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Climate impact of alternative organic fertilizers using life cycle assessment

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Abstract

Anaerobic digestion is a common method for managing liquid manure and other biomasses, generating biogas as a renewable energy source. The resulting digestate can be processed into organic fertilizers to enhance nutrient recycling, but its environmental impact warrants investigation. In this study, a life cycle assessment was conducted to examine the impact of fertilizers derived from cattle slurry and grass–clover co-digestion on global warming (measured in CO_2 equivalents) compared to untreated cattle slurry (CA). The different treatments analyzed include CA, digestate, liquid fractions (LFs) from digestate separation, and an enriched liquid nitrogen–sulfur product derived from post-processing of biogas and drying of the solid fraction. The functional units of this study were 100 kg of total nitrogen in the final organic fertilizer (FU1) with the cradle-to-processing gate boundary, and the harvesting of 1 ton of spring barley dry matter (FU2) with the cradle-to-field application boundary. The carbon footprint ranged from 24% to 49% of the baseline scenario for FU1, and from −6% to 177% of the baseline scenario for FU2. The main contributors to the carbon footprint of fertilizers included greenhouse gas emissions from storage and field application. However, biogas production from anaerobic digestion, together with the concurrent mitigation of CH_4 emissions during storage, contributed most to a reduction in the overall global warming potential associated with anaerobic digestate and its LF. This study showed large climate prospects in replacing untreated slurry as organic fertilizer with alternatives resulting from its anaerobic digestion and post-treatment.

1. Introduction

Livestock production in the European Union generated a total of 1400 million tons of liquid manure (slurry) annually during 2016–2019 (Köninger *et al* 2021). Liquid manure is rich in nutrients such as nitrogen, phosphorus, and potassium and can be used as an organic fertilizer, replacing industrially produced mineral fertilizers for crops (Zubair *et al* 2020). To achieve a more sustainable agricultural production, it is necessary to increase nutrient use efficiency and reduce environmental impacts of manure, including the greenhouse gas (GHG) emissions generated from its production, transportation, storage, field application and harvest.

Anaerobic digestion offers the advantage of biogas production as a renewable energy source. Biogas typically consists of 53%–70% CH_4, 30%–47% CO_2, and small amounts of impurities such as N_2, H_2, H_2S and CO (Pellegrini *et al* 2018). After removing CO_2 and other impurities, biogas can be injected into an existing gas grid. The residue from the digestion process, the digestate, is rich in ammonium and has a lower organic matter content compared to the untreated biomasses. Digestate is commonly used as a fertilizer on farms and has been reported to have less GHG emissions during storage and field application compared to untreated slurry (Holly *et al* 2017). Separation is also widely used for slurry management which, results in both liquid and solid fractions. The liquid fraction (LF) is more concentrated in ammonium (NH_4^+), whereas the solid fraction has higher organic dry matter (DM) content, organic N, and P, making it more suitable for composting or bedding, and with lower transportation costs compared to the LF (Petersen *et al* 2007).

Slurry treatment processes, including co-digestion and separation, result in fertilizers with different chemical and physical properties regarding pH, organic and mineral N content, DM content, and C/N ratio, all of which can impact GHG emissions associated with processing, storage, and field application. Studies have shown that anaerobic digestion reduces CH_4 emissions during storage by reducing degradable organic matter (Maraseni and Maroulis 2008, Maldaner *et al* 2018). However, higher pH and ammonium content in digestates can lead to increased NH_3 emissions compared to untreated slurry (Külling *et al* 2003). The LF of digestate after separation has lower organic matter content and may further reduce CH_4 emissions (Meng *et al* 2023).

Following the field application of nitrogen fertilizer materials, N_2O is the main contributor to GHG emissions. N_2O emissions depend on various factors such as climate, soil characteristics, fertilizer composition, and application method. Some studies found that digestate, compared to untreated slurry, may exhibit higher N_2O emissions possibly due to its higher ammonium content (Abubaker *et al* 2013). However, contrasting trends with reduced N_2O emissions from digestates have also been reported, attributed to the lower availability of degradable organic matter (Holly *et al* 2017). Meng *et al* (2023) reported similar levels of N_2O from digested and untreated organic fertilizers.

To assess the climate impact of manure treatment technologies, and improve manure management towards a lower carbon footprint, a comprehensive method such as life cycle assessment (LCA) is required. LCA is a widely used approach to assess the environmental impacts and resource use throughout a product's life cycle (Wolf *et al* 2010). However, previous studies using LCA to investigate the environmental impacts of manure-based fertilizers have some limitations, such as failure to consider the transport and reuse of organic fertilizers for crop production (Mezzullo *et al* 2013), the use of co-substrates from waste streams (Stucki *et al* 2011) and soil carbon sequestration after field application (Prapaspongsa *et al* 2010). Additionally, some studies only focused on specific stages of the treatment process or evaluated a single treatment rather than alternative treatment strategies (Basset-Mens and Van der Werf 2005, Van der Werf *et al* 2007). The present study compared GHG emissions from a baseline scenario using untreated cattle slurry (CA) as an organic fertilizer with two alternative scenarios based on co-digestion of cattle slurry and grass–clover, with or without additional post-processing of the digestate.

2. Materials and methods

2.1. Goal and system boundary

The aim of this study was to assess and compare the global warming potential (GWP) of both the production and use of different cattle slurry-based organic fertilizers. Using a LCA approach, the GWP in a 100 year time horizon (GWP_{100}) was determined for production, storage, and field application of the fertilizers, including upstream processes. As shown in figure 1, the system analysis is divided into three principal life cycle stages: (I) feedstock production, (II) feedstock processing, and (III) use of fertilizers for crop production. Stage (I) includes all production activities and processes related to the production of the raw materials (i.e. transport, energy use and emissions). Stage (II) comprises all activities at the biogas plant and further processing, including anaerobic digestion, separation, solids drying with ammonia stripping, and sulfur oxidation in a bio-filter, trapping ammonia in sulfuric acid produced (AmS). Stage (III) includes C and N turnover of organic N fertilizers in the field and associated emissions such as N_2O, CH_4 and NH_3.

This study examined three different scenarios of organic N fertilizers reflecting an increasing level of processing in the co-digestion of cattle slurry and grass–clover (figure 1):

- S_0: CA as the reference system, including storage in the housing unit, tank storage, field application and all related transportation.
- S_1: Anaerobic co-digestion of cattle slurry and grass–clover silage resulted in anaerobic digestate fertilizer (DF), including storage in the housing unit, anaerobic digestion, tank storage, field application of fertilizer and all associated transportation.
- S_2: This scenario includes the same process steps as S_1, alongside separation of anaerobic digestate, storage of LF, addition of NH_3 stripping from the solid fraction of digestate, sulfur stripping from the crude biogas, storage of ammonium sulfate (AmS), field application of fertilizers and all associated transportation.

The primary data for the N fertilizers investigated in this study were obtained from an experimental study (Meng *et al* 2023) and supplemented with available literature and background data from the Ecoinvent 3 and Agrifootprint 5 databases. To investigate the GHG emissions of both production and field application of organic N fertilizers, two functional units were considered. The first functional unit (FU1) was 100 kg of total nitrogen (TN) in each type of fertilizer,

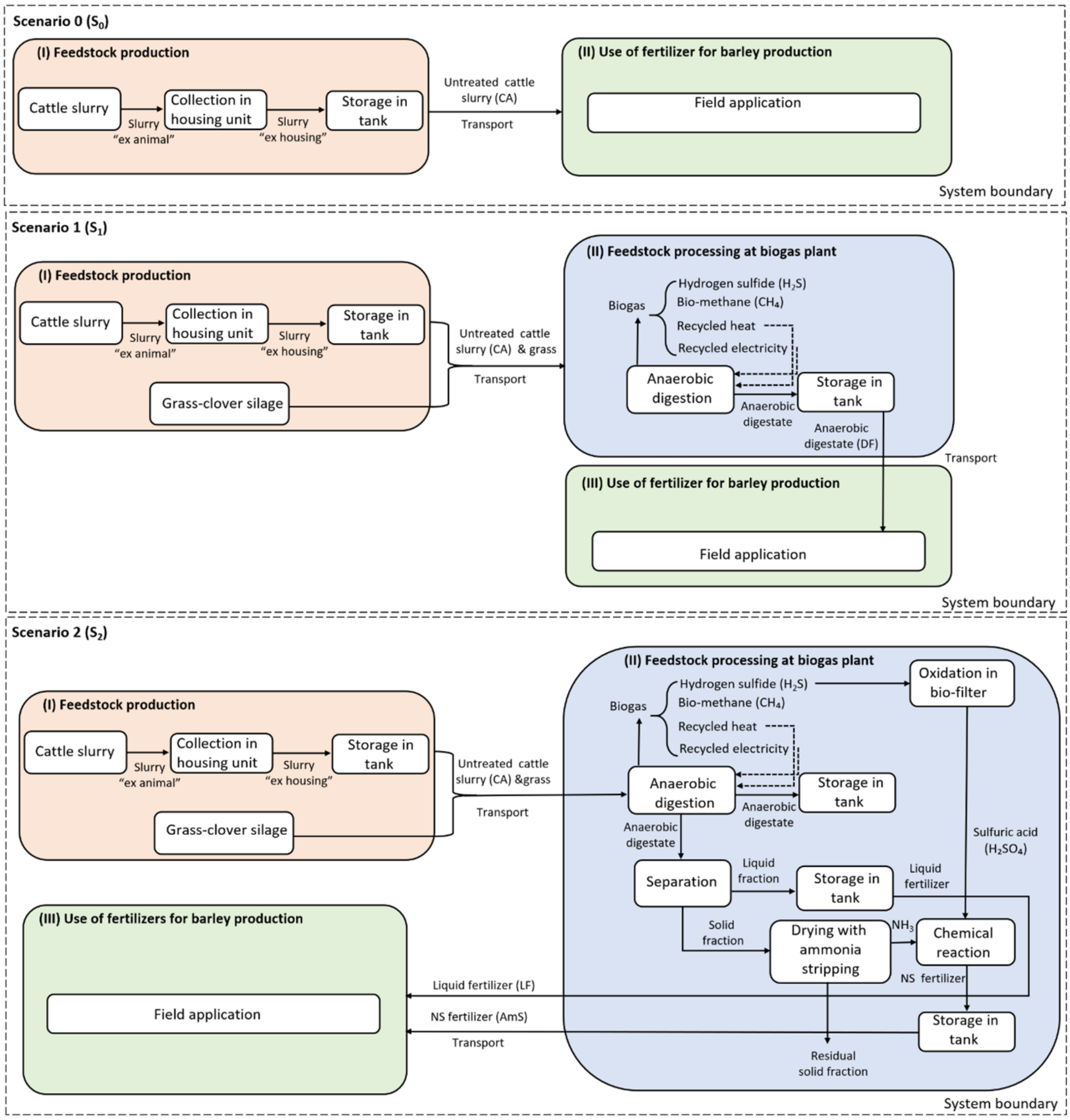

Figure 1. System boundary for production, processing and application of organic N fertilizers derived from co-digestion of cattle slurry and grass–clover for barley production in Denmark. The scenarios include: baseline (S_0) with untreated cattle slurry (CA); scenario 1 (S_1) with anaerobic co-digestion resulting in a digestate fertilizer (DF); and scenario 2 (S_2) with co-digestion followed by separation of the digestate into liquid fraction (LF) and a solid fraction, which is dried and processed to produce ammonium sulfate (AmS) fertilizer, utilizing ammonia captured in sulfuric acid solution from biogas oxidation. Black dashed lines represent the system boundary.

which was the application rate assumed for 1 ha with spring barley. In this way, the assessment results represent the environmental cost of organic fertilizer per hectare. The second functional unit (FU2) was 1 ton of spring barley grain yield. The first functional unit (FU1) included life cycle stages I and II until the processing gate (see life cycle stages in figure 1) and the second functional unit (FU2) included life cycle stages I, II and III until the field gate.

2.2. System overview and life cycle inventory for organic fertilizers

2.2.1. Feedstock production

Untreated slurry from Holstein-Friesian cattle, with selected physiochemical properties detailed in table 1, was sourced from a housing unit of the cattle research center on the Viborg campus of Aarhus University in Western Denmark in 2020. This slurry was either used for anaerobic co-digestion or stored. As agricultural waste, emissions related to the production of manure were not included in this study.

Storage emissions of N_2O, CH_4, and NH_3 from cattle slurry were estimated based on the average emissions observed in a pilot-scale storage experiment (Meng *et al* 2023), accounting for variations in storage durations both in housing units and outside. Detailed inventory data related to the storage of cattle slurry can be found in tables A2 and A3 (appendix A). The emission factors relevant for calculating GHG emissions in this study are presented in table 2. Table A4 (appendix A) presents the data used for grass–clover production.

Table 1. Physiochemical characteristics of organic fertilizers used for storage and field application.

	Fertilizer	DM (kg ton^{-1})	VS[e] (kg ton^{-1})	TN[f] (kg ton^{-1})	TC[g] (kg ton^{-1})	$NH_4{}^+$–N (kg ton^{-1})	pH	$NH_4{}^+$–N TN^{-1}	AR[h] (kg TN ha^{-1})	AR (kg $NH_4{}^+$–N ha^{-1})
Storage	CA[a]	51.7	40	2.3	—	1.2	7.4	0.53		
	DF[b]	46.7	32.3	2.8	—	1.7	7.8	0.61		
	LF[c]	32.3	23.6	2.5	—	1.7	7.8	0.66		
	AmS[d]	—	—	—	—	—	—	—	—	
Field application	CA	51.7	—	2.5	78	1.2	7.7	0.48	107	51.4
	DF	40.5	—	2.8	56	1.6	8	0.57	98.8	56.3
	LF	27.4	—	2.5	45	1.5	8	0.6	95.1	57
	AmS	10.5	—	1.2	22	0.8	7.3	0.67	114.9	77

[a] CA: untreated cattle slurry.

[b] DF: anaerobic digestate of cattle slurry co-digested with grass–clover.

[c] LF: liquid fraction of DF.

[d] AmS: liquid ammonium sulfate fertilizer, which is considered not to emit greenhouse gases during storage.

[e] Volatile solid.

[f] Total nitrogen.

[g] Total carbon.

[h] Application rate.

Table 2. Summary of emission factors, environmental impacts and productivity parameters used for carbon footprint estimations of organic fertilizers.

Process/parameter	Emission factor	Unit	Source
Combustion of diesel	2.82	kg CO_2-eq per l^{-1} diesel	Agrifootprint 5 database, 2015
Transport (1 tkm[a])	0.37	kg CO_2-eq tkm^{-1}	Agrifootprint 5 database, 2015
Provision of electricity[b]	0.56	kg CO_2-eq kWh^{-1}	Ecoinvent 3 database, 2013
Provision of heat (natural gas)	0.07	kg CO_2-eq MJ^{-1}	Ecoinvent 3 database, 2013
Provision of heat (oil)	0.09	kg CO_2-eq MJ^{-1}	Ecoinvent 3 database, 2013
Carbon sequestration (100 year)	0.09	kg CO_2-eq kg^{-1} applied C	(Petersen *et al* 2013)
Biogas production rate[c]	18.4	m^3 of biogas ton^{-1} substrate	Measured in this study
Biogas fugitive loss	2.7%	The percentage of biogas produced	(Naroznova *et al* 2016)
Heat value	22.4	MJ m^{-3} biogas	Measured in this study
Energy production from biogas	3.01	kg CO_2-eq m^{-3} natural gas	(Paolini *et al* 2018)
Avoided energy use by the substitution of biogas for natural gas	31.51	kg CO_2-eq ton^{-1} substrate	(Paolini *et al* 2018) and data in this study

[a] 1 tkm represents the transportation of 1 ton of organic fertilizer for 1 km by truck.

[b] Electricity mix in Denmark.

[c] Biogas consists of approximately 60% CH_4 and 40% of CO_2.

2.2.2. Feedstock processing at the biogas plant

2.2.2.1. Anaerobic digestion and storage of the DF

A mixture of 92.5% cattle slurry and 7.5% grass–clover (by fresh weight) was used as a substrate for anaerobic digestion in an experimental biogas reactor. The transport distances for cattle slurry and grass–clover to the biogas plant were 8 and 10 km, respectively. A reactor with a capacity of 10 m^3 and an active volume of 9 m^3 was loaded with 270 kg of the substrate mixture each day, and the digestion process temperature was 51 °C with a hydraulic retention time (HRT) of 33 d. Emissions related to physical buildings and manufacturing of equipment used for treatment, storage and field application were not considered in this study due to their long lifespan. Tables A5 and A6 (appendix A) provide the data used for anaerobic digestion and storage of DF fertilizer, respectively.

2.2.2.2. Separation of digestate and storage of the LF fertilizer

The digestate was stored at 20 °C for a period of 30 d during which residual biogas was collected. This temperature was maintained without the need for additional energy input. Tables A7 and A8 (appendix A) present the data used for separation of the anaerobic digestate and storage of LF fertilizer, respectively.

2.2.2.3. Production of ammonium sulfate concentrate from drying of the solid fraction and biogas cleaning (AmS fertilizer)

To produce AmS fertilizer, hydrogen sulfide (H_2S) in the crude biogas was oxidized to sulfur (S) or sulfuric acid (H_2SO_4) in a desulfurization filter with a biofilm of bacteria fed with nutrients from filtered (50 μm, Sepcom, Italy) LF material. The pH of the filtrate thereby decreased from 7.9 to 2.0. Ammonia (NH_3) volatilized during drying of the solid fraction, derived from digestate separation, was then trapped in the acid solution. More details regarding AmS fertilizer is provided in the study by Meng *et al* (2023). Tables A9–A11 (appendix A) present the data used for the production of AmS fertilizer and its storage, respectively.

2.2.3. Field application of fertilizers

In April 2021, the CA, DF, LF and AmS fertilizers were applied to spring barley (Hordeum vulgare L) on a loamy sandy soil. No insecticides or herbicides were used, and all fertilizers were surface-applied at a rate of 100 kg TN ha^{-1} and immediately incorporated by plowing (Meng *et al* 2023).

The soil organic carbon change related to the application of slurry-based fertilizers is defined as a carbon deficit (or credit, indicated by negative values) with the unit tons C $year^{-1}$, and represents the amount of extra carbon temporarily added to or removed from the soil compared to a reference system (I Canals *et al* 2007). This was calculated considering a 100 year perspective and assumed an emission reduction potential of 9.7% of the net C input (Petersen *et al* 2013). Possible changes in GHG emissions due to the substitution of mineral fertilizers were not considered because the nutrients in cattle slurry will be recycled to land in any case. The total CO_2 uptake by plants and biogenic emissions were not considered as it was assumed to be re-released during residue decomposition or consumption of harvested plant material. The

Table 3. Parameters considered for the sensitivity analysis in this study.

Fertilizer	Value for sensitivity analysis	Storage CH_4 emission (kg C kg OM^{-1})	Field N_2O emissions (kg N kg TN^{-1})	Fugitive biogas (%)	
CA[a]	This study	0.032	0.008	This study	2.7
	Lower value	0.014[e]	0.002[g]		
	Higher value	0.043[f]	0.018[g]		
DF[b]	This study	0.009	0.012		
	Lower value	0.004[e]	0.002[g]	Lower value	0.005[h]
	Higher value	0.012[f]	0.018[g]		
LF[c]	This study	0.002	0.008		
	Lower value	0.001[e]	0.002[g]		
	Higher value	0.003[f]	0.018[g]	Higher value	10.7[i]
AmS[d]	This study	0.002	0.032		
	Lower value	0.001[e]	0.002[g]		
	Higher value	0.003[f]	0.018[g]		

[a] CA: untreated cattle slurry.

[b] DF: digestate resulting from anaerobically digesting cattle slurry and grass–clover together.

[c] LF: liquid component extracted from anaerobic digestate.

[d] AmS: liquid fertilizer from ammonia released during drying of solid digestate, captured with biogas-derived sulfuric acid.

[e] De Vries *et al* (2012).

[f] Ten Hoeve *et al* (2014).

[g] IPCC (2019).

[h] Liebetrau *et al* (2013).

[i] Baldé *et al* (2022).

inventory data related to the field application of CA, DF, LF and AmS fertilizers are further detailed in tables A12–A15 (appendix A).

2.3. Sensitivity analysis

Sensitivity analysis was performed by considering different values for CH_4 emission during storage, and for N_2O emissions in the field, and different percentages of fugitive biogas (table 3). The upper and lower values of CH_4 emission during storage were based on Ten Hoeve *et al* (2014) and De Vries *et al* (2012), respectively. Both the lower and upper emission factors of field N_2O emissions used in the sensitivity analysis were obtained from IPCC (2019). The different percentages of fugitive biogas, ranging from 0.005% (Liebetrau *et al* 2013) to 10.7% (Baldé *et al* 2022).

3. Results

3.1. Carbon footprint of 100 kg TN in organic fertilizers (FU1)

Without avoided emissions, the gross GWP of 100 kg TN in cattle manure (CA) was estimated to be 2490 kg CO_2-eq (table 4). The GWP of DF, LF, and AmS fertilizers corresponded to 49%, 41%, and 79%, respectively, of the gross GWP for CA (table 4). By considering the avoided emissions from biogas use, the net GWP was found to be even lower for DF, LF and AmS fertilizers corresponding to 24%, 21% and 49%, respectively, of the net GWP of CA (table 4).

The storage period contributed the most to gross GWP, accounting for 83%, 48%, 23% and 24% for CA, DF, LF and AmS fertilizers, respectively (figure 2(a)). Anerobic digestion was the second-largest contributor, accounting for 13%, 21% and 17% of the gross GWP for DF, LF, and AmS fertilizers, respectively (figure 2(a)). Meanwhile the substitution of natural gas by biogas (avoided energy use) accounted for 51%, 48% and 39% of the gross GWP for DF, LF, and AmS fertilizers, respectively (figure 2(a)). Among GHGs, CH_4 emissions from tank storage contributed the most to GWP for CA and DF (figure 3). The CH_4 emissions from anerobic digestion and transport emissions were the main contributors to GWP for LF and AmS. The contributions of NH_3 emission and electricity and heat consumption to the total GWP were marginal.

3.2. Carbon footprint of 1 ton of spring barley grain yield (FU2)

For the production of 1 ton of spring barley grain yield (FU2), the gross and net GWP for the baseline scenario were 501 and 409 kg CO_2-eq, respectively (table 4). The gross GWP for DF, LF and AmS fertilizers were 62%, 53% and 178% of the gross GWP of CA, respectively, while the net GWP were −6%, 6% and 177% of the net GWP of CA, respectively (table 4).

The largest contributor to GWP was again storage prior to field application for CA and DF fertilizers, while field application was the main source for LF and

Table 4. The GWP (kg CO_2-eq per FU) of different organic fertilizers.

GWP	Fertilizer	FU1[g]	FU2[i]
Gross[a]	CA[c]	2490 (100%)[h]	501 (100%)
Gross	DF[d]	1229 (49%)	311 (62%)
Gross	LF[e]	1020 (41%)	265 (53%)
Gross	AmS[f]	1974 (79%)	890 (178%)
Net[b]	CA	2490 (100%)	409 (100%)
Net	DF	599 (24%)	−24 (−6%)
Net	LF	528 (21%)	24 (6%)
Net	AmS	1212 (49%)	725 (177%)

[a] Gross GWP indicates global warming potential without considering avoided emissions.
[b] Net GWP indicate global warming potential considering avoided emissions.
[c] CA: untreated cattle slurry.
[d] DF: anaerobic digestate of cattle slurry co-digested with grass–clover.
[e] LF: liquid fraction of DF.
[f] AmS: liquid ammonium sulfate fertilizer.
[g] Functional unit 1 (FU1): 100 kg of total nitrogen (TN) of each type of fertilizer.
[h] The percentage indicates the GWP percentage of CA.
[i] Functional unit 2 (FU2): 1 ton of spring barley grain yield.

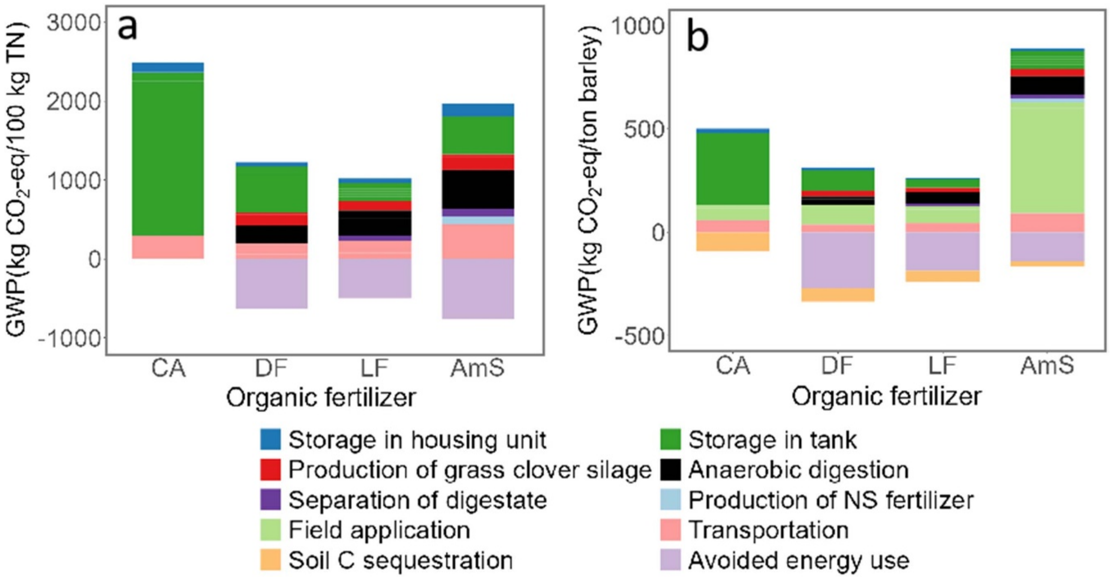

Figure 2. GWP [kg CO_2-eq per FU] of different organic fertilizers, evaluated under two functional units: FU1 (a) representing 100 kg of total nitrogen (TN) for each fertilizer type, and FU2 (b) representing 1 ton of spring barley grain yield. Fertilizer types include CA (untreated cattle slurry), DF (anaerobic digestate of cattle slurry co-digested with grass–clover), LF (liquid fraction of DF), and AmS (liquid ammonium sulfate fertilizer).

AmS fertilizers (figure 2(b)). The saved biogas energy in this study accounted for 86%, 71% and 16% of the GWP for DF, LF and AmS fertilizers, respectively. The main source of GHG emissions for the CA was CH_4 emission during storage, while N_2O emissions during field application contributed the most to the GWP of other fertilizers (figure 3). Meanwhile, LF fertilizers had a very small contribution of CH_4 (figure 3(f)).

3.3. Sensitivity analysis

Figure 4 shows the results of the sensitivity analysis on carbon footprint (net GWP) values for CH_4 emission during storage (figure 4(a)), three measured, upper, and lower values for field N_2O emission (figure 4(b)) and three percentages of fugitive biogas (figure 4(c)).

The results of the sensitivity analysis showed similar rankings of GHG emissions from organic fertilizers considering measured, upper, and lower values for CH_4 emission during storage and N_2O emission after field application, and three percentages of fugitive biogas (figure 4). When considering upper and lower boundaries related to CH_4 emission during storage, CA showed greater differences in the amounts of GHG emissions compared to DF, whereas this difference was not significant for LF or AmS fertilizers (figure 4(a)).

With the upper value for field N_2O emissions, the overall GWP increased by 18%, 168% and 328% for CA, DF and LF, respectively. In contrast, the value decreased by 31% for AmS (figure 4(b)). When

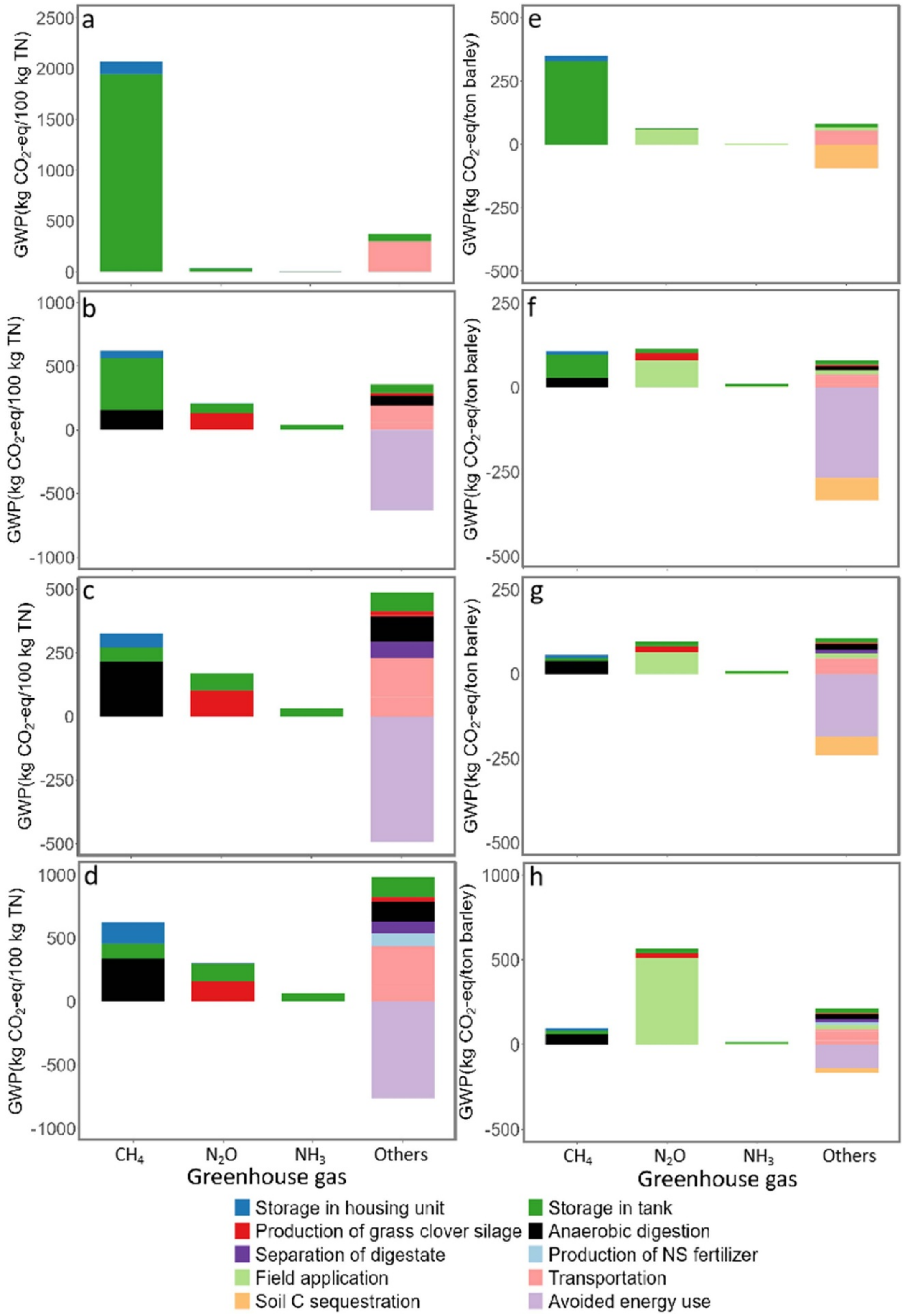

Figure 3. Contribution of different GHG emissions to GWP [kg CO_2-eq per 100 kg TN] for different organic fertilizers considering FU1: 100 kg of total nitrogen (TN) in each type of fertilizer (a: CA, b: DF, c: LF, d: AmS). Contribution of different GHG emissions to GWP [kg CO_2-eq per ton spring barley grain yield] for different organic fertilizers considering FU2: 1 ton of spring barley grain yield (e: CA, f: DF, g: LF, h: AmS). 'Others' include GHG emissions from transportation, energy and electricity use, and avoided emissions. CA refers to untreated cattle slurry, DF to anaerobic digestate of cattle slurry co-digested with grass–clover, LF to liquid fraction of DF, and AmS to liquid ammonium sulfate fertilizer.

applying a lower value for field N_2O emission, the GWP of CA decreased slightly, while the GWP of DF, LF and AmS decreased by 280%, 197% and 66%, respectively (figure 4(b)). Considering the lowest percentage of fugitive biogas in the calculation of GHG emissions, the GWP of all the fertilizers decreased

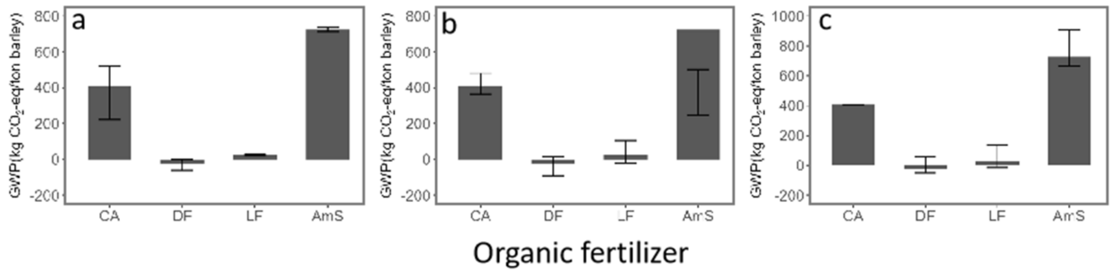

Figure 4. Sensitivity analysis of net GWP [kg CO_2-eq/ton barley] considering three measured, upper and lower values for CH_4 emissions during storage (a), three measured, upper and lower values for field N_2O emissions (b) and three percentage of fugitive biogas (c). Negative error bars represent values with low emission coefficients, and the positive error bars represent values with high emission coefficients. CA: untreated cattle slurry; DF: CA refers to untreated cattle slurry, DF to anaerobic digestate of cattle slurry co-digested with grass–clover, LF to liquid fraction of DF, and AmS to liquid ammonium sulfate fertilizer.

slightly (figure 4(c)). In contrast, with the highest percentage of fugitive biogas, the GWP of DF, LF and AmS increased by 80, 113 and 184 kg CO_2-eq per 1 ton barley, respectively, while the GWP of CA increased slightly (figure 4(c)).

4. Discussion

4.1. Effectiveness of anaerobic digestion and post-treatments

Based on the results of this study, both anaerobic digestion of cattle slurry with grass–clover, and the subsequent separation of digestate, significantly reduced the GWP of the partially or highly processed cattle slurry compared to CA. Mezzullo *et al* (2013) and Valenti *et al* (2020) similarly concluded that anaerobic digestion of slurry is a sustainable approach with high potential for reducing GHG emissions.

In a study by Pexas *et al* (2020), anaerobic digestion significantly reduced the GWP by 9.2% compared to untreated pig slurry with respect to housing, storage and field application. However, screw press slurry separation increased the GWP by 6.4% compared to untreated pig slurry. Similarly, Prapaspongsa *et al* (2010) found that anaerobic digestion significantly reduced the GWP when considering treatment, storage, transport and field application, and the substitution of biogas for electricity and heat resulted in an overall negative GWP. Hamelin *et al* (2011) conducted a comparative analysis of the GWP associated with various separation technologies used in the treatment of slurry digestate. Processes such as in-house and outdoor storage, anaerobic digestion, field operations and transportation were considered as positive values, and yield increase, avoided fertilizers, electricity, and heat, as negative values. Notably, the most efficient separation technology in their study resulted in a significant GWP reduction of up to 40% compared to untreated pig slurry.

Biogas from anaerobic digestion of organic materials can be converted on-site to heat or energy, or purified to produce biomethane for injection into the natural gas grid as credit for the fertilizer production (Patterson *et al* 2011). Biogas typically contains 45%–65% CH_4 and 35%–55% CO_2 by volume, the composition varying depending on temperature, pH, and HRT (Møller *et al* 2009). The amount of biogas produced is a key factor in the total GHG balance and depends on temperature, organic loading rate, and HRT (Nasir *et al* 2012). The GHG emissions from the biogas facility can range from 5 to 76 CO_2-eq per ton of substrate depending on the technology used considering both upstream and downstream processes, and waste management (Møller *et al* 2009). Weiske *et al* (2006) reported that the net GWP of digestate can vary from −96% to +26% compared to the CA depending on the quantity and quality of organic matter in substrates, as well as biogas production efficiency. The use of biogas in combined heat and power units (CHPU), or for production of biomethane intended for transport, shows different energy potentials. Thyø and Wenzel (2007) reported that the optimal strategy for achieving GHG mitigation lies in using biogas on-site for CHPU rather than converting it to biomethane. On the other hand, when biogas replaces different types of energy sources, the CO_2 savings are different, and the savings are lower if biogas is burned in flares or transported excessively (Poeschl *et al* 2012).

4.2. Effectiveness of alternative organic fertilizers

The LCA of cattle slurry-based fertilizers considering 100 kg TN (FU1) showed a net GWP reduction of 76%–79% when CA was replaced with digestate or the LF of digestate, and reductions of 94%–106% when considering the harvest of 1 ton grain yield of spring barley (FU2), and disregarding the AmS treatment. Methane emission during storage of CA was identified as the main contributor to GWP with both functional units, which is consistent with results reported by Ten Hoeve *et al* (2014) and Ramírez-Islas *et al* (2020). The level of GHG emissions during storage depends on several factors, including pH, storage period, temperature, and crust formation (Meng *et al* 2023).

Ten Hoeve *et al* (2014) reported a gross GWP of approximately 60 kg CO_2-eq per ton of untreated pig slurry in an LCA accounting for storage, field emissions, and transport processes. Using field application of 1 ton of untreated pig slurry as functional unit, they compared this treatment with processed pig slurry using four different separation technologies. Ten Hoeve *et al* (2014) found higher field emissions than our study, but separation, especially by centrifugation, had a lower environmental impact potential compared to the application of untreated pig slurry. In their study, the solid fraction was assumed to be covered air-tight, or to be composted, whereas in the present study the solid fraction was assumed to be dried with trapping of ammonia; both methods can ensure low emissions during storage (Hansen *et al* 2006, Cao *et al* 2020).

For LF and AmS fertilizers with low or no emissions during storage, field emissions were identified as the main contributor to GWP, while for DF fertilizer the GWP contribution from farm emissions was comparable to that from storage. The amount of GHG emissions after field application is influenced by various factors such as fertilizer properties, soil type, climate, and application method. Prapaspongsa *et al* (2010) compiled results indicating higher N_2O emissions, corresponding to about 10–20 kg CO_2-eq per ton of slurry, for injected compared to surface-applied slurry, showing the importance of defining field application method when estimating GWP. However, the variability of N_2O emission factors among underlying studies was high and probably influenced by the other factors listed above. The LCA presented here aimed to reduce the uncertainty of this source by experimental determination of N_2O emissions from the fertilizers investigated.

In the present study, AmS fertilizer showed a higher GWP compared to other fertilizers. This was primarily attributed to the experimental design, which involved a direct comparison of fertilizer materials at a fixed N rate. The AmS fertilizer required an application of 83 t ha^{-1} to achieve 100 kg TN per hectare, necessitating split application over two days and resulting in wet and compacted soil supporting high N_2O emissions after field application (Meng *et al* 2023). In practical terms, the AmS concentrate may be more suitable as a starter fertilizer to winter cereals or maize, typically applied at rates around 30 kg N ha^{-1}. Soil water-filled pore space plays a crucial role in creating conditions conducive to N_2O emissions through processes such as nitrification and denitrification (Davidson *et al* 2000), suggesting that reducing the volume of liquid manure could mitigate the risk for N_2O emissions.

Selecting an appropriate functional unit is critical, as it determines how results are expressed and compared within the same agricultural system. Different functional units can lead to results that are not directly comparable or may emphasize different aspects of the system (Martínez-Blanco *et al* 2011). For example, one FU may focus on nutrient application, while another focuses on crop yield, each highlighting different environmental impacts or benefits. In previous studies, the functional unit has often been based on the handling of a specific quantity (Styles *et al* 2018) or the production of organic fertilizers (Gaidajis and Kakanis 2020). However, the nutritional value of organic products and the resulting crop yields are also important for the evaluation of agricultural systems. Nevertheless, comparisons of treatments based on crop production are relatively rare in LCA studies. Our results indicate that both the individual environmental impacts and their ranking among treatment options were significantly influenced by the chosen functional unit.

The manure treatment influences the quantities of direct and indirect N_2O emissions, CH_4 emissions, and food production. Since these gases are produced through microbial activity, factors such as the DM content and nitrogen availability in manure are crucial in determining the extent of GHG emissions. During storage, the labile C and N in manure contribute to CH_4 and N_2O emissions, partly through NH_3 emissions. Anaerobic digestion of livestock manure has been shown to lower CH_4 emissions during storage (Im *et al* 2020). Additionally, the LF of manure helps to reduce surface crust formation. While a crust significantly lowers NH_3–N and CH_4 emissions, it can also encourage the production and emission of N_2O (Petersen 2018). After applying manure to fields, optimizing the timing within the growing season can help reduce soil N_2O emissions. Although the LF of digestate has the benefit of delivering nutrients (e.g. N) to plants more quickly, soil N immobilization has been observed in biomasses with a C/N ratio exceeding 25–30 (Chiyoka *et al* 2014). Therefore, characterizing a production system using experimental data can significantly decrease the uncertainty associated with carbon footprints.

Carbon in slurry-based fertilizers is partially sequestered in the soil following field application. All organic fertilizers in this study showed potential for C sequestration, but CA showed more potential for carbon sequestration compared to the treated organic fertilizers. This is consistent with a study by Quirós *et al* (2015) on LCA of organic and mineral fertilizers in a crop sequence of cauliflower and tomato. In comparisons of organic fertilizers it is important to consider C losses prior to field application, and Thomsen *et al* (2013) in a study comparing cattle feed, cattle slurry, digested cattle feed, and digested cattle slurry, concluded that long-term soil C sequestration of untreated and treated organic fertilizers would be similar to that of C in the original cattle feed at 12%–14%.

4.3. Sensitivity analysis

Among the three parameters used to evaluate the robustness of the GWP results of organic fertilizers in this study, adjusting the N_2O emission factor had the greatest impact for DF, LF and AmS, increasing net GWP by 168%, 328% and 31% for DF, LF and AmS, respectively. This reduced the advantage of LF over DF and highlighted the importance of quantifying soil N_2O emissions after field application. Ten Hoeve *et al* (2014) also emphasized the importance of carefully selecting N_2O emission factors when calculating GWP based on soil properties, fertilizer characteristics, and environment by performing sensitivity analysis on different parameters.

The impact of uncertainty in estimating CH_4 emissions during storage was greater for CA fertilizer due to its higher content of degradable organic matter. Baral *et al* (2018), based on a pilot-scale study, concluded that the most effective mitigation strategy for untreated slurry involves reducing CH_4 emissions during storage. At the highest percentage of fugitive methane emissions, the net GWP of DF and LF increased approximately twofold, primarily due to the very low carbon footprint from clean energy production. Evangelisti *et al* (2014) conducted an LCA of energy from waste via anaerobic digestion in England and reported that fugitive emissions strongly influence the GWP of anaerobic digestate. In their study, increasing fugitive emissions from 2% to 5% resulted in a more than 55% increase in GWP, while further increasing to 15% reversed the GWP decrease to a net increase. Similar results were reported by Møller *et al* (2022).

4.4. Evaluation of methodology and uncertainties

While the environmental benefits of organic fertilizers are widely acknowledged, much of the supporting evidence, including that from the present study, comes from small-scale studies, pilot projects, or modeling efforts. There are still gaps in understanding the optimal use of alternative feedstock and processes for anaerobic digestion, as well as the effects of post-treatment technologies.

In the experiment behind this study, grass–clover silage was used for co-digestion which degrades easily and primarily affects nutrient composition, but other biomasses may produce digestates with significantly different properties (Møller *et al* 2009, 2022). Evaluation of the organic fertilizer quality, however, was beyond the scope of this study. Our study showed that using partially or highly processed fertilizer could reduce the GWP compared to untreated slurry, contributing to CO_2 reduction. However, this potentially important finding needs to be corroborated in studies with other organic fertilizers that differ in feedstock, treatment, and field application conditions.

The environmental assessment of this study was limited to GWP but could have included other environmental impacts such as eutrophication potential, acidification potential, and particulate matter formation. Integrating this with an economic assessment could illustrate the additional costs required to achieve CO_2 reduction and explore various organic fertilizers as well as methods for estimating GHG emissions. Such integration would enable decision-makers to select new CO_2 reduction solutions based on effective and efficient post-treatment technologies for organic fertilizer production.

4.5. Perspectives

This work demonstrated how alternative organic N fertilizers can potentially reduce GHG emissions compared to untreated manure. Achieving additional CO_2 reduction from organic fertilizers requires a focus on mitigating GHG emissions during storage of slurry-based fertilizers (mainly CH_4), and after their field application (mainly N_2O). This implies the need to select and evaluate GHG mitigation measures to maximize GHG reduction while avoiding crop yield loss. Mitigation options mentioned include acidification during storage, cooling, using a porous cover to support CH_4 oxidation, and adopting appropriate field application methods and timing. Importantly, the effectiveness of organic fertilizers in terms of CO_2 reduction is affected by the type of the feedstock and their specified post-treatment stages. The potential effectiveness of organic fertilizers depends on identifying combinations of agricultural waste materials and residues as feedstock that can be processed using low-energy processing technologies.

5. Conclusion

Our study shows that the carbon footprint of the production and storage of alternative organic fertilizers per 100 kg TN ranged from 21% to 49% of the baseline scenario using CA. Upon application to the field, the carbon footprint per ton DM spring barley grain yield ranged from −6% to 177% of the baseline scenario. The primary contributors to the carbon footprint of these fertilizers were emissions during storage and after field application, while biogas production from anaerobic digestion significantly reduced the GWP. This highlights the substantial potential for mitigating climate impact by replacing untreated slurry with alternative fertilizers derived from anaerobic digestion, primarily through reduced emissions during storage and biogas production.

Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files).

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