V, Amon T, Zechmeister-Boltenstern S (2006) Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agric Ecosyst Environ 112:153–162. https://doi.org/10.1016/j.agee.2005.08.030

Andersen BJ, Samarappuli DP, Wick A, Berti MT (2020) Faba Bean and Pea can provide late-fall forage grazing without affecting maize yield the following season. Agronomy 10:80. https://doi.org/10.3390/AGRONOMY10010080

ARVALIS - Institut du végétal, CETIOM, ITB (2011) Cultures intermédiaires : impacts et conduite, Editions A. Flers

Bacenetti J, Sala C, Fusi A, Fiala M (2016) Agricultural anaerobic digestion plants: What LCA studies pointed out and what can be done to make them more environmentally sustainable. Appl Energy 179: 669–686. https://doi.org/10.1016/j.apenergy.2016.07.029

Balsari P, Dinuccio E, Gioelli F (2013) A floating coverage system for digestate liquid fraction storage. Bioresour Technol 134:285–289. https://doi.org/10.1016/J.BIORTECH.2013.02.021

Baral KR, Labouriau R, Olesen JE, Petersen SO (2017) Nitrous oxide emissions and nitrogen use efficiency of manure and digestates applied to spring barley. Agric Ecosyst Environ 239:188–198. https://doi.org/10.1016/J.AGEE.2017.01.012

Bareha Y (2018) Modélisation des processus de transformation de l'azote en digestion anaérobie : application à l'optimisation de la valorisation des digestats

Bareha Y, Girault R, Jimenez J, Trémier A (2018) Characterization and prediction of organic nitrogen biodegradability during anaerobic digestion: a bioaccessibility approach. Bioresour Technol 263: 425–436. https://doi.org/10.1016/j.biortech.2018.04.085

Bareha Y, Girault R, Guezel S, Chaker J, Trémier A (2019) Modeling the fate of organic nitrogen during anaerobic digestion: development of a bioaccessibility based ADM1. Water Res 154:298–315. https://doi.org/10.1016/j.watres.2019.02.011

Bareha Y, Affès R, Buffet J, Girault R (2021) SYS-Metha : Outil de prédiction des flux d'azote et de carbone sur les filières de méthanisation et des propriétés des digestats. Portail Data INRAE V1:. 10.15454/U4S6OF

Béghin-Tanneau R (2020) Etude des conséquences de la digestion anaérobie du maïs sur ses caractéristiques amendantes : devenir des matières organiques, stockage du carbone, vie microbienne, capacité d'adsorption et stabilité structurale de l'horizon de surface de Luvisols cultiv. Université de Toulouse

Béghin-Tanneau R, Guérin F, Guiresse M, Kleiber D, Scheiner JD (2019) Carbon sequestration in soil amended with anaerobic digested matter. Soil Tillage Res 192:87–94. https://doi.org/10.1016/j.still.2019.04.024

Bergtold JS, Ramsey S, Maddy L, Williams JR (2017) A review of economic considerations for cover crops as a conservation practice. Renew Agric Food Syst 34:62–76. https://doi.org/10.1017/S1742170517000278

Berti M, Johnson B, Ripplinger D, Gesch R, Aponte A (2017) Environmental impact assessment of double- and relay-cropping with winter camelina in the northern Great Plains, USA. Agric Syst 156:1–12. https://doi.org/10.1016/j.agsy.2017.05.012

Bertrand I, Viaud V, Daufresne T, Pellerin S, Recous S (2019) Stoichiometry constraints challenge the potential of agroecological practices for the soil C storage. A review. Agron Sustain Dev 39. https://doi.org/10.1007/s13593-019-0599-6

Binder JM, Karsten HD, Beegle DB, Dell CJ (2020) Manure injection and rye double cropping increase nutrient recovery and forage production. Agron J:1–10. https://doi.org/10.1002/agj2.20181

Blanco-Canqui H, Shaver TM, Lindquist JL, Shapiro CA, Elmore RW, Francis CA, Hergert GW (2015) Cover crops and ecosystem services: Insights from studies in temperate soils. Agron J 107:2449–2474. https://doi.org/10.2134/agronj15.0086

Börjesson P, Berglund M (2007) Environmental systems analysis of biogas systems-Part II: The environmental impact of replacing various reference systems. Biomass Bioenergy 31:326–344. https://doi.org/10.1016/j.biombioe.2007.01.004

Brozyna MA, Petersen SO, Chirinda N, Olesen JE (2013) Effects of grass-clover management and cover crops on nitrogen cycling and nitrous oxide emissions in a stockless organic crop rotation. Agric Ecosyst Environ 181:115–126. https://doi.org/10.1016/j.agee.2013.09.013

Carmona GI, Delserone LM, Campos JND et al (2021) Does cover crop management affect arthropods in the subsequent corn and soybean crops in the United States? A systematic review. Ann Entomol Soc Am 114:151–162

Carrer D, Pique G, Ferlicoq M et al (2018) What is the potential of cropland albedo management in the fight against global warming? A case study based on the use of cover crops. Environ Res Lett 13. https://doi.org/10.1088/1748-9326/aab650

Chambre Régionale d'Agriculture Nouvelle-Aquitaine (2020) MERCI, Méthode d'Estimation des Restitutions par les Cultures Intermédiaires. www.methode-MERCI.fr. Accessed 12 Aug 2021

Chantigny MH, Rochette P, Angers DA, Massé D, Côté D (2004) Ammonia volatilization and selected soil characteristics following application of anaerobically digested pig slurry. Soil Sci Soc Am J 68:306–312. https://doi.org/10.2136/sssaj2004.3060

Chantigny MH, Angers DA, Rochette P, Bélanger G, Massé D, Côté D (2007) Gaseous nitrogen emissions and forage nitrogen uptake on soils fertilized with raw and treated swine manure. J Environ Qual 36:1864–1872. https://doi.org/10.2134/jeq2007.0083

Chantigny MH, Angers DA, Bélanger G, Rochette P, Eriksen-Hamel N, Bittman S, Buckley K, Massé D, Gasser MO (2008) Yield and nutrient export of grain corn fertilized with raw and treated liquid swine manure. Agron J 100:1303–1309. https://doi.org/10.2134/agronj2007.0361

Chantigny MH, MacDonald JD, Beaupré C et al (2009) Ammonia volatilization following surface application of raw and treated liquid swine manure. Nutr Cycl Agroecosyst 85:275–286. https://doi.org/10.1007/s10705-009-9266-7

Chantigny MH, Rochette P, Angers DA, Bittman S, Buckley K, Massé D, Bélanger G, Eriksen-Hamel N, Gasser MO (2010) Soil nitrous oxide emissions following band-incorporation of fertilizer nitrogen and swine manure. J Environ Qual 39:1545–1553. https://doi.org/10.2134/jeq2009.0482

Chen G, Weil RR (2010) Penetration of cover crop roots through compacted soils. Plant Soil 331:31–43. https://doi.org/10.1007/s11104-009-0223-7

Chen R, Blagodatskaya E, Senbayram M, Blagodatsky S, Myachina O, Dittert K, Kuzyakov Y (2012) Decomposition of biogas residues in soil and their effects on microbial growth kinetics and enzyme activities. Biomass Bioenergy 45:221–229. https://doi.org/10.1016/j.biombioe.2012.06.014

Chenu C, Angers DA, Barré P, Derrien D, Arrouays D, Balesdent J (2019) Increasing organic stocks in agricultural soils: Knowledge gaps and potential innovations. Soil Tillage Res 188:41–52. https://doi.org/10.1016/j.still.2018.04.011

Clemens J, Trimborn M, Weiland P, Amon B (2006) Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. Agric Ecosyst Environ 112:171–177. https://doi.org/10.1016/j.agee.2005.08.016

Clements LJ, Salter AM, Banks CJ, Poppy GM (2012) The usability of digestate in organic farming. Water Sci Technol 66:1864–1870. https://doi.org/10.2166/wst.2012.389

Coban H, Miltner A, Elling FJ, Hinrichs KU, Kästner M (2015) The contribution of biogas residues to soil organic matter formation and CO2 emissions in an arable soil. Soil Biol Biochem 86:108–115. https://doi.org/10.1016/j.soilbio.2015.03.023

Coelho JJ, Hennessy A, Casey I, Bragança CRS, Woodcock T, Kennedy N (2020) Biofertilisation with anaerobic digestates: a field study of effects on soil microbial abundance and diversity. Appl Soil Ecol 147:103403. https://doi.org/10.1016/j.apsoil.2019.103403

Constantin J, Mary B, Laurent F, Aubrion G, Fontaine A, Kerveillant P, Beaudoin N (2010) Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments. Agric Ecosyst Environ 135:268–278. https://doi.org/10.1016/j.agee.2009.10.005

Constantin J, Beaudoin N, Laurent F, Cohan JP, Duyme F, Mary B (2011) Cumulative effects of catch crops on nitrogen uptake, leaching and net mineralization. Plant Soil 341:137–154. https://doi.org/10.1007/s11104-010-0630-9

Constantin J, Beaudoin N, Launay M, Duval J, Mary B (2012) Long-term nitrogen dynamics in various catch crop scenarios: Test and simulations with STICS model in a temperate climate. Agric Ecosyst Environ 147:36–46. https://doi.org/10.1016/j.agee.2011.06.006

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

Corré WJ, Conijn JG (2016) Biogas production and digestate utilisation from agricultural residues. Deliverable n°6.2.1. https://research.wur.nl/en/publications/biogas-production-and-digestate-utilisation-from-agricultural-res. Accessed 27 Apr 2021

Couëdel A, Alletto L, Justes E (2018a) Crucifer-legume cover crop mixtures provide effective sulphate catch crop and sulphur green manure services. Plant Soil 426:61–76. https://doi.org/10.1007/s11104-018-3615-8

Couëdel A, Alletto L, Tribouillois H, Justes É (2018b) Cover crop crucifer-legume mixtures provide effective nitrate catch crop and nitrogen green manure ecosystem services. Agric Ecosyst Environ 254:50–59. https://doi.org/10.1016/j.agee.2017.11.017

Couëdel A, Kirkegaard J, Alletto L, Justes É (2019) Crucifer-legume cover crop mixtures for biocontrol: Toward a new multi-service paradigm. Adv Agron 157:55–139. https://doi.org/10.1016/bs.agron.2019.05.003

Dale BE, Sibilla F, Fabbri C, Pezzaglia M, Pecorino B, Veggia E, Baronchelli A, Gattoni P, Bozzetto S (2016) Biogasdoneright™: an innovative new system is commercialized in Italy. Biofuels, Bioprod Biorefining 10:341–345. https://doi.org/10.1002/bbb.1671

Daryanto S, Fu B, Wang L, Jacinthe PA, Zhao W (2018) Quantitative synthesis on the ecosystem services of cover crops. Earth-Sci Rev 185:357–373. https://doi.org/10.1016/j.earscirev.2018.06.013

Du X, Jian J, Du C, Stewart RD (2022) Conservation management decreases surface runoff and soil erosion. Int Soil Water Conserv Res 10:188–196. https://doi.org/10.1016/J.ISWCR.2021.08.001

Duttmann R, Schwanebeck M, Nolde M, Horn R (2014) Predicting soil compaction risks related to field traffic during silage maize harvest. Soil Sci Soc Am J 78:408–421. https://doi.org/10.2136/SSSAJ2013.05.0198

Elfstrand S, Båth B, Mårtensson A (2007) Influence of various forms of green manure amendment on soil microbial community composition, enzyme activity and nutrient levels in leek. Appl Soil Ecol 36:70–82. https://doi.org/10.1016/j.apsoil.2006.11.001

Ellis KE, Barbercheck ME (2015) Management of overwintering cover crops influences floral resources and visitation by native bees. Environ Entomol 44:999–1010. https://doi.org/10.1093/ee/nvv086

Ernst G, Müller A, Göhler H, Emmerling C (2008) C and N turnover of fermented residues from biogas plants in soil in the presence of three different earthworm species (Lumbricus terrestris, Aporrectodea longa, Aporrectodea caliginosa). Soil Biol Biochem 40:1413–1420. https://doi.org/10.1016/j.soilbio.2007.12.026

Esnouf A, Brockmann D, Cresson R (2021) Analyse du Cycle de Vie du biométhane issu de ressources agricoles - Rapport d'ACV

EurObserv'ER (2020) Biogas Barometer – A study carried out by EurObserv'ER. https://www.eurobserv-er.org/biogas-barometer-2020/. Accessed 8 Feb 2021

Finney DM, Buyer JS, Kaye JP (2017) Living cover crops have immediate impacts on soil microbial community structure and function. J Soil Water Conserv 72:361–373. https://doi.org/10.2489/jswc.72.4.361

Florence AM, McGuire AM (2020) Do diverse cover crop mixtures perform better than monocultures? A systematic review. Agron J 112:3513–3534. https://doi.org/10.1002/agj2.20340

Frøseth RB, Bakken AK, Bleken MA, Riley H, Pommeresche R, Thorup-Kristensen K, Hansen S (2014) Effects of green manure herbage management and its digestate from biogas production on barley yield, N recovery, soil structure and earthworm populations. Eur J Agron 52:90–102. https://doi.org/10.1016/j.eja.2013.10.006

Fuchs JG, Berner A, Mayer J, et al (2008) Influence of compost and digestates on plant growth and health : potentials and limits. In: Proceedings of the CODIS 2008, Solothurn, Switzerland, 27-29 February 2008, pp 101–110

García-González I, Quemada M, Gabriel JL, Alonso-Ayuso M, Hontoria C (2018) Legacy of eight-year cover cropping on mycorrhizae, soil, and plants. J Plant Nutr Soil Sci 181:818–826. https://doi.org/10.1002/jpln.201700591

Girault R, Affes R, Akkal N et al (2019) Projet MethaPolSol : Quelles stratégies territoriales de METHAnisation pour combiner lutte contre le changement climatique, lutte contre les POLlutions diffuses azotées et amélioration de la qualité des SOLs. Rapport

Giuliano S, Alletto L, Deswarte C, Perdrieux F, Daydé J, Debaeke P (2021) Reducing herbicide use and leaching in agronomically performant maize-based cropping systems: An 8-year study. Sci Total Environ 788:147695. https://doi.org/10.1016/j.scitotenv.2021.147695

Goberna M, Podmirseg SM, Waldhuber S, Knapp BA, García C, Insam H (2011) Pathogenic bacteria and mineral N in soils following the land spreading of biogas digestates and fresh manure. Appl Soil Ecol 49:18–25. https://doi.org/10.1016/j.apsoil.2011.07.007

Gómez-Brandón M, Juárez MFD, Zangerle M, Insam H (2016) Effects of digestate on soil chemical and microbiological properties: A comparative study with compost and vermicompost. J Hazard Mater 302:267–274. https://doi.org/10.1016/j.jhazmat.2015.09.067

Graß R, Heuser F, Stülpnagel R, Piepho HP, Wachendorf M (2013) Energy crop production in double-cropping systems: Results from an experiment at seven sites. Eur J Agron 51:120–129. https://doi.org/10.1016/j.eja.2013.08.004

Guenet B, Gabrielle B, Chenu C et al (2020) Can N 2 O emissions offset the benefits from soil organic carbon storage? Glob Chang Biol 00:1–20. https://doi.org/10.1111/gcb.15342

Guilayn F (2018) First fertilizing-value typology of digestates. Associated R code and database. Suez data omitted. https://doi.org/10.5281/ZENODO.1248996

Gunnarsson A, Lindén B, Gertsson U (2011) Biodigestion of plant material can improve nitrogen use efficiency in a red beet crop sequence. HortScience 46:765–775. 10.21273/HORTSCI.46.5.765

Hansen V, Eriksen J, Jensen LS, et al (2021) Towards integrated cover crop management: N, P and S release from aboveground and belowground residues. Agric Ecosyst Environ 313. https://doi.org/10.1016/j.agee.2021.107392

Heggenstaller AH, Anex RP, Liebman M, Sundberg DN, Gibson LR (2008) Productivity and nutrient dynamics in bioenergy double-cropping systems. Agron J 100:1740–1748. https://doi.org/10.2134/agronj2008.0087

Hénault C, Grossel A, Mary B et al (2012) Nitrous oxide emission by agricultural soils: a review of spatial and temporal variability for mitigation. Pedosphere 22:426–433. https://doi.org/10.1016/S1002-0160(12)60029-0

Herrmann A, Kage H, Taube F, Sieling K (2017) Effect of biogas digestate, animal manure and mineral fertilizer application on nitrogen flows in biogas feedstock production. Eur J Agron 91:63–73. https://doi.org/10.1016/j.eja.2017.09.011

Hijazi O, Munro S, Zerhusen B, Effenberger M (2016) Review of life cycle assessment for biogas production in Europe. Renew Sustain Energy Rev 54:1291–1300. https://doi.org/10.1016/J.RSER.2015.10.013

Holly MA, Larson RA, Powell JM, Ruark MD, Aguirre-Villegas H (2017) Greenhouse gas and ammonia emissions from digested and separated dairy manure during storage and after land application. Agric Ecosyst Environ 239:410–419. https://doi.org/10.1016/J.AGEE.2017.02.007

Igos E, Golkowska K, Koster D, Vervisch B, Benetto E (2016) Using rye as cover crop for bioenergy production: an environmental and economic assessment. Biomass Bioenergy 95:116–123. https://doi.org/10.1016/j.biombioe.2016.09.023

Jensen LS, Salo T, Palmason F, Breland TA, Henriksen TM, Stenberg B, Pedersen A, Lundström C, Esala M (2005) Influence of biochemical quality on C and N mineralisation from a broad variety of plant materials in soil. Plant Soil 273:307–326. https://doi.org/10.1007/s11104-004-8128-y

Jian J, Du X, Reiter MS, Stewart RD (2020) A meta-analysis of global cropland soil carbon changes due to cover cropping. Soil Biol Biochem 143:107735. https://doi.org/10.1016/j.soilbio.2020.107735

Johansen A, Pommeresche R, Riley H, Loes A-K (2015) Anaerobic digestion of animal manure - implications for crop yields and soil biota in organic farming. In: “Nordic View to Sustainable Rural Development”, Proceedings of the 25th NJF Congress, Riga, Latvia, 16-18 June 2015, pp 97–102

Justes E, Mary B, Nicolardot B (2009) Quantifying and modelling C and N mineralization kinetics of catch crop residues in soil: Parameterization of the residue decomposition module of STICS model for mature and non mature residues. Plant Soil 325:171–185. https://doi.org/10.1007/s11104-009-9966-4

Justes E, Beaudoin N, Bertuzzi P et al (2012) Réduire les fuites de nitrate au moyen de cultures intermédiaires: conséquences sur les bilans d’eau et d’azote, autres services écosystémiques. Rapport d’étude, INRA (France). https://www6.paris.inrae.fr/depe/content/download/3183/32222/version/2/file/. Accessed 27 Apr 2021

Kaye JP, Quemada M (2017) Using cover crops to mitigate and adapt to climate change. A review. Agron Sustain Dev 37:4. https://doi.org/10.1007/s13593-016-0410-x

Kemp L, Lyutse S (2011) Second Harvest : Bioenergy from Cover Crop Biomass. In: NRDC Issue Pap. https://www.nrdc.org/sites/default/files/covercrop_ip.pdf. Accessed 27 Apr 2021

Koblenz B, Tischer S, Rücknagel J, Christen O (2015) Influence of biogas digestate on density, biomass and community composition of earthworms. Ind Crops Prod 66:206–209. https://doi.org/10.1016/j.indcrop.2014.12.024

Kolbe H (2007) Site adjusted humus balance method for use in arable farming systems. In: Practical Solutions for Managing Optimum C and N Content in Agricultural Soils IV, Praha, June 20-22

Krueger ES, Ochsner TE, Porter PM, Baker JM (2011) Winter rye cover crop management influences on soil water, soil nitrate, and corn development. Agron J 103:316–323. https://doi.org/10.2134/agronj2010.0327

Lantz M, Börjesson P (2014) Greenhouse gas and energyassessment of the biogas from co-digestion injected into the natural gas grid: a Swedish case-study including effects on soil properties. Renew Energy 71:387–395. https://doi.org/10.1016/J.RENENE.2014.05.048

Launay C, Crépeau M, Girault R et al (2020) 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. In: Journées Recherche Innovation (JRI) 8-10 september 2020 in Toulouse. Toulouse

Launay C, Constantin J, Chlebowski F, Houot S, Graux AI, Klumpp K, Martin R, Mary B, Pellerin S, Therond O (2021) Estimating the carbon storage potential and greenhouse gas emissions of French arable cropland using high-resolution modeling. Glob Chang Biol 27:1645–1661. https://doi.org/10.1111/gcb.15512

Liebetrau J, Clemens J, Cuhls C, Hafermann C, Friehe J, Weiland P, Daniel-Gromke J (2010) Methane emissions from biogas-producing facilities within the agricultural sector. Eng Life Sci 10: 595–599. https://doi.org/10.1002/ELSC.201000070

Maier S, Szerencsits M, Shahzad K (2017) Ecological evaluation of biogas from catch crops with Sustainable Process Index (SPI). Energy. Sustain. Soc. 7

Malone RW, Obrycki JF, Karlen DL, Ma L, Kaspar TC, Jaynes DB, Parkin TB, Lence SH, Feyereisen GW, Fang QX, Richard TL, Gillette K (2018) Harvesting fertilized rye cover crop: simulated revenue, net energy, and drainage nitrogen loss. Agric Environ Lett 3:170041. https://doi.org/10.2134/ael2017.11.0041

Marcillo GS, Miguez FE (2017) Corn yield response to winter cover crops: an updated meta-analysis. J Soil Water Conserv 72:226–239. https://doi.org/10.2489/jswc.72.3.226

Maris SC, Capra F, Ardenti F, Boselli R, Pochintesta D, Beone GM, Tabaglio V, Fiorini A (2021) The interaction between types of cover crop residue and digestate application methods affects ammonia volatilization during maize cropping season. J Environ Qual 50: 504–512. https://doi.org/10.1002/jeq2.20205

Marsac S, Heredia M, Bazet M et al (2019) Optimisation de la mobilisation de CIVE pour la méthanisation dans les systèmes d'exploitation. Rapport. https://librairie.ademe.fr/dechets-economie-circulaire/3993-opticive.html. Accessed 27 Apr 2021

Matsunaka T, Sawamoto T, Ishimura H, Takakura K, Takekawa A (2006) Efficient use of digested cattle slurry from biogas plant with respect to nitrogen recycling in grassland. Int Congr Ser 1293:242–252. https://doi.org/10.1016/j.ics.2006.03.016

Matthiessen J, Kirkegaard J (2006) Biofumigation and enhanced biodegradation: opportunity and challenge in soilborne pest and disease management. CRC Crit Rev Plant Sci 25:235–265. https://doi.org/10.1080/07352680600611543

Meyer N (2020) Evaluation de l'impact des cultures intermédiaires multi-services sur le bilan hydrique des sols : approche par expérimentation et simulation. Application au bassin Adour-Garonne. INP Toulouse

Meyer N, Bergez J-E, Constantin J, Justes E (2019) Cover crops reduce water drainage in temperate climates: a meta-analysis. Agron Sustain Dev 39:3. https://doi.org/10.1007/s13593-018-0546-y

Meyer N, Bergez J-E, Constantin J, Belleville P, Justes E (2020) Cover crops reduce drainage but not always soil water content due to interactions between rainfall distribution and management. Agric Water Manag 231:105998. https://doi.org/10.1016/j.agwat.2019.105998

Meza FJ, Silva D, Vigil H (2008) Climate change impacts on irrigated maize in Mediterranean climates: Evaluation of double cropping as an emerging adaptation alternative. Agric Syst 98:21–30. https://doi.org/10.1016/j.agsy.2008.03.005

Ministère de la transition écologique et solidaire (2020) Stratégie française pour l'énergie et le climat. Programmation pluriannuelle de l'énergie. Synthèse. https://www.ecologie.gouv.fr/sites/default/files/20200422SynthèsedelaPPE.pdf. Accessed 27 Apr 2021

Molinuevo-Salces B, Larsen SU, Ahring BK, Uellendahl H (2013) Biogas production from catch crops: evaluation of biomass yield and methane potential of catch crops in organic crop rotations. Biomass Bioenergy 59:285–292. https://doi.org/10.1016/j.biombioe.2013.10.008

Molinuevo-Salces B, Larsen SU, Ahring BK, Uellendahl H (2014) Biogas production from catch crops: Increased yield by combined harvest of catch crops and straw and preservation by ensiling. Biomass Bioenergy 79:3–11. https://doi.org/10.1016/j.biombioe.2015.04.040

Möller K (2015) Effects of anaerobic digestion on soil carbon and nitrogen turnover, N emissions, and soil biological activity. A review. Agron Sustain Dev 35:1021–1041. https://doi.org/10.1007/s13593-015-0284-3

Möller K, Müller T (2012) Effects of anaerobic digestion on digestate nutrient availability and crop growth: A review. Eng Life Sci 12: 242–257. https://doi.org/10.1002/elsc.201100085

Möller K, Stinner W (2009) Effects of different manuring systems with and without biogas digestion on soil mineral nitrogen content and on gaseous nitrogen losses (ammonia, nitrous oxides). Eur J Agron 30: 1–16. https://doi.org/10.1016/j.eja.2008.06.003

Muhammad I, Wang J, Sainju UM, Zhang S, Zhao F, Khan A (2021) Cover cropping enhances soil microbial biomass and affects microbial community structure: a meta-analysis. Geoderma 381:114696. https://doi.org/10.1016/j.geoderma.2020.114696

Negri M, Bacenetti J, Brambilla M, Manfredini A, Cantore A, Bocchi S (2014) Biomethane production from different crop systems of cereals in Northern Italy. Biomass Bioenergy 63:321–329. https://doi.org/10.1016/j.biombioe.2014.01.041

Ni K, Pacholski A, Gericke D, Kage H (2012) Analysis of ammonia losses after field application of biogas slurries by an empirical model. J Plant Nutr Soil Sci 175:253–264. https://doi.org/10.1002/jpln.201000358

Nicholson FA, Bhogal A, Cardenas L, Chadwick D, Misselbrook T, Rollett A, Taylor M, Thorman R, Williams J (2017) Nitrogen losses to the environment following food-based digestate and compost applications to agricultural land. Environ Pollut 228:504–516. https://doi.org/10.1016/j.envpol.2017.05.023

Nielsen DC, Lyon DJ, Hergert GW, Higgins RK, Calderón FJ, Vigil MF (2015) Cover crop mixtures do not use water differently than single-species plantings. Agron J 107:1025–1038. https://doi.org/10.2134/agronj14.0504

Nkoa R (2014) Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. Agron Sustain Dev 34: 473–492. https://doi.org/10.1007/s13593-013-0196-z

Nyord T, Hansen MN, Birkmose TS (2012) Ammonia volatilisation and crop yield following land application of solid-liquid separated, anaerobically digested, and soil injected animal slurry to winter wheat. Agric Ecosyst Environ 160:75–81. https://doi.org/10.1016/j.agee.2012.01.002

Obiang Ndong G, Villerd J, Cousin I, Therond O (2021) Using a multivariate regression tree to analyze trade-offs between ecosystem services: application to the main cropping area in France. Sci Total Environ 764:142815. https://doi.org/10.1016/j.scitotenv.2020.142815

Pan B, Lam SK, Mosier A, Luo Y, Chen D (2016) Ammonia volatilization from synthetic fertilizers and its mitigation strategies: a global synthesis. Agric Ecosyst Environ 232:283–289. https://doi.org/10.1016/j.agee.2016.08.019

Pelster DE, Chantigny MH, Rochette P, Angers DA, Rieux C, Vanasse A (2012) Nitrous oxide emissions respond differently to mineral and organic nitrogen sources in contrasting soil types. J Environ Qual 41:427–435. https://doi.org/10.2134/jeq2011.0261

Peters D, Zabeti M, Kühner A-K et al (2016) Assessing the case for sequential cropping to produce low ILUC risk biomethane: final report. https://research.wur.nl/en/publications/assessing-the-case-for-sequential-cropping-to-produce-low-iluc-ri. Accessed 27 Apr 2021

Petersen SO (1999) Nitrous Oxide emissions from manure and inorganic fertilizers applied to spring barley. J Environ Qual 28:1610–1618. https://doi.org/10.2134/jeq1999.00472425002800050027x

Peu P, Picard S, Diara A, Girault R, Béline F, Bridoux G, Dabert P (2012) Prediction of hydrogen sulphide production during anaerobic

digestion of organic substrates. Bioresour Technol 121:419–424. https://doi.org/10.1016/j.biortech.2012.06.112
Plöchl M, Hilse A, Heiermann M, et al (2009) Application of hydrolytic enzymes for improving biogas feedstock fluidity. CIGR Ejournal IX:1–16
Poeplau C, Don A (2015) Carbon sequestration in agricultural soils via cultivation of cover crops - a meta-analysis. Agric Ecosyst Environ 200:33–41. https://doi.org/10.1016/j.agee.2014.10.024
Qi Z, Helmers MJ (2010) Soil water dynamics under winter rye cover crop in central Iowa. Vadose Zo J 9:53. https://doi.org/10.2136/vzj2008.0163
Quakernack R, Pacholski A, Techow A, Herrmann A, Taube F, Kage H (2012) Ammonia volatilization and yield response of energy crops after fertilization with biogas residues in a coastal marsh of Northern Germany. Agric Ecosyst Environ 160:66–74. https://doi.org/10.1016/j.agee.2011.05.030
Quennesson A, Decaux S (2020) Filabiom : produire de la biomasse et du méthane avec la double culture dédiée. http://www.agro-transfert-rt.org/filabiom/outils/production/double-culture/. Accessed 27 Apr 2021
Renaud M, Chelinho S, Alvarenga P, Mourinha C, Palma P, Sousa JP, Natal-da-Luz T (2017) Organic wastes as soil amendments – effects assessment towards soil invertebrates. J Hazard Mater 330:149–156. https://doi.org/10.1016/j.jhazmat.2017.01.052
Riau V, Burgos L, Camps F, Domingo F, Torrellas M, Antón A, Bonmatí A (2021) Closing nutrient loops in a maize rotation. Catch crops to reduce nutrient leaching and increase biogas production by anaerobic co-digestion with dairy manure. Waste Manag 126:719–727. https://doi.org/10.1016/j.wasman.2021.04.006
Risberg K, Cederlund H, Pell M, Arthurson V, Schnürer A (2017) Comparative characterization of digestate versus pig slurry and cow manure – chemical composition and effects on soil microbial activity. Waste Manag 61:529–538. https://doi.org/10.1016/j.wasman.2016.12.016
Rodhe LKK, Ascue J, Willén A, Persson BV, Nordberg Å (2015) Greenhouse gas emissions from storage and field application of anaerobically digested and non-digested cattle slurry. Agric Ecosyst Environ 199:358–368. https://doi.org/10.1016/j.agee.2014.10.004
Rollett AJ, Bhogal A, Scullion J et al (2020) The effect of field application of food-based anaerobic digestate on earthworm populations. Soil Use Manag 00:1–10. https://doi.org/10.1111/sum.12615
Sadet-Bourgeteau S, Houot S, Dequiedt S, Nowak V, Tardy V, Terrat S, Montenach D, Mercier V, Karimi B, Chemidlin Prévost-Bouré N, Maron PA (2018) Lasting effect of repeated application of organic waste products on microbial communities in arable soils. Appl Soil Ecol 125:278–287. https://doi.org/10.1016/j.apsoil.2018.02.006
Sandler L, Nelson KA, Dudenhoeffer C (2015) Winter wheat row spacing and alternative crop effects on relay-intercrop, double-crop, and wheat yields. Int J Agron 2015:1–8. https://doi.org/10.1155/2015/369243
Sarker TC, Incerti G, Spaccini R, Piccolo A, Mazzoleni S, Bonanomi G (2018) Linking organic matter chemistry with soil aggregate stability: Insight from 13C NMR spectroscopy. Soil Biol Biochem 117:175–184. https://doi.org/10.1016/j.soilbio.2017.11.011
Schipanski ME, Barbercheck M, Douglas MR, Finney DM, Haider K, Kaye JP, Kemanian AR, Mortensen DA, Ryan MR, Tooker J, White C (2014) A framework for evaluating ecosystem services provided by cover crops in agroecosystems. Agric Syst 125:12–22. https://doi.org/10.1016/j.agsy.2013.11.004
Schröder JJ, Uenk D, Hilhorst GJ (2007) Long-term nitrogen fertilizer replacement value of cattle manures applied to cut grassland. Plant Soil 299:83–99. https://doi.org/10.1007/s11104-007-9365-7
Senbayram M, Chen R, Wienforth B et al (2014) Emission of N2O from biogas crop production systems in Northern Germany. Bioenergy Res 7:1223–1236. https://doi.org/10.1007/s12155-014-9456-2
Sieling K, Herrmann A, Wienforth B, Taube F, Ohl S, Hartung E, Kage H (2013) Biogas cropping systems: short term response of yield performance and N use efficiency to biogas residue application. Eur J Agron 47:44–54. https://doi.org/10.1016/j.eja.2013.01.002
Sigurnjak I, Vaneeckhaute C, Michels E, Ryckaert B, Ghekiere G, Tack FMG, Meers E (2017) Fertilizer performance of liquid fraction of digestate as synthetic nitrogen substitute in silage maize cultivation for three consecutive years. Sci Total Environ 599–600:1885–1894. https://doi.org/10.1016/j.scitotenv.2017.05.120
Sinsabaugh RL, Turner BL, Talbot JM, Waring BG, Powers JS, Kuske CR, Moorhead DL, Follstad Shah JJ (2016) Stoichiometry of microbial carbon use efficiency in soils. Ecol Monogr 86:172–189. https://doi.org/10.1890/15-2110.1
Sizmur T, Martin E, Wagner K, Parmentier E, Watts C, Whitmore AP (2017) Milled cereal straw accelerates earthworm (Lumbricus terrestris) growth more than selected organic amendments. Appl Soil Ecol 113:166–177. https://doi.org/10.1016/j.apsoil.2016.12.006
Smith RG, Warren ND, Cordeau S (2020) Are cover crop mixtures better at suppressing weeds than cover crop monocultures? Weed Sci 68:186–194. https://doi.org/10.1017/wsc.2020.12
Snapp SS, Swinton SM, Labarta R, Mutch D, Black JR, Leep R, Nyiraneza J, O'Neil K (2005) Evaluating cover crops for benefits, costs and performance within cropping system niches. Agron J 97:322–332. https://doi.org/10.2134/agronj2005.0322
Sommer SG (1997) Ammonia volatilization from farm tanks containing anaerobically digested animal slurry. Atmos Environ 31:863–868. https://doi.org/10.1016/S1352-2310(96)00250-6
Sommer SG, Jensen LS, Clausen SB, Søgaard HT (2006) Ammonia volatilization from surface-applied livestock slurry as affected by slurry composition and slurry infiltration depth. J Agric Sci 144:229–235. https://doi.org/10.1017/S0021859606006022
Soti PG, Rugg S, Racelis A (2016) Potential of cover crops in promoting mycorrhizal diversity and soil quality in organic farms. J Agric Sci 8:42. https://doi.org/10.5539/jas.v8n8p42
Stinner PW (2015) The use of legumes as a biogas substrate - potentials for saving energy and reducing greenhouse gas emissions through symbiotic nitrogen fixation. Energy Sustain Soc 5:1–7. https://doi.org/10.1186/s13705-015-0034-z
Stumpe B, Werner S, Jung R, Heinze S, Jüschke E, Strippel C, Marschner B (2012) Organic carbon dynamics and enzyme activities in agricultural soils amended with biogas slurry, liquid manure and sewage sludge. Agric Sci 03:104–113. https://doi.org/10.4236/as.2012.31014
Styles D, Gibbons J, Williams AP, Dauber J, Stichnothe H, Urban B, Chadwick DR, Jones DL (2015) Consequential life cycle assessment of biogas, biofuel and biomass energy options within an arable crop rotation. GCB Bioenergy 7:1305–1320. https://doi.org/10.1111/gcbb.12246
Svoboda N, Taube F, Wienforth B, Kluß C, Kage H, Herrmann A (2013) Nitrogen leaching losses after biogas residue application to maize. Soil Tillage Res 130:69–80. https://doi.org/10.1016/j.still.2013.02.006
Szerencsits M (2014) Synergetische Biogaserzeugung aus Zwischenfrüchten und nachhaltigen Fruchtfolgesystemen. https://www.energieforschung.at/projekte/632/syn-energy-ii-synergetische-biogaserzeugung-aus-zwischenfruechten-und-nachhaltigen-fruchtfolgesystemen. Accessed 27 Apr 2021
Teixeira Franco R, Buffière P, Bayard R (2016) Ensiling for biogas production: critical parameters. A review. Biomass Bioenergy 94:94–104
Terhoeven-Urselmans T, Scheller E, Raubuch M, Ludwig B, Joergensen RG (2009) CO2 evolution and N mineralization after biogas slurry application in the field and its yield effects on spring barley. Appl Soil Ecol 42:297–302. https://doi.org/10.1016/j.apsoil.2009.05.012

Thomsen IK, Pedersen AR, Nyord T, Petersen SO (2010) Effects of slurry pre-treatment and application technique on short-term N2O emissions as determined by a new non-linear approach. Agric Ecosyst Environ 136:227–235. https://doi.org/10.1016/j.agee.2009.12.001

Thomsen IK, Olesen JE, Møller HB, Sørensen P, Christensen BT (2013) Carbon dynamics and retention in soil after anaerobic digestion of dairy cattle feed and faeces. Soil Biol Biochem 58:82–87. https://doi.org/10.1016/j.soilbio.2012.11.006

Thorup-Kristensen K, Magid J, Jensen LS (2003) Catch crops and green manures as biological tools in nitrogen management in temperate zones. Adv Agron 79:227–302. https://doi.org/10.1016/S0065-2113(02)79005-6

Thrän D, Schaubach K, Majer S, Horschig T (2020) Governance of sustainability in the German biogas sector—adaptive management of the Renewable Energy Act between agriculture and the energy sector. Energy Sustain Soc 101 10:3. https://doi.org/10.1186/S13705-019-0227-Y

Tigini V, Franchino M, Bona F, Varese GC (2016) Is digestate safe? A study on its ecotoxicity and environmental risk on a pig manure. Sci Total Environ 551–552:127–132. https://doi.org/10.1016/j.scitotenv.2016.02.004

Tonitto C, David MB, Drinkwater LE (2006) Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: a meta-analysis of crop yield and N dynamics. Agric Ecosyst Environ 112:58–72. https://doi.org/10.1016/j.agee.2005.07.003

Tosti G, Benincasa P, Farneselli M, Tei F, Guiducci M (2014) Barley-hairy vetch mixture as cover crop for green manuring and the mitigation of N leaching risk. Eur J Agron 54:34–39. https://doi.org/10.1016/j.eja.2013.11.012

Tribouillois H, Fort F, Cruz P, Charles R, Flores O, Garnier E, Justes E (2015) A functional characterisation of a wide range of cover crop species: Growth and nitrogen acquisition rates, leaf traits and ecological strategies. PLoS ONE 10:1–17. https://doi.org/10.1371/journal.pone.0122156

Tribouillois H, Cohan J-P, Justes E (2016) Cover crop mixtures including legume produce ecosystem services of nitrate capture and green manuring: assessment combining experimentation and modelling. Plant Soil 401:347–364. https://doi.org/10.1007/s11104-015-2734-8

Tribouillois H, Constantin J, Justes E (2018) Cover crops mitigate direct greenhouse gases balance but reduce drainage under climate change scenarios in temperate climate with dry summers. Glob Chang Biol 24:2513–2529. https://doi.org/10.1111/gcb.14091

Tsachidou B, Scheuren M, Gennen J, Debbaut V, Toussaint B, Hissler C, George I, Delfosse P (2019) Biogas residues in substitution for chemical fertilizers: a comparative study on a grassland in the Walloon Region. Sci Total Environ 666:212–225. https://doi.org/10.1016/j.scitotenv.2019.02.238

Valkama E, Lemola R, Känkänen H, Turtola E (2015) Meta-analysis of the effects of undersown catch crops on nitrogen leaching loss and grain yields in the Nordic countries. Agric Ecosyst Environ 203:93–101. https://doi.org/10.1016/j.agee.2015.01.023

Vallejo A, Skiba UM, García-Torres L et al (2006) Nitrogen oxides emission from soils bearing a potato crop as influenced by fertilization with treated pig slurries and composts. Soil Biol Biochem 38:2782–2793. https://doi.org/10.1016/J.SOILBIO.2006.04.040

Viard A, Hénault C, Rochette P, Kuikman P, Flénet F, Cellier P (2013) Le protoxyde d'azote (N2O), puissant gaz à effet de serre émis par les sols agricoles : méthodes d'inventaire et leviers de réduction. Oléagineux, Corps gras, Lipides 20:108–118. https://doi.org/10.1051/ocl.2013.0501

Walling E, Vaneeckhaute C (2020) Greenhouse gas emissions from inorganic and organic fertilizer production and use: a review of emission factors and their variability. J Environ Manage 276:111211

Walsh JJ, Jones DL, Edwards-Jones G, Williams AP (2012a) Replacing inorganic fertilizer with anaerobic digestate may maintain agricultural productivity at less environmental cost. J Plant Nutr Soil Sci 175:840–845. https://doi.org/10.1002/jpln.201200214

Walsh JJ, Rousk J, Edwards-Jones G, Jones DL, Williams AP (2012b) Fungal and bacterial growth following the application of slurry and anaerobic digestate of livestock manure to temperate pasture soils. Biol Fertil Soils 48:889–897. https://doi.org/10.1007/s00374-012-0681-6

Wannasek L, Ortner M, Kaul HP et al (2019) Double-cropping systems based on rye, maize and sorghum: impact of variety and harvesting time on biomass and biogas yield. Eur J Agron 110. https://doi.org/10.1016/j.eja.2019.125934

Webb J, Pain B, Bittman S, Morgan J (2010) The impacts of manure application methods on emissions of ammonia, nitrous oxide and on crop response—a review. Agric Ecosyst Environ 137:39–46. https://doi.org/10.1016/J.AGEE.2010.01.001

Wentzel S, Joergensen RG (2016) Effects of biogas and raw slurries on grass growth and soil microbial indices. J Plant Nutr Soil Sci 179:215–222. https://doi.org/10.1002/jpln.201400544

Wentzel S, Schmidt R, Piepho HP, Semmler-Busch U, Joergensen RG (2015) Response of soil fertility indices to long-term application of biogas and raw slurry under organic farming. Appl Soil Ecol 96:99–107. https://doi.org/10.1016/j.apsoil.2015.06.015

White CM, DuPont ST, Hautau M et al (2017) Managing the trade off between nitrogen supply and retention with cover crop mixtures. Agric Ecosyst Environ 237:121–133. https://doi.org/10.1016/j.agee.2016.12.016

Wilcoxen CA, Walk JW, Ward MP (2018) Use of cover crop fields by migratory and resident birds. Agric Ecosyst Environ 252:42–50. https://doi.org/10.1016/j.agee.2017.09.039

Wolf U, Fuß R, Höppner F, Flessa H (2014) Contribution of N2O and NH3 to total greenhouse gas emission from fertilization: results from a sandy soil fertilized with nitrate and biogas digestate with and without nitrification inhibitor. Nutr Cycl Agroecosystems 100:121–134. https://doi.org/10.1007/s10705-014-9631-z

Wulf S, Maeting M, Clemens J (2002) Application technique and slurry co-fermentation effects on ammonia, nitrous oxide, and methane emissions after spreading. J Environ Qual 31:1795–1801. https://doi.org/10.2134/jeq2002.1795

WWF France (2020) Méthanisation agricole, quelles conditions de la durabilité de la filière en France ? https://www.wwf.fr/sites/default/files/doc-2020-03/20200317_Rapport_Methanisation-agricole_WWF_GRDF-min.pdf. Accessed 27 Apr 2021

Yang G, Zhang G, Zhuan R, Yang A, Wang Y (2016) Transformations, Inhibition and inhibition control methods of sulfur in sludge anaerobic digestion: a review. Curr Org Chem 20:2780–2789. https://doi.org/10.2174/1385272820666160513152913

Yu Y, Loiskandl W, Kaul HP, Himmelbauer M, Wei W, Chen L, Bodner G (2016) Estimation of runoff mitigation by morphologically different cover crop root systems. J Hydrol 538:667–676. https://doi.org/10.1016/j.jhydrol.2016.04.060

Zilio M, Pigoli A, Rizzi B, Geromel G, Meers E, Schoumans O, Giordano A, Adani F (2021) Measuring ammonia and odours emissions during full field digestate use in agriculture. Sci Total Environ 782:146882. https://doi.org/10.1016/j.scitotenv.2021.146882

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