Life cycle environmental impacts of compressed biogas production through anaerobic digestion of manure and municipal organic waste

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Acronyms

BAU	Business-as-Usual
CBG	Compressed Biogas
CSTR	Continuously Stirred Tank Reactor
EU	European Union
GHG	Greenhouse Gas
GWP	Global Warming Potential
JRC	Joint Research Centre
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
MOW	Municipal Organic Waste
NG	Natural Gas
PAO	Potential Ammonia Oxidation rate
RED	Renewable Energy Directive
SE	Sweden

1. Introduction

Climate change mitigation is one of the main goals of the European biofuels policy, along with energy security and rural development (Su et al., 2015). For this reason, the latest update of the European Union (EU) Renewable Energy Directive (RED) raised the target of renewable energy in the transport sector to 14% by 2030 ("Council directive 2018/2001", 2018). For the light-duty sector, battery electric vehicles are the best alternative to fossil-fueled cars from a life cycle perspective considering all environmental impact categories (Hooftman et al., 2016; Messagie et al., 2014), but the heavy-duty and water-borne sectors are not likely to be fully electrified soon (Çabukoglu et al., 2018; European Commission, 2017a). Therefore, advanced biofuels are also actively being promoted by the EU to mitigate climate change with a minimum blending target of 3.5% by 2030 ("Council directive 2018/20018", 2018). Compressed biogas (CBG) is produced by upgrading and compressing biogas from anaerobic digestion of second-generation feedstock and it is one of the most promising advanced biofuels (Puricelli et al., 2020), with a reported carbon footprint between -102.9 and 26.3 gCO_2eq/MJ_{fuel} (JEC, 2020). Thus, this fuel complies with the RED requirements of 60% greenhouse gas (GHG) emissions savings compared to fossil diesel and is eligible for financial support, causing the European anaerobic digestion market to grow rapidly (Pablo-Romero et al., 2017). Recently, engine technology for methane combustion has greatly improved, achieving similar or higher efficiencies than the diesel engine (Sawamoto, 2018). It is the most mature biofuel for heavy-duty vehicles and vessels at the moment (European Commission, 2020). On top of that, increased biogas utilization as a transportation fuel can be expected over the coming years to a certain extent. For example, in Belgium, the Walloon region aims to fuel 25% of their trucks, 10% of their busses, and 18% of their cars with Compressed Natural Gas (CNG) in 2030, of which 15% will be CBG (National Climate Commission, 2019). The estimated biogas potential in Europe is over four times the actual biogas production (Scarlat et al., 2018). However, there is a limit to this growth. The total biogas potential from manure is estimated to be 736 PJ (Scarlat et al., 2018), around 4% of the final energy consumption in the transport sector in 2017 (European Commission, 2020). Still, CBG can be one option among many others in the combat against climate change.

But climate change is not the only environmental concern that matters. Many Life Cycle Assessment (LCA) studies on biogas systems including other environmental impact categories have been published in recent years, but the results are often not comparable. Methodological choices can significantly influence the results and there is a need for common methodological guidelines for LCA on biogas systems (Bacenetti et al., 2016; Hijazi et al., 2016). Fusi et al. (2016) found that many environmental impacts, such as ozone layer depletion potential, ecotoxicity potential, human toxicity potential, and abiotic depletion potential, are mainly related to the electricity requirements of the digester. The performance of CBG on these impact categories relies on the assumptions taken regarding electricity use. Another important link was found between the methodological choices and results. Digestate application to fields and synthetic fertilizer application have been identified as environmental hotspots (Hartmann, 2006; Stuckl et al., 2011; Timonen et al., 2019). When these life phases are considered, acidification and eutrophication are the most important impact categories (Fusi et al., 2016; Hartmann, 2006; Hijazi et al., 2016). These issues are closely linked to fertilization. Fertilizers largely contribute to ammonia, nitrate, and phosphate pollution, hence their use contributes to acidification and eutrophication in terrestrial and aquatic ecosystems (Czyrnek-Delêtre et al., 2017; Sumner, 2009). Not only synthetic fertilizers lead to these environmental problems. For example, biofertilizers such as manure and digestate, a by-product of anaerobic digestion, have various benefits, such as improving soil structure and soil organic carbon content (Carvajal-Muñoz and Carmona-Garcia, 2012). However, they can significantly impact eutrophication and acidification, particularly when poorly managed (Durlinger et al., 2017). Due to their high moisture content, ammonification and nitrification occur easily in biofertilizers, which leads to volatilization of nitrogen species and nitrate leaching (Bernal et al., 2015). The ratio of nutrients in biofertilizers is often unbalanced, which is why excess supply to the soil is a risk. Moreover, the nutrients in biofertilizers are not as quickly available to the plants, as is the case with synthetic fertilizer, and farmers might be tempted to oversupply (Bernal et al., 2015). It is thus important to look at climate change when studying biogas, but it also relevant to assess eutrophication and acidification to know the potentially harmful impact of the produced biofertilizers.This will become only more relevant in the future, as the growing demand for agricultural commodities is driving the fertilizer consumption to rise.by 30% by 2050 (FAO, 2018). The importance of including these impact categories is stressed even more by the planetary boundary framework (Steffen et al., 2015). This framework tries to define to what degree the human population can safely change its environment without inciting unrecoverable perturbations that threaten the Holocene-state of the Earth. The current understanding is that the alteration in biochemical flows, causing acidification and eutrophication, is more likely to cause unrecoverable disruptions rather than climate change (Steffen et al., 2015).

Three different views have been identified on how to deal with the spreading of digestate for the case where biogas is upgraded and used as a transportation fuel: *allocation*, *partial substitution*, and *full substitution.* The RED guidelines are an example of the first view and recommend energy allocation to solve multifunctionality, considering that no impacts are attributed to the digestate generation as its energy content is zero ("Council directive 2018/2001", 2018). There is one exception. If the use of digestate as a fertilizer enhances the soil carbon content, this increase of carbon stock can be included in the GHG balance. RED also gives some further sustainability criteria concerning the conservation of biodiversity and the prevention of land-use change, but acidification and terrestrial eutrophication are not mentioned. Manninen et al. (2013) calculated the climate change impacts of biogas from a mix of waste streams following four different interpretations of the RED methodology resulting in emissions savings ranging from 42% to 80%, depending on whether the solid and liquid fraction of the digestate was considered a co-product or a by-product. These results show that even when following the RED guidelines, different interpretations of the rules can lead to very different results. Therefore, the development of clear guidelines for the classification of relevant materials was their recommendation to enhance the comparability of results. The "*JEC Well-to-Tank report v5*", regarded as the reference study on biofuels in the EU (JEC, 2020), henceforward referred to as "JEC" is an example of what is here called: partial substitution. This study calculated the global warming potential of CBG using the substitution method. They gave an environmental credit for the substitution of synthetic fertilizer *production*, but not for its *use*, as spreading the digestate was outside the system boundary.The last view, in our work defined as full substitution, means that both synthetic fertilizer production and their use are considered. Tufvesson et al. (2013) performed an LCA on biogas production from industrial residues, also following the full substitution approach, and they substituted for both synthetic fertilizer *production* and *use*. For the impact category climate change, they compare the case where the industrial residues and digestate have no current utilization and the case where they displace respectively animal feed and mineral fertilizers to the RED methodology. They concluded that RED's methodology has a limited systems perspective and showed that the displacement of animal feed production decreases the climate change benefit considerably. In contrast, the displacement of mineral fertilizers has a moderate reducing effect on the impact. They also calculated the acidification and eutrophication potential with the substitution approach, and the spreading of digestate was found to be a hotspot for acidification. Börjesson et al. (2015) investigated the GHG performance of CBG from energy crops using the substitution approach for both synthetic fertilizer *production* and *use*. They showed that for energy crops, the RED methodology leads to lower savings than the substitution method. They find the latter approach more holistic and suggested that RED should use it as well. It is clear that the RED guidelines, using allocation, are regularly used in CBG LCAs as a benchmark, but most authors prefer the partial or full substitution method. For the impact category climate change, the full substitution method leads to a more favorable result than the RED guidelines, but acidification and eutrophication impacts are either neglected (Börjesson et al., 2015; JEC, 2020; Manninen et al., 2013) or only briefly discussed (Tufvesson et al., 2013). A comparison between the partial and full substitution method has not yet been made.

In summary, (i) CBG from wastes and residues will likely play a role in the future in the combat against climate change, (ii) it is important to not only look at climate change but also to assess the impacts on acidification and eutrophication since there is a specific risk due to digestate production and spreading, (iii) the *partial and full substitution* methods are preferred but how this affects acidification and eutrophication and what this means for biofuel policies is not clear. This renders it difficult to draw consistent conclusions on the order of magnitude of the environmental effectiveness of CBG.

In this work, we address these issues performing an LCA of CBG production through anaerobic digestion of animal manure and municipal organic waste (MOW) using both the partial and full substitution method to answer the following research questions: (i) what is the impact on climate change, acidification, terrestrial and freshwater eutrophication of CBG production via anaerobic digestion of MOW and manure compared to diesel? (ii) How do the partial and full substitution methods affect the results? What is the uncertainty of the results and what causes this uncertainty? (iii) Which guidelines could be proposed if RED were to use substitution? (iv) Should acidification, terrestrial and freshwater eutrophication be included in the RED requirements for biofuels and, if yes, how?

2. Materials and Methods

This work has been performed following the LCA methodology outlined in the ISO 14040:2006 standard (ISO, 2006a), the ISO 14044:2006 standard (ISO, 2006b), and the guidelines of the Joint Research Centre (JRC) ILCD Handbook (JRC-IES, 2000). The timeframe of this study is the near future (2020-2030) and the geographic coverage is Europe. This LCA is attributional, and thus most bigger system consequences have been neglected. Substitution was chosen for solving multifunctionality. The functional unit, and in this case also the reference flow of all studied scenarios, is the production of 1 MJ_{LHV} compressed biomethane delivered to the vehicle tank.

2.1. Scenarios definition

Two scenarios have been studied: the production of CBG in Europe from MOW digestion (Figure 1) and animal manure digestion (Figure 2). Both scenarios have been assessed with partial and full substitution. Business as usual (BAU) for MOW is collection and transportation to a central treatment plant. The waste must be collected in any case; its generation, collection, and transportation are excluded from the system boundary. The MOW is digestated and the biogas is upgraded by removing CO_2 and H_2S from the raw biogas after digestion (Anaerobic digestion). The products leaving the digester are biogas and digestate. The first is compressed and transported to the dispensing station through the natural gas (NG) low-pressure network (JEC, 2020), during which fugitive methane emissions may occur. The digestate is stored in a closed facility and applied to land as a fertilizer (Digestate application). When partial substitution is used (MOW-PS), Digestate application is not included in the system boundary, and therefore the co-production of digestate only substitutes for Synthetic fertilizer production. When full substitution is applied (MOW-FS), the system boundaries are being expanded to include the function of land fertilization. Digestate application is included, and both Synthetic fertilizer production and Synthetic fertilizer application are substituted.

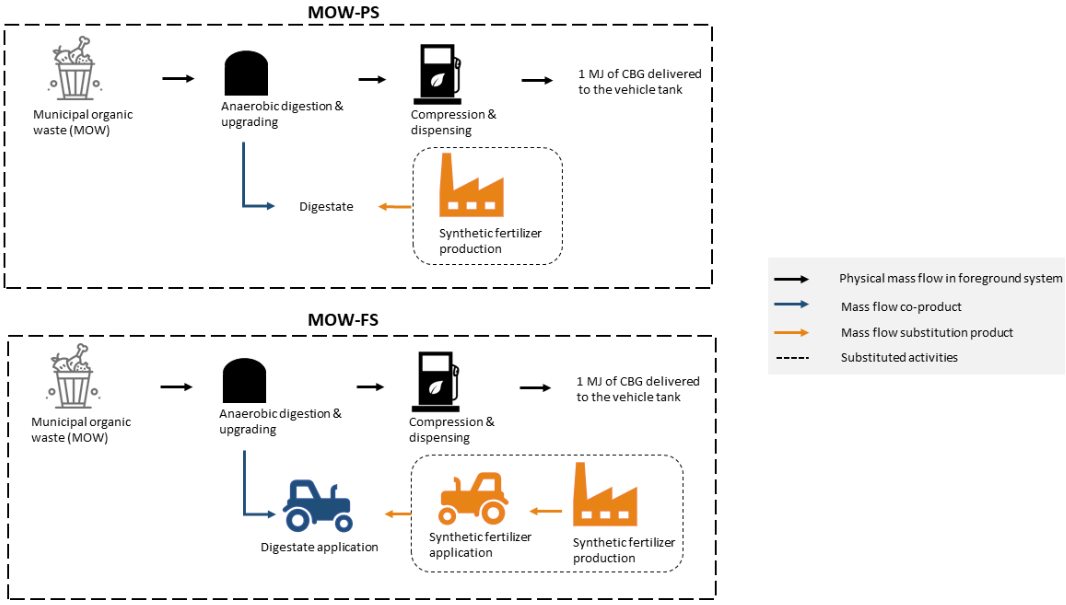

Figure 1: System boundary of compressed biogas (CBG) production via anaerobic digestion of Municipal Organic Waste (MOW) with partial substitution (PS) and full substitution (FS). MOW was considered a waste without economic value and no burden was attributed to its generation. The main difference between MOW-PS and MOW-FS is the absence or presence of Digestate application in the system boundary.

BAU for manure in Europe is storage at the farm for about 6 months (Bernal et al., 2015). It is subsequently applied to agricultural land as biofertilizer (Henning Lyngsø et al., 2011). Typically, 70% of all manure in Europe is stored in a closed system, whereas 30% of storage occurs in an open system (Eurostat, 2013). The feedstock must be transported to the digestion plant, which is here included in the process Anaerobic digestion. After arriving at the digester, the manure is stored before entering the reactor (Dauriat et al., 2011), which is also accounted for in Anaerobic digestion. In Manure-PS (Figure 2), it was assumed that digestate displaces manure and no synthetic fertilizer substitution takes place. This approach was also taken by JEC. In Manure-FS a different approach was taken. As ammonia and dinitrogen emissions due to manure storage have been prevented, the nitrogen content of digestate is higher than the nitrogen content of manure at the field would have been. The digestate has a higher fertilizing value and the additional nutrients displace Synthetic fertilizer production and Synthetic fertilizer application. Manure and digestate application have also been included. In fact, the digestate has a lower mass, which potentially reduced the energy requirements for land application.

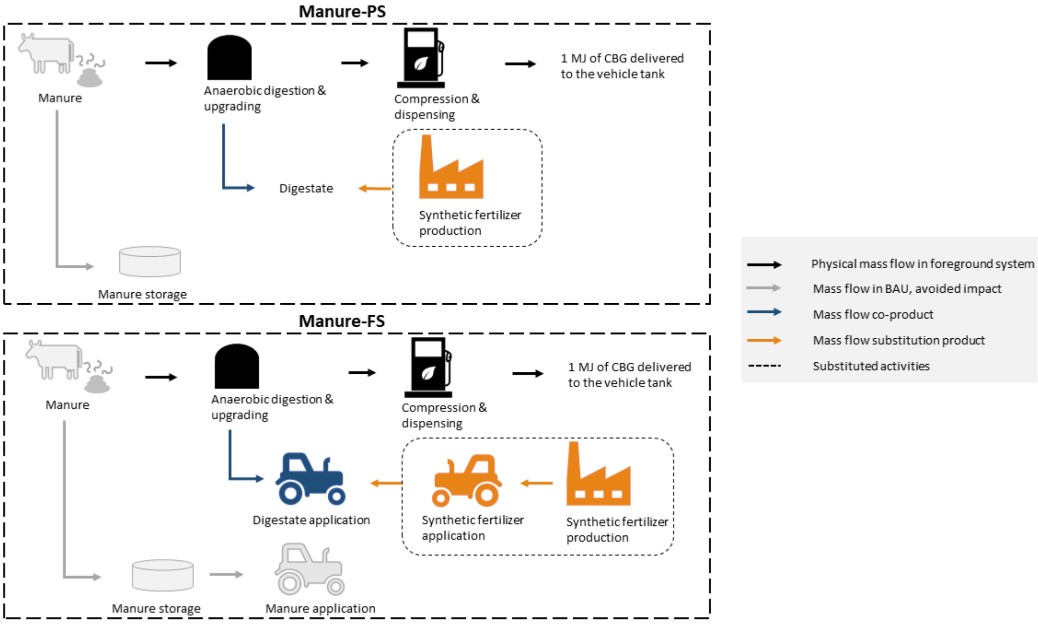

Figure 2: System boundary of compressed biogas (CBG) production via anaerobic digestion of manure with partial substitution (PS) and full substitution (FS). Manure was considered a by-product of animal husbandry. The alternative use (Manure storage) was included as an avoided impact. In both Manure-PS and Manure-FS the difference in nutrient content between digestate and manure is assumed to displace synthetic fertilizers. The main difference between Manure-PS and Manure-FS is the absence or presence of Manure and Digestate application in the system boundary.

The main difference between the two scenarios is the classification of the feedstocks. MOW is considered a waste, which disposal has financial costs (Eunomia Research & Consulting, 2017), whereas manure is considered a useful by-product of animal husbandry (Leip et al., 2019). Wastes do not carry any burden since all impacts are allocated to the process that generated the waste (European Commission et al., 2010). By-products do carry a burden, but in the case of manure there is a lack of consent on how to allocate this burden, since there are no clear biophysical relationships between the inputs of animal husbandry and manure (Mackenzie et al., 2017). Our approach assumes that manure generation is burden-free (all burdens are allocated to the main products of animal husbandry), but its diversion from the current use is not. The burdens related to Business-as-Usual (BAU) for manure are included as avoided impact in the system boundary of Manure-FS. This strategy allows to evaluate the effect of diverting manure from direct land application and has previously been used by (Hill et al., 2020; JEC, 2020; Tufvesson et al., 2013). MOW-FS and Manure-FS also differ in the way the amount of synthetic fertilizers for substitution are defined. In Manure-FS Digestate application has a higher fertilizing value than Manure application and the difference substitutes synthetic fertilizer. In MOW-FS, all of the produced digestate substitutes synthetic fertilizer.

As the main goal of biofuels is replacing fossil fuels, it is important to also understand the impact of CBG compared to the fossil benchmark. Therefore, all scenarios have been compared to fossil diesel, the main fuel used in the heavy-duty sector (ACEA, 2020).

Table 1: Overview of the main assumptions for BAU and the different scenarios considered in our study.

	BAU MOW	MOW-PS	MOW-FS	BAU Manure	Manure-PS	Manure-FS
Feedstock	MOW	MOW	MOW	Manure	Manure	Manure
Feedstock type	Waste	Waste	Waste	By-product	By-product	By-product
Feedstock treatment	Collection and waste treatment.	Anaerobic digestion and upgrading to transportation fuel.	Anaerobic digestion and upgrading to transportation fuel.	Storage at the farm and usage as biofertilizer.	Anaerobic digestion and upgrading to transportation fuel.	Anaerobic digestion and upgrading to transportation fuel.
Transport of feedstock	Transport to central treatment plant.	Transport not included, because it is assumed to have the same impact as in BAU MOW.	Transport not included, because it is assumed to have the same impact as in BAU MOW.	No transport, manure used close to the farm.	Manure is transported to digestion plant.	Manure is transported to digestion plant.
Storage of manure	-	-	-	70% in closed facilities, 30% in open storage.	70% in closed facilities, 30% in open storage.	70% in closed facilities, 30% in open storage.
Storage of digestate	-	Closed storage at the plant.	Closed storage at the plant.	-	Closed storage at the plant.	Closed storage at the plant.
Digestate substitutes	-	Synthetic fertilizer production.	Synthetic fertilizer production and application.	-	Manure fertilizer application.	Synthetic fertilizer production and application

For both scenarios, all subprocesses that contribute more than 5% to at least one of the chosen impact categories have been included in the system boundaries. For this reason, infrastructure and facility construction were not included in the system boundary.

2.2. Inventory analysis

The detailed Life Cycle Inventory (LCI) for MOW-PS, MOW-FS, Manure-PS and Manure-FS, is given in Appendix A. For the composition of MOW in MOW-FS and MOW-PS (Figure 1), average data for all of Europe were not available, mainly because separate collection systems for organic waste do not exist in all member states. Data was therefore taken from a detailed Czech study as a proxy for the European situation (Hanc et al., 2011). This study makes a distinction between the composition of MOW in urban areas and rural areas. In the latter, the share of garden waste is bigger. According to Eurostat, 75% of the European population lives in urban areas and 25% lives in rural areas (Eurostat, 2016), so the results from Hanc et al. (2011) were adapted to represent the European population. The digestion plant is a large-scale facility and treats more than 50,000 tons of substrates yearly. This assumption is based on the fact that CBG production and integration in the NG network are only viable for large-scale plants (JEC, 2020). The digester is a Continuously Stirred Tank Reactor (CSTR). It implies that the dry matter content of the feedstock should not exceed 15%, as this is the requirement for CSTR. This condition can be met by adding water (Bachmann, 2013). It was assumed that no emissions occur during closed storage of the digestate (Fusi et al., 2016) and that no direct methane leakages occur from the digester itself (Liebetrau et al., 2010). Data for Compression & dispensing was taken from JEC (2020). The fertilizer application rate in Digestate application and Synthetic fertilizer application has been based on the average fertilizer application in the EU per fertilized utilized agricultural area: 74.4 kg N/ha, 7.4 kg P/ha and 7.9 kg K/ha (Eurostat, 2015; Fertilizers Europe, 2013). The fertilizer application emissions for all scenarios have been calculated following the IPCC Tier 1 methodology (IPCC, 2006a) and the calculations can be found in Tables A 4, A 6, A 8, A 10 and A 12 in Appendix A. This methodology estimates the national average field emissions, which are no representation of individual field emissions (JEC, 2020). Only emissions that differ between biofertilizer and synthetic fertilizers are considered. These are indirect dinitrogen monoxide emissions after volatilization, ammonia emissions after volatilization and phosphorus emissions after leaching. Changes in soil organic carbon due to digestate application were not considered, as several studies have shown these changes (Barłóg et al., 2020; Möller, 2015; Šimon et al., 2015) and their impact (Tufvesson et al., 2013) are negligible. Data for Synthetic fertilizer production was taken from ecoinvent 3.6 (Wernet et al., 2016). The amounts of synthetic N, P and K fertilizers substituted was based on the N, P and K content of the digestate. But, as the nutrient ratio in digestate from MOW differs from the fertilizer application rate (Table 2), the substitution potential depends on the use behaviour of the farmers (Vadenbo et al., 2017). Either the farmers will balance each nutrient separately by applying the digestate according to the nutrients in excess (P,K), approach that would require the farmers to complement the digestate with synthetic fertilizers, or the farmers maximises their use from digestate to prevent the use of additional fertilizers, which leads to excess application of P and K. The latter approach was chosen in this work, as it is assumed that farmers will choose the cheapest solution. This means that all nitrogen in the digestate of MOW was substituted, but excess P and K was not substituted (Table 2).

The feedstock in Manure-PS and Manure-FS (Figure 2) is a virtual mix composed of 79% pig slurry, 12% cow manure, and 9% poultry litter – based on the average European manure generation mix (Bernal et al., 2015). In Manure storage, 70% of the manure is stored in a closed facility, in which case storage emissions were assumed to be zero (IPCC, 2006b). During open storage, several pollutants are emitted into the air. IPPC Tier 1 formulas (IPCC, 2006b) were used to quantify manure storage emissions (Table A 7 in Appendix A). Used values and sources of these parameters in these equations can be found in Table A 1 in the appendix. It has been assumed that the average temperature in Europe is 10 °C (Cornes et al., 2016).

The application emissions in Manure application (and Digestate application and Synthetic fertilizer application) were calculated similarly to the MOW scenarios. It is assumed that the emission profiles of both biofertilizers (digestate and manure) only depend on the nitrogen content, and the methodology does not distinguish between fertilizers applied by broadcast, band spreading or injection. This means that if the nitrogen contents of digestate and manure were the same, the nitrogen field emissions would be equal too.

Manure is transported to the digester by standard road truck (JEC, 2020), which is included in the process Anaerobic digestion. It is assumed that the storage emissions at the digestion site represent 10% of what the emissions would be at the farm in BAU, because storage time at the digestion plant is much shorter than storage at the farm (Dauriat et al., 2011). The amount of nutrients substituted was calculated by subtracting the nutrients in the manure from the nutrients in the digestate (Table 2). All other data have been taken from the same sources as for MOW-PS and MOW-FS.

Direct CO_2 emissions from biogenic origin (manure and MOW) have been excluded in all scenarios, as carbon neutrality was assumed.

Table 2: Overview of the average EU synthetic fertilizer application rate, the nutrient content of digestate from municipal organic waste (MOW), manure and digestate from manure, and the amount of nutrients substituted.

	N	P	K
Average EU application rate (kg/ha)	74.4	7.4	7.9
Average EU application ratio	1	0.10	0.11
Digestate (MOW) nutrient content (kg nutrient/MJ CBG)	1.1E-3	1.2E-4	1.1E-3
Digestate (MOW) nutrient ratio	1	0.11	1.00
Nutrients in digestate (MOW) substituted (kg nutrient/MJ CBG)	1.1E-3	1.1E-4	1.2E-4
Manure nutrient content (kg nutrient/MJ CBG)	1.1E-2	1.5E-3	3.8E-3
Manure nutrient ratio	1	0.13	0.33
Digestate (manure) nutrient content (kg nutrient/MJ CBG)	1.2E-2	1.5E-3	3.8E-3
Digestate (manure) nutrient ratio	1	0.13	0.33
Nutrients in digestate (manure) substituted (kg nutrient/MJ CBG)	1.0E-3	0	0

2.3. Impact assessment

The EF 3.0 life cycle impact assessment method was used to assess the environmental impacts at the midpoint level (Fazio et al., 2018). The chosen impact categories are climate change, acidification, terrestrial eutrophication (N), and freshwater eutrophication (P). Land use change and biodiversity loss were not investigated, as they are not considered relevant for waste streams and by-products. Water consumption, ionizing radiation and photochemical ozone formation were excluded, because their impact on the total environmental single score impact was minimal (See Figure B 1 and B 2 in Appendix B). Ozone layer depletion potential, ecotoxicity potential, human toxicity potential, and abiotic depletion potential, are mainly related to the electricity requirements of the digester and were excluded, because they largely depend on the assumptions for electricity use, which is not the focus of this paper. Neither is particulate matter formation, although it shortly mentioned in the discussion. For climate change, emissions savings were calculated relative to the RED comparator of 94 gCO_2eq/MJ. For the other impact categories, the EF 3.0 method was applied to the ecoinvent dataset for low-sulfur diesel in Europe (Jungbluth, 2018), and the results were used as a benchmark to calculate the emissions savings. The assessment has been done in SimaPro 8.5.2.0.

2.4. Uncertainty analysis

For all impact categories, a Monte Carlo simulation was done with 10000 runs. When available, the uncertainty distribution of the input data was directly retrieved from the source. When not, the distribution was built using the pedigree matrix approach (Ciroth et al., 2016; Weidema, 1998; Weidema and Wesnaes, 1997), using default uncertainties (U_i) and base uncertainties (U_b). The pedigree matrix, default, and base uncertainties used in this study were retrieved from (Ciroth et al., 2016). The total uncertainty, expressed as the square of the geometric standard deviation (σ_g) was calculated following Equation 1 (Frischknecht et al., 2005):

$$\sigma_g^2 = e^{\sqrt{(\ln U_1)^2+(\ln U_2)^2+(\ln U_3)^3+(\ln U_4)^4+(\ln U_5)^2+(\ln U_b)^2}} \quad (1)$$

2.5. Sensitivity analysis

The parameters for which sensitivity has been tested for MOW-FS are shown in **Error! Reference source not found.**. The nitrogen content in municipal organic waste varies quite a lot. Syauqi et al. (2013) assessed the nitrogen content in MOW in Indonesia and found values between 0.544 and 2.249 %. Since for MOW-FS 1.52% was assumed (See Table 3), the sensitivity to lower (0.54%) and higher (2.25%) nitrogen contents was tested.

In MOW-FS and Manure-FS it was assumed that no methane emissions occurred during digestate application, due to aerobic conditions. In spite of that, some methane emissions might occur (Czubaszek and Wysocka-Czubaszek, 2017). This effect was included in a scenario were the methane emissions from digestate application of MOW-FS were approximated with the IPCC equations for the daily spread of manure (IPCC, 2006b).

Especially for the digestion of MOW, electricity consumption contributes considerably to the total energy input of a digestion plant (Berglund and Börjesson, 2006). Given electricity production is likely to become more renewable in the coming years, its impact on the results was tested using the electricity mix of Sweden (SE), which has the highest share (56%) of renewables sources in the EU

(Eurostat, 2020) instead of the EU mix. The use of more efficient plants was also modelled, assuming a 10% reduction of the electricity consumption for the digestion process.

Table 3: Overview of analyzed parameters in the sensitivity analysis of MOW-FS.

Sensitivity scenario	**Explanation**
High N content	MOW has a nitrogen content of 2.25 % instead of 1.52%
Low N content	MOW has a nitrogen content of 0.54 % instead of 1.52%
Residual methane emissions digestate	Digestate application leads to direct methane emissions at the field
Efficient digester	Digester uses 10% less electricity
SE electricity	Instead of the EU electricity mix, the Swedish mix is used as an example of a more renewable mix

Table 4 gives an overview of the parameters for which sensitivity has been tested in Manure-FS. The sensitivity analysis was performed for the impact categories climate change, acidification, and terrestrial eutrophication, and not for freshwater eutrophication since the investigated parameters did not have impact on it.

In the base scenario, it is assumed that the availability of nutrients is the same, i.e., 1 kg N in digestate or manure can replace 1 kg N in synthetic fertilizer. This is probably true for digestate, as most of the nitrogen is directly available for plants. However, in manure, most of the nitrogen is still incorporated in organic matter and mineralized only later. Whether this is an advantage or not depends on the nutrient requirements of the crop, the soil composition, and the timing of fertilizing. The case in which manure has a lower fertilizing capacity than digestate or synthetic fertilizer, as proved by pot experiments (Möller and Müller, 2012), is tested by assuming that 1 kg of N in manure replaces 0.9 kg synthetic N.

The pH of digestate is generally 0.5-2 units higher than the pH of the undigested material. Several processes are responsible for this increase, among which $(NH_4)_2CO_3$ precipitation and CO_2 removal during the digestion process are most important (Möller and Müller, 2012). The rise in pH leads to more NH_3 emissions as the equilibrium between NH_4^+ and NH_3 shifts. To test the effect that increased pH has on NH_3 emissions, it is assumed that the application of digestate gives rise to 10% more NH_3 emissions than the manure utilization (Stuckl et al., 2011).

The potential ammonia oxidation rate (PAO) is an indication for the N_2O and NO_3^- emissions after land application. Two different Swedish studies show that the PAO for digestates from cow and pig manures is higher than for manures (Odlare et al., 2008; Risberg et al., 2017). The hypothesis is that PAO is negatively related to volatile fatty acids (Risberg et al., 2017). The PAO effect is amplified by the fact that digestate contains more free ammonia (NH_4-N), as organic molecules containing nitrogen have been mineralized by the microorganisms (UNITO, 2014). These results are confirmed by the fact that several studies show that the application of digestate leads to higher N_2O emissions. However, some other studies report no effect or the opposite result. It is assumed that soil type plays an important role in this contradiction (UNITO, 2014). To investigate the possible impact of higher N_2O emissions, it is assumed that fertilizing with digestate leads to 10% more N_2O emissions than is the case with manure.

In the base scenario it is assumed that 30% of manure storage occurs in the open air. The most optimistic situation where storage emissions are prevented is also studied. Here it is assumed that all storage is closed and that no methane is released during storage. Of course, even in closed storage, some emissions will occur, for instance, due to ventilating systems. These emissions can be further reduced by combining closed storage with other manure management techniques, such as acidification and thermal drying.

Transport of manure is included in the anaerobic digestion process, and it has previously been identified as a sensitive parameter in the life cycle of biogas production from manure (ADEME, 2011; Esteves et al., 2019). We have tested this parameter by assuming a transport distance of 30 km instead of 15 km.

Table 4: Overview of analyzed parameters in the sensitivity analysis of Manure-FS.

Sensitivity scenario	Explanation
Lower bioavailability manure	1 kg of N in manure replaces 0.9 kg of synthetic fertilizer N instead of 1 kg, due to lower bioavailability.
Higher pH	Due to the shift in NH_4^+/NH_3 equilibrium, 10% more NH_3 emissions occur when digestate is applied.
More N_2O	N_2O emissions due to digestate application are 10% higher.
Closed storage	No emissions occur at the manure storage stage due to closed storage and other management techniques.
Transport distance doubled	Manure is being transported from the farm to the digester over 30 km instead of 15 km.

3. Results and discussion

3.1. Climate change

Figure 3 summarizes the results for climate change due to CBG production from MOW and manure. The numerical values can be found in Table B 1 in appendix B. GHG emissions savings (expressed as percentages) compared to the GHG balance of fossil diesel production and combustion according to RED are shown, as well as the GHG emissions savings limit required to contribute towards the Union targets to be eligible for financial support.

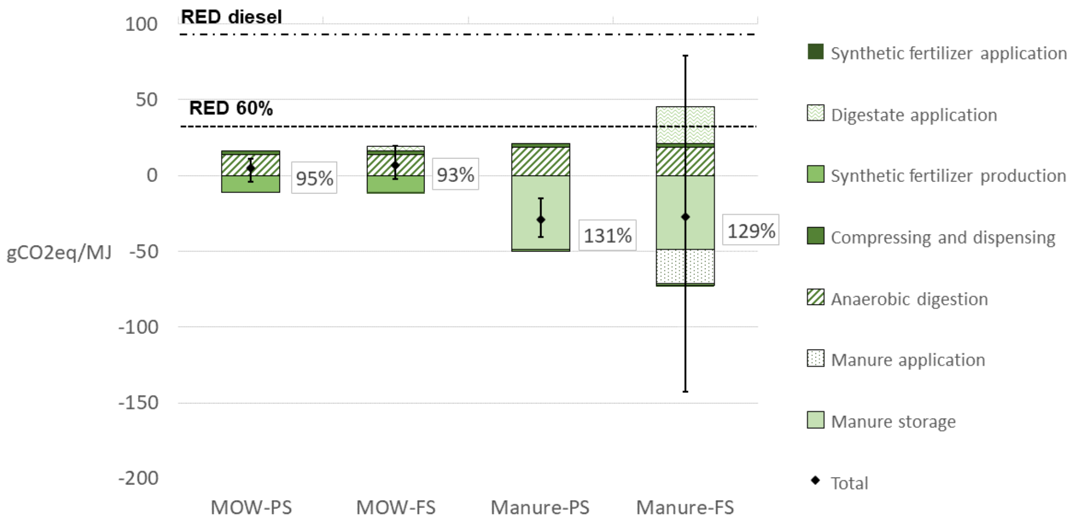

Figure 3: Well-to-Tank (WTT) GHG balance for compressed biogas from municipal organic waste (MOW) with partial (MOW-PS) and full substitution (MOW-FS) method and from manure with partial (Manure-PS) and full substitution (Manure-FS) method. Error bars represent the 95% confidence interval of the results. Percentages show the impact reduction compared to 1 MJ fossil diesel (94 gCO_2eq/MJ, from RED). The lower horizontal dotted line indicates the GHG emissions saving thresholds for eligibility towards financial support in RED, i.e., more than 60% savings compared to fossil diesel ("Council directive 2018/2001", 2018).

The GHG emissions savings of CBG in comparison to fossil diesel calculated in this study are 93% and 131% for MOW-PS and Manure-FS, and 93% and 129% for MOW-FS and Manure-FS, respectively. Thus, both pathways lead to a substantial improvement from a carbon footprint perspective compared to fossil diesel, regardless the substitution method chosen.

For CBG from MOW, the main contribution comes from Anaerobic digestion and Synthetic fertilizer production. Compression and dispensing contribute relatively little to climate change, in coherence with other works (ADEME, 2011; Börjesson et al., 2015; JEC, 2020). Our results for anaerobic digestion are valid for relatively new plants with closed storage of digestate, because we assumed no fugitive methane emissions during anaerobic digestion, as experimentally determined by Liebetrau et al. (2010). In contrast, Tufvesson et al. (2013) mention that in older digestion plants, leakages can be higher (1.5-2.5%). Various studies (Buratti et al., 2013; Evangelisti et al., 2014; Tufvesson et al., 2013) show that these older plants fugitive methane emissions represent up to 20% of the total climate change impact of biogas. Note that methane leakage during compression and dispensing have been included in this study, but they turned out negligible.

Manure scores better than MOW and leads to negative emissions due to the avoided emissions that come with manure storage, as also found in previous works (Bacenetti et al., 2016; Esteves et al., 2019; Hijazi et al., 2016). Synthetic fertilizer production and Synthetic fertilizer application displacement have a minor impact on Manure-FS, as only a small amount of synthetic fertilizers is displaced due to the prevention of manure storage emissions. The largest contribution to the climate change impact of Manure-FS is Digestate application, but this effect is compensated by Manure application. It can concluded that for the impact category climate change, the assumption from JEC that manure spreading and digestate spreading have a similar impact is justified. However, if a fraction of the nutrients would be lost at the digestion plant, the climate change impact for Manure-FS could increase. This would, for example, be the case if digestate is stored in open containers, as discussed by Fusi et al. (2016) and Bacenetti et al. (2016).

Comparison of MOW-FS and Manure-FS shows that the impact of Synthetic fertilizer production displacement is much higher for MOW-FS, as more nutrients are being displaced in the latter scenario. In contrast, Digestate application in MOW-FS has a lower impact because MOW has a higher organic carbon content than manure. Per MJ of produced fuel, less nitrogen is involved, leading to lower N_2O emissions at the field. Therefore, improving biofertilizer application techniques is of high importance for feedstock with a high N to C ratio, provided the goal is to maximize the climate change mitigation benefits of CBG. In general, synthetic fertilizer application has a relatively small effect on climate change and the avoided GHG emissions related to synthetic fertilizer production outweigh the higher N_2O emissions of biofertilizer. This is in agreement with other works (Boulamanti et al., 2013; González-García et al., 2013; Timonen et al., 2019). Nevertheless, Hartmann (2006) found that broadcast application of digestate gives the opposite result. This can be explained by their assumption of a pessimistic situation with a very high ammonia volatilization factor of 90% (*vs*. 20% in this study). As part of the ammonia leads to N_2O emissions in the atmosphere, this also results in much higher N_2O emissions that offset the emissions savings of synthetic fertilizer production displacement.

The change of system boundary has a small effect on the net climate change impact of biogas and the results with partial substitution are very close to the ones with full substitution. However, the uncertainty increases drastically using the latter method due to the inherent uncertainty of fertilizing emissions and the equations used for calculating them. These equations were designed by IPCC to calculate the average emissions of fertilizing practices on a national level and are not fit to represent a specific field. For an individual situation, the emissions can therefore be much lower or higher than those calculated in Manure-FS, depending on temperature, soil type, crop species, and agricultural techniques applied. This high uncertainty also means that RED compliance is not necessarily ensured for each digestion plant, whereas calculation with the partial substitution method would suggest this. The same is true for the current RED methodology that does not allocate any burden to digestate generation, and thus it does not take into account fertilizing emissions. We can conclude that, on average, biogas can already now lead to high impact reduction but, on an individual scale, significant improvements are still possible and are necessary to maximize biogas climate change mitigation benefits. For example, anaerobic digestion could be accompanied by other recovery techniques, such as solid-liquid separation, pelletizing and ammonia stripping with subsequent ammonium salt recovery (Henning Lyngsø et al., 2011), yielding concentrated and mineralized fertilizers that cause less field emissions. Also, fertilizing by trail hose instead of broadcaster and directly incorporating the applied biofertilizer can further decrease N_2O emissions (Hartmann, 2006). Therefore, it can be expected that, in the near future, the average GHG balance for CBG will continue to improve, provided that correct incentives are given by policy-makers, which is not the case with current RED methodology for GHG emissions calculations, as it does not take into account the application of biofertilizers.

Error! Reference source not found. Table 5 compares the results of Manure-PS and MOW-PS to the typical savings calculated by RED and the savings calculated by JEC. As referred, RED does not use substitution, but energy allocation. This is a first cause for the divergence of RED with ours and JEC results. Another part of the difference can be explained by the use of updated GWP values in our study, whereas RED and JEC applied old GWP values. Although there are no differences for N_2O and CO_2, there is a +21% increase for CH_4. This affects especially the results for Manure storage, as methane emissions dominate over carbon dioxide emissions in this process (see **Error! Reference source not found.**). Based on the contribution analysis of the GHG balance of scenario Manure-PS (**Error! Reference source not found.**) and the discussed GWP values, the lower GHG emissions savings from our work compared to JEC and RED can partly be explained. Additionally, different credits were given for manure storage in the three studies. JEC assume a credit of 1.5g CH_4/MJ_{manure}, which corresponds to 115g CO_2eq/MJ_{CBG} using the latest GWP values. RED assumes 45g CO_2eq/MJ_{manure}, which corresponds to 101g CO_2eq/MJ_{CBG}, whereas this study calculated a credit of 49g CO_2eq/MJ_{CBG}. RED and JEC don't justify their assumptions, but possibly more open storage was presumed. For MOW-PS, an inverse trend can be observed. Our work shows the biggest savings, followed by JEC and RED. This is the result of different assumptions on fertilizer displacement. RED does not give a credit for synthetic fertilizer displacement, hence the lowest GHG emissions savings. JEC attributed a credit for nitrogen fertilizer only, whereas this study also gives a credit for phosphorus and potassium fertilizer. Synthetic fertilizer displacement has a positive effect on the climate change impact reduction of biogas production.

Table 5: Comparison of the results on climate change with RED ("Council directive 2018/2001",2018) and JEC (2020) GHG emissions savings compared to fossil diesel. Typical values represent an estimate for the average European consumption of a specific biofuel.

	Manure-PS/MOW-PS	**RED typical**	**JEC**
Biogas of wet manure	131%	190%	209%
Biogas of MOW	95%	70%	90%

The benchmark for the GWP impact of diesel production used to calculate the GHG emissions savings was taken from RED, and was therefore calculated with energy allocation. The validity of this figure and its comparability with our results using a different allocation method, may therefore be questionable. Nevertheless, it is common practice to calculate GHG emissions savings this way (Börjesson et al., 2010; Börjesson et al., 2015; Tufvesson et al., 2013). The reason is that the use of a common benchmark enhances comparability between studies, and this is sometimes preferred over methodological consistency. The bias due to this inconsistency is generally small. In fact, if we would have used a methodologically consistent benchmark, for example the result from JEC, i.e., 88.6 gCO_2eq/MJ, the GHG emissions savings for Manure-FS and MOW-FS would have been 131% and 92%, respectively. These results are very close to the ones obtained with the RED benchmark and within the defined error range.

A limitation of our study is the carbon neutrality assumption and the neglection of land use change emissions. For manure, the carbon stored in the manure comes from the feed given to the animals and for MOW the carbon originates from garden wastes and food left overs. For some of the feed and food, the carbon neutrality assumption or the zero land use change emissions might not be justified. For co-products with economic value, the additional emissions would have to be allocated between the main product and the co-product (Wiloso et al., 2016). In our work, manure and MOW generation were considered burden-free, therefore, it is consistent to also disregard these emissions. Due to the complex relation between the CO_2 in manure and MOW and the original products, it is not possible to anticipate the potential impact of including a full carbon inventory without a thorough study. This is out of the scope of the present work, but future research could investigate this effect.

3.2. Acidification

This part studies the impact of the different scenarios on the acidification potential. The results are shown in Figure 4 (the numerical values can be found in Table B **3** in appendix B).

CBG from MOW leads to more acidifying emissions than CBG from manure. This is in apparent contradiction to the results of Borjesson et al. (2010), who assumed no ammonia emissions during manure storage and no credit for not using manure as a fertilizer. If these credits were omitted in our work, CBG from manure would also have a higher acidification impact than MOW, due to the higher N to C ratio. The differences are thus only due to different methodological choices.

The contribution of Anaerobic digestion, Compression and dispensing, and Synthetic fertilizer production is negligible for both scenarios, but the choice of system boundaries has a sensible effect. The acidification potential is higher when the full substitution method is used because the digestate fertilizer application leads to high ammonia emissions. This has important implications: if partial substitution is used, MOW seems to lead to impact reduction compared to diesel, but if full substitution is used, MOW leads to higher emissions compared to diesel. If the acidification impact were included in the RED sustainability criteria, the system boundary would have to be standardized to prevent such contradicting results.

Manure leads to a high impact reduction compared to fossil diesel, caused by the prevented manure storage emissions. The Manure-FS results are again extremely uncertain due to the high amount of nutrients involved in the life cycle and the broad range of fertilizing emissions on an individual level. For climate change, being a site-generic impact, especially the average outcome is important. But acidification and eutrophication, among others, are site-specific impacts, and they occur close to the point of emission of ammonia (European Commission, 2019). For example, low acidification in one region does not necessarily compensate for high acidification in another region, which is why individual results are more meaningful than average results. Therefore, development of correct application techniques for biofertilizer is crucial to reduce ecosystem acidification. A reduction in ammonia volatilization can be reached when the application is performed by trail hose and directly followed by incorporating the biofertilizer in the soil (Hartmann, 2006).

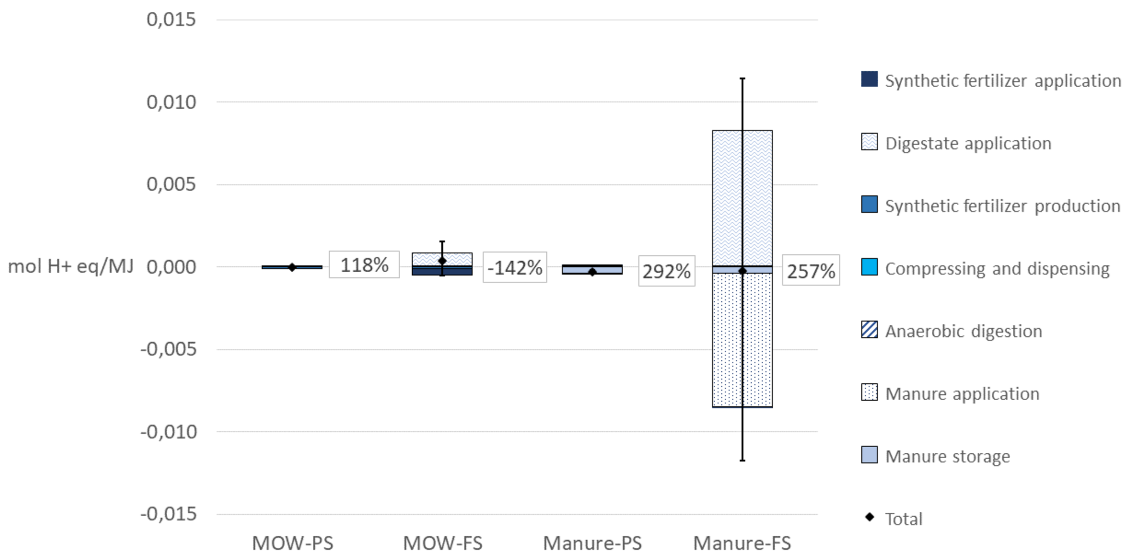

Figure 4: Well-to-Tank (WTT) acidification impact for compressed biogas from municipal organic waste (MOW) with partial (MOW-PS) and full substitution (MOW-FS) method and from manure with partial (Manure-PS) and full substitution (Manure-FS) method. Error bars represent the 95% confidence interval of the results. Percentages show the impact reduction compared to 1 MJ fossil diesel (1.5E-4 mol H^+ eq/MJ, EF 3.0 method on dataset "market for diesel, low-sulfur" (Jungbluth, 2018)).

3.3. Terrestrial eutrophication

The results for terrestrial eutrophication are shown in Figure 5 and the numerical values can be found in Table B **4** in appendix B. The same trends seen for acidification (Figure 4) can be observed, being both impacts mostly caused by ammonia emission. The main difference is that, for terrestrial eutrophication, the results for all scenarios diverge more from fossil diesel in relative terms. This can partly be explained by the fact that diesel production leads to SOx emissions, which contribute about 80% to its acidification impact, but not to terrestrial eutrophication. The diesel comparator is, therefore, larger for acidification than for terrestrial eutrophication. We stress that the relative difference with diesel would be different for terrestrial eutrophication and acidification if combustion was included in the system boundaries due to tailpipe NOx emissions. Even so, from a well-to-wheel perspective, CBG from manure would still perform better than diesel and diesel would perform better than CBG from MOW. For trucks, the NOx emissions of vehicles on CBG are similar to those on conventional diesel (European Federation for Transport and Environment, 2018). For passenger cars, natural gas vehicles have proven to perform slightly better in terms of NOx emissions than EURO 6 diesel cars. Still, the Euro 7 requirements that are currently being developed will probably eliminate this benefit. (European Federation for Transport and Environment, 2018).

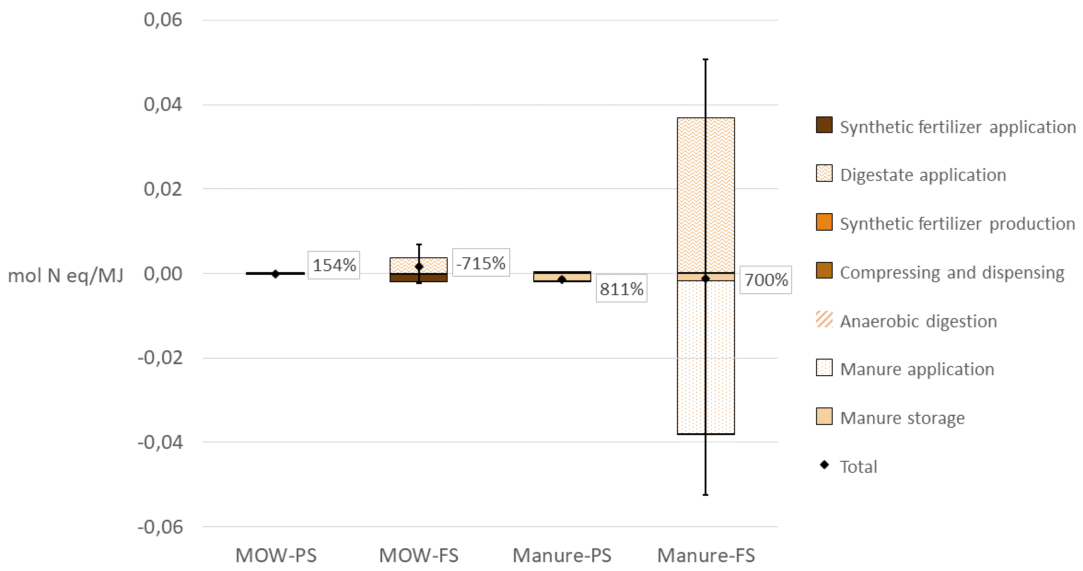

Figure 5: Well-to-Tank (WTT) terrestrial eutrophication impact for compressed biogas from municipal organic waste (MOW) with partial (MOW-PS) and full substitution (MOW-FS) method and from manure with partial (Manure-PS) and full substitution (Manure-FS) method. Error bars represent the 95% confidence interval of the results. Percentages show the impact reduction compared to 1 MJ fossil diesel (2.1E-4 mol N eq/MJ, EF 3.0 method on dataset "market for diesel, low-sulfur" (Jungbluth, 2018)).

3.4. Freshwater eutrophication

All scenarios lead to a higher impact than fossil diesel, as displayed in Figure 6 and Table B 5 in appendix B. The main contributor in all scenarios except Manure-FS is the electricity use in Anaerobic digestion and Compression & dispensing. In counter to what was observed earlier for climate change, acidification, and terrestrial eutrophication, the full substitution method leads to more favorable result than thepartial substitution method, although the difference is very small. This is due to the displaced synthetic fertilizers. In contrast to nitrogen, phosphorus emissions are larger for synthetic fertilizers than for biofertilizers. Manure-FS leads to high uncertainty and potentially to high phosphorus emissions. The 97.5 percentile corresponds to 3.5E-5, i.e., 3000% increase relative to fossil diesel. This could be the result for Manure-FS if phosphorus precipitates in the digester instead of ending up in the digestate. Digestate application would then displace less synthetic fertilizer than manure would have, resulting in a net increase of synthetic fertilizer production and application. Limiting phosphorus losses in the digester is crucial to obtaining the calculated impact reduction for CBG from manure.

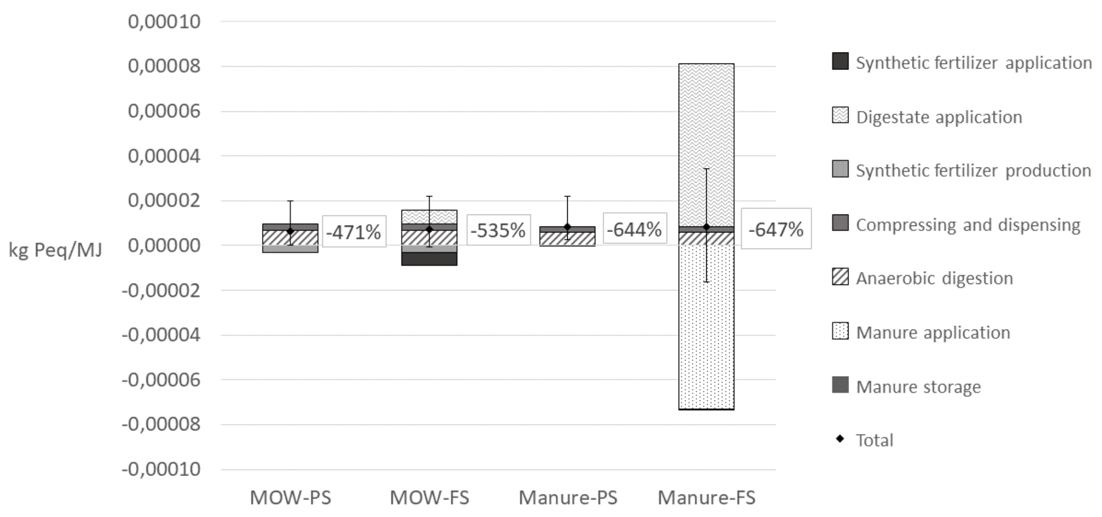

Figure 6: Well-to-Tank (WTT) freshwater eutrophication impact compressed biogas from municipal organic waste (MOW) with partial (MOW-PS) and full substitution (MOW-FS) method and from manure with partial (Manure-PS) and full substitution (Manure-FS) method. Error bars represent the 95% confidence interval of the results. Percentages show the impact reduction compared to 1 MJ fossil diesel (1.1E-6 kg P eq/MJ, EF 3.0 methodon dataset "market for diesel, low-sulfur" (Jungbluth, 2018)).

3.5. Sensitivity analysis

The results of the sensitivity analysis for MOW-FS is summarized in**Error! Reference source not found.** Figure 7 (the numerical values can be found in Table B 7 in appendix B). When considering residual methane emissions due to the spreading of digestate, the climate change impact increases about 1%. Their neglection in Manure-FS is thus justified. When comparing MOW with a varying nitrogen content, it becomes clear that the higher the nutrients in the biomass is , the larger the acidification and eutrophication impact. It is thus essential to apply correct fertilizing practices to limit these unwanted impacts. Despite this, a high nutrient content also means a high potential for synthetic fertilizer displacement and hence, an opportunity for further GHG emissions savings.

A digester that uses 10% less electricity leads to a proportional 10% decrease in GHG emissions. This highlights the relative importance of electricity consumption in determining the impact on climate change. When the Swedish electricity mix is used, the total climate change impact decreases by 32%. This happens because when low carbon electricity is used, the climate change impact of producing fertilizer is sensibly reduced. Since the share of renewable in the EU energy mix will increase in the future (Capros et al., 2016). Also the performance of MOW-based biogas production in terms of GWP impact will improve. On the contrary, the effect of electricity on acidification and terrestrial eutrophication is negligible.

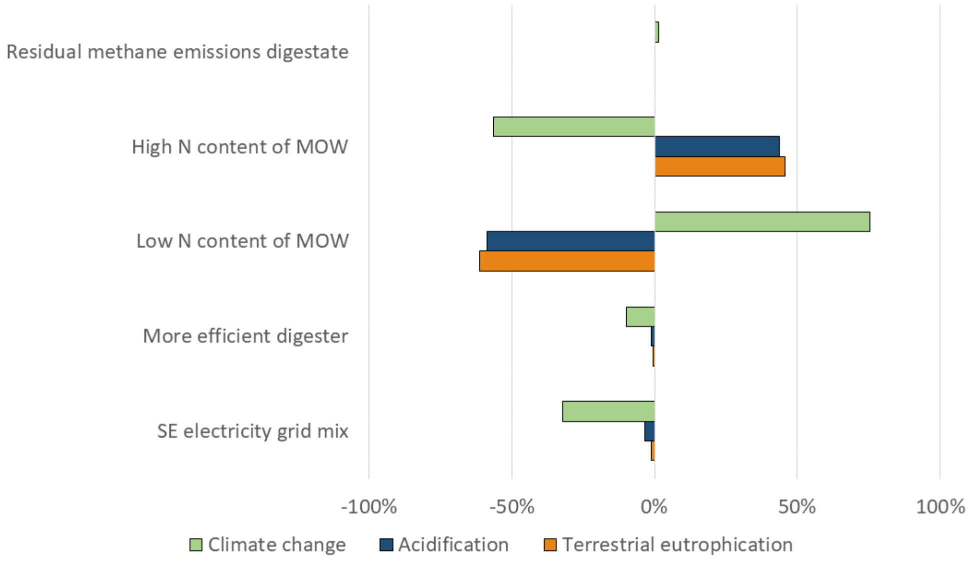

Figure 7: Results of the sensitivity analysis on MOW-FS. Percentages express deviation from MOW-FS.

Figure 8 displays the results of the sensitivity analysis on Manure-FS. The numerical values can be found in Table B 6 in appendix B. A 10% lower bioavailability of nitrogen in manure results in a 39% lower climate change impact, a 191 % lower acidification, and a -162% lower terrestrial eutrophication impact due to a higher net synthetic fertilizer production and application displacement. This high sensitivity has important implications because it means that the impacts of digestate spreading and manure spreading do not necessarily cancel each other out, as often assumed. The increasing bioavailability of nitrogen in digestate can have an overall positive effect provided care is taken to decrease the ammonia volatilization during spreading. For farmers this means that good agricultural practices for spreading of digestate are crucial. For the LCA community it means that the assumption that digestate and manure fully substitute each other is not a correct representation of reality. Therefore, we recommend either using the full substitution

method, or using the partial substitution method with an additional credit for different fertilizing quality of digestate.
Increased pH in the digestate leads to a 300% increase of acidification and terrestrial eutrophication, due to higher NH_3 emissions. There is also a moderate increase of dinitrogen monoxide emissions, leading to a 8% higher climate change impact. This is another example of a difference between manure and digestate that cannot be neglected. Directly incorporating digestate in the soil after application is important to decrease the ammonia volatilization. The application of moderately alkaline digestate can also affect soil quality that have not been assessed quantitatively in our study, but which are important to mention. Increasing the alkalinity of soils can have deleterious effects on plant growth as, for most plants, soils with pH above 7 represents a suboptimal condition (IGBP-DIS, 1998). Most European soils are moderately acidic (Reuter et al., 2008) so this effect is of relative importance in this region, but in many other regions of the world, with neutral or slightly alkaline soils, usage of manure can be preferred over digestate.
A 10% increase in N_2O emissions after digestate application due to a higher PAO rate results in an almost proportional raise of GHG emissions by 8%. N_2O emissions are clearly important, but not the most important contribution to climate change in this model. As more ammonia is converted to dinitrogen monoxide, less terrestrial eutrophication and acidification occur, though the effect is very small (-3%).
When emissions from manure storage are prevented, e.g. due to closed storage, the biggest total impact on results can be observed. In this case, Manure-FS would still comply with the RED limit, but its climate change impact would increase with over 160%, becoming also twice as large as the one of MOW-FS. As the occurrence of emissions from manure storage is a rather sensitive parameter, it is important to evaluate the justification of including the avoided impact of this process. It is currently justified for some parts of Europe, where the nitrogen and phosphorus in the produced manure is lower than the amount of synthetic fertilizers applied in these regions. However, in many regions in North-Western Europe, the amount of manure nitrogen and phosphorus exceeds the fertilizer needs (Potter and Ramankutty, 2011). In this case, part of the manure is a waste stream that must be disposed, and the storage of manure at the farm and the application to land should not be considered business-as-usual for all of the produced manure. The avoided impact for manure storage should only be considered for the fraction of manure that is effectively applied to land. Also, it is important to note that our model is a simplification of reality, as it is assumed that the only alternative management of manure is storage and spreading. This is a limitation, because other manure management techniques, such as acidification, solid-liquid separation and thermal drying can also be used, although none of these techniques is currently being applied on a large scale (Bernal et al., 2015). Our work investigates the special link between the

prevention of manure storage and the impact of CBG, but for a more comprehensive insight on the effects of diverting manure, or any waste or residue, a consequential LCA is required.

The last parameter is the transport distance. Doubling it leads to a 10% decrease in climate change impact and it has virtually no effect on acidification and eutrophication. The transport distance in our work was estimated based on a livestock unit density of 0.98 livestock units/ha. However, in countries with lower densities, like France, Italy, Poland and others, transport distances are probably higher than in our Manure-FS. Based on our sensitivity analysis, this larger distance is acceptable albeit to a certain extend. Manure-FS would not reach the RED target anymore when the transport distance of manure exceeds 340 km. Poeschl et al. (2012) studied this more in-depth, including also other impact categories. They concluded that the distance should not exceed 64 km to have a net environmental benefit compared to fossil fuels. They did not mention any credit for manure storage, which explains the much lower transport distance limit.

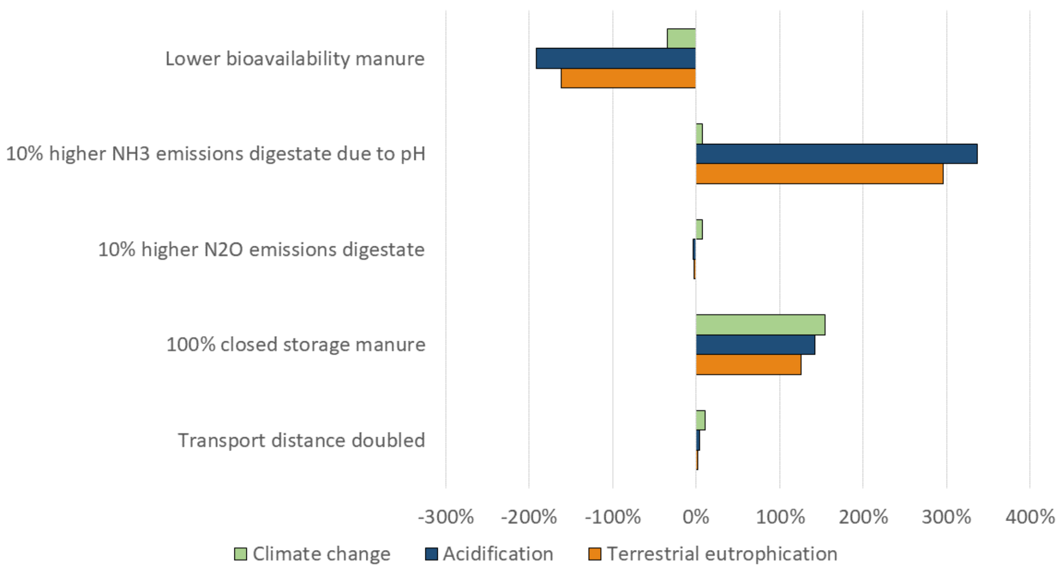

Figure 8: Results of the sensitivity analysis on Manure-FS. Percentages express deviation from Manure-FS.

Our results show a very high uncertainty, mainly driven by the high error range of the IPCC emission factors used. Using these factors is not necessarily wrong, but what is often lacking is a correct interpretation of the results. The IPCC equations are made to estimate average national fertilizing emissions and, in principle, should not be used for LCAs on individual plants. When done, it is important to show the 95% confidence interval and discuss its implications as it was done correctly in this work. Of course, there is also uncertainty on this interval, as not all the uncertainty distributions are known. Still, an estimation of the order of magnitude allows the reader to put the results in perspective. Furthermore, notwithstanding the well-known difficulties of acquiring original data, we again stress the need to use as much as possible primary data on field emissions to obtain more robust results.

Regionality is also an important aspect of biogas LCA, especially for the aforementioned site-specific impact categories. For example, in regions where more manure than can be applied to agricultural fields, is produced such as the Netherlands (CBS et al., 2020), it is not justified to assume that manure is a by-product used as fertilizer as it is BAU to dispose of part of the manure. Giving a credit for the avoidance of manure storage does not make sense in this case, at least not for the fraction of manure that can be considered a waste. This has a significant effect on the outcome. While some authors suggested defining clear guidelines on the classification of materials, i.e., specify for each feedstock and product whether it is a waste, a by-product or a co-product (Manninen et al., 2013). We instead recommend developing clear guidelines on the system boundaries. The classification of materials can differ between regions and change over time. For example, MOW could become a resource for animal feed production in the near future (Dou et al., 2018), in which case it is no longer a waste stream. These differences should be taken into account in LCAs and biofuel policy. Comparability among studies could be achieved by standardizing the system boundaries and substitution method for each application of biogas. For CBG production from waste streams and by-products, we recommend using our system boundaries with the full substitution method. In other words, the generation of waste and by-product feedstock should be left out of the system boundaries, but the avoided impact for the alternative use of by-products should be taken into account. The digestate can be considered a waste or a by-product, but its disposal or usage as fertilizer should be included in the system boundaries of CBG, because the demand for anaerobic digestion is driven by biofuel regulations and not by the need for biofertilizer. This is particularly important for CBG from manure and MOW, as the current RED regulations subsidize these fuels (since they comply with the emissions savings target), causing their production and the generation of digestate to increase.

Although we recommend the use of the full substitution method, the results of our study should be used with caution, since the combination of substitution and attributional modelling can be misleading. The goal of this study was to investigate the specific interaction between the production of CBG and the production and use of biofertilizers (manure and digestate). This justifies the means, since substitution "is also applicable for attributional modelling that is interested to include existing interactions with other systems" (European Commission et al., 2010). But a full consequential LCA is required to assess the large-scale effect of the projected increased production of CBG in Europe and its link to the use of biofertilizers.

To conclude, we listed some remarks on the methodology of RED. Concerning the use of energy allocation RED states: "it is easy to apply, is predictable over time, minimizes counter-productive incentives and produces results that are generally comparable with those produced by the substitution method". These are all valid reasons, but the last remark is not always true. To close the gap between the results obtained via the substitution method and via the energy allocation method, fixed credits could be defined for digestate application, e.g. the credit for manure storage. Even better would it be if RED were to adopt the substitution method. As discussed above, we recommend making clear guidelines on the system boundaries in that case. For calculating typical and default values in RED, we advise making different calculations for the case where the feedstock is a waste and the case where the feedstock is a by-product. Each operator or national authority should decide which case applies more to them.

Also, considering that the goal of this directive is climate change mitigation, it is in principle obvious that only GHG emissions are calculated. Nevertheless, acidification and eutrophication are very important impact categories when it comes to anaerobic digestion. In our opinion, they should also be included in the RED sustainability criteria to ensure that encouraging anaerobic digestion does not lead to unsustainable nutrient management and unwanted negative side-effects. Particulate matter formation was not assessed in our study, but it is potentially also an important impact category for CBG, since ammonia emissions contribute to it. Future research should investigate particulate matter formation to determine its impact and to what extend it should be added to the RED sustainability criteria.

Currently, RED requires economic operators and national authorities to monitor and report on the soil quality and soil carbon when biomass is used for biofuel production, without further specifying the definition of "soil quality". To account for acidification and eutrophication, this requirement should not only apply to the soils from which biomass is harvested but also to soils to which biofertilizer is applied. Besides, the definition of soil quality could be better defined and broadened. The current definition only consideres the soil carbon content, whereas also the nutrient content and pH should be taken into account. At last, volatilizing emissions from the application of biofertilizers should be monitored as well, as they cause acidification and terrestrial eutrophication.

4. Conclusions

The production of CBG can lower impacts on climate change potentially, when well-managed, below the reduction target for biofuels set by the RED. But if the goal is to assess the sustainability of a biofuel, solely looking at its climate change impact is not sufficient. Acidification and terrestrial eutrophication were found to be environmental burdens of CBG from MOW when the full substitution method was used, but also the most important environmental benefits for CBG from manure. This apparent contradiction was explained by the avoided impact of manure storage. In the case were considering this avoided impact is not acceptable, CBG from manure has larger acidification and terrestrial eutrophication burdens than CBG from MOW. Thus, the main take-home messages are that better manure-storage techniques are required to prevent nutrient losses in the process as much as possible, and that field emissions due to biofertilizer application should be reduced. Freshwater eutrophication is also an environmental burden of CBG, although mainly caused by the electricity use of the digester, and not so much by digestate application, as virtually no phosphorus losses occur. This study also demonstrated that using the full substitution method, including the digestate fertilizing emissions in the system boundaries of the LCA of CBG production, has a significant impact on the outcome. It was demonstrated that the impact of digestate spreading and manure spreading is different and should not be considered equal.Furthermore, a high error range on the results exists caused by the use of IPCC emission factors for the quantification of fertilizing emissions, and our study shows significant environmental risks linked to the production of CBG. Different potential scenarios exist, for instance, caused by the uncertainty of fertilizing emissions, in which the climate change impact of CBG approach the one of fossil diesel or in which the acidification and eutrophication impacts largely exceed the one of diesel. Hence there is still a need for improvement and good agricultural practices are one aspect that may lead to better performance. CBG can be part of the solution to climate change mitigation, but constant monitoring of the real systematic life cycle emissions remains pertinent to lower environmental risks.

To summarize, active and long-term promotion of biogas production by the EU can mitigate climate change. However, EU regulations should include the acidification and eutrophication potential of biofuel production in their sustainability criteria to incentivize better fertilizer application practices and innovative nutrient recovery techniques.

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