Supplementary material - Appendix

Linking emissions, anaerobic digestion and flows of manure and biomass

In this conceptual model CH_4 emissions are affected by amounts of slurry organic matter (volatile solids, VS), temperature and storage time and VS in deep litter in the building or outside store. Transformation of VS to gases in manure during storage in animal buildings, stored outside and in AD are calculated. Included in the calculation of N losses are transformation of N between organic and inorganic pools during storage and in AD and gaseous emissions of NH_3 are related to the transformation of organic acids and changes in the concentration of total inorganic carbon. Nitrous oxide emission is affected of the changes in the pools of N in the biomass and related to degradable organic matter concentration. Therefore the calculation includes all these changes in the different compartments of the biomass management chain and how changes in one compartment may affect transformations and emissions downstream the compartment.

1. Biogas production

The CH_4 produced is determined for each biomass, and through modelling using the Gompertz equation, the gas potential at a given time is calculated as follows:

$$M(t) = BMP \cdot (1 - e^{-k \cdot t}) \quad \text{(eq. 1)}$$

where M is the cumulative CH_4 yield (ml g^{-1} VS), BMP is the theoretical CH_4 yield (ml g^{-1} VS), k is a first-order kinetic rate constant, and t is time in days. The biogas production are related to biomass and retention time in the biogas reactor (table 1).

The theoretical gas potential states the theoretical CH_4 yield upon complete conversion of all organic matter and is used to determine the share of the organic matter, which is degraded in the biogas process. The theoretical gas yield can be calculated if the ratio C:H:N:O:S in the biomass is known, or by determining the content of the main organic components, such as carbohydrate, protein, lipid, VFA, lignin, and glycerol. It will also be possible to estimate the gas yield based on literature data.

Table S1. Assumptions on dry matter content and gas potential of different biomasses used in biogas production. The ultimate CH_4 yield is the yield achieved at a retention time of more than 90 days. The CH_4 yields after 45 and 60 days and ultimate gas yield are based on data from tests at the Foulum biogas plant Aarhus University. Sources; 1) Average of 50 analyses of slurry supplied to two biogas plants, 2) Olesen et al. (2018), 3) Data from Foulum biogas plant, Aarhus University and 4) Data from tests at Foulum biogas plant, Aarhus University.

Biomass	Dry matter,	VS	Total-N	CH_4 yield 30 days	CH_4 yield 60 days	BMP yield	Hydrolysis constant	Source
	% (wb)	%(dm)	N g kg^{-1}	L CH_4 kg^{-1}(VS)			d^{-1}	
Cattle slurry	7.7	80	3.98	230	250	275	0,08	1
Pig slurry	5.4	80	5.67	335	345	350	0,10	1
Cattle deep litter	30.0	80	9.49	263	271	275	0,07	2
Grass ensilage	35.0	95	8.75	324	325	325	0,08	2
Maize ensilage	31.0	95	3.91	325	325	325	0,12	3
Wheat straw	84.0	95	4.24	278	286	290	0,15	2
Slaughterho use waste	15.0	85	3.90	488	490	490	0,07	4
Biowaste	22.5	88	5.20	424	425	425	0,12	4
Glycerol	70.0	95	0.00	450	450	450	0,14	4

Generally, gas yields calculation and measurements are subject to uncertainty, and differ considerably between studies. A large number of factors affect gas yield of manure, including the livestock housing conditions, litter, feed, etc. Moreover, the method used to determine gas yields is a source of uncertainty, and two laboratories may often get different values from the same substrate (Holliger et al. 2021). For cattle slurry, Olesen et al. (2018) state a CH_4 yield of 13.9 m^3 ton^{-1}, but retention time is not mentioned in this study. In this report, the yield is 11.8 m^3 ton^{-1} for 30 days retention time and 14.1 m^3 ton^{-1} for 45 days.

Table S2. Data used for the calculation of energy consumption on a standard Danish biogas plant.

Unit and operation	Assumptions

Tank type	Steel tank
Liquid volume	8000 m^3
Approx. dimension	ø16*H=20 m (H=1.25xD)
Insulation	200 mm mineral wool + trapezoidal sheet
Agitation	Suspended vertical agitator(s)
Average outdoor temperature	8°C
Average biomass temperature (Møller et al. 2019)	15°C
Average temperature of biomass leaving reactor	25°C
Average temperature of biomass leaving biogas plant	20°C
Specific heat capacity for biomass[1]	4.2 kJ K^{-1} kg^{-1}

1 Cp for dry organic matter (straw, wood, etc.) is approx. 1.3 - 2 kJ K-1 kg-1. For water, it is 4.2 kJ K-1 kg-1. As digestate contains about 90-95% water, Cp as in water is assumed for all biomas categories (Møller et al. 2008).

Calculating CO_2 balance due to substitution of fossil fuel and emission from transport

The calculations are based on the displacement of natural gas ie. CH_4 substitute CO_2 corresponding to 0.057 kg MJ^{-1} (Møller et al. 2008). The electricity demand for agitators, pumps, etc. at the plant is assumed to be covered by a mix of the Danish electricity production, which in 2019 is estimated at 0.150 g CO_2 kWh^{-1}.

The transport volume of the biomass transported to and from the biogas plant varies, and diesel consumption is given per kilometre driven (Table S3). The distances are calculated as the average additional transport of biomass compared to the reference scenario without AD.

Table S3. Energy use and CO_2 emission due to transportation of biomass. CO_2 emissions of 2.7 kg per litre of diesel is assumed.

Biomass category	Category	Distance,	Transport	Diesel consumption		CO_2 emission
		km	t $load^{-1}$	km L^{-1}	L ton^{-1}	kg CO_2 t^{-1}
Cattle slurry	Slurry	20	38,000	1.2	0.4	1.2
Pig slurry	Slurry	20	38,000	1.2	0.4	1.2
Deep litter cattle	Solid biomass	30	25,000	2.5	0.5	1.3
Poultry manure	Solid biomass	30	25,000	2.5	0.5	1.3
Grass ensilage	Other biomass	15	20,000	2.5	0.3	0.8
Maize ensilage	Other biomass	15	20,000	2.5	0.3	0.8
Wheat straw	Other biomass	20	15,000	2.8	0.5	1.3
Slaughterhouse waste	Industry	50	25,000	2.5	0.8	2.2
Biowaste	Industry	55	20,000	2.5	1.1	3.0

Glycerol	Industry	300	35,000	2.0	4.3	11.6

2. Principles of calculating methane emissions in reference and biogas scenarios

Slurry and digestate:

Separate calculations of daily methane emissions from barns and outside storage were made for estimating annual methane emissions. Based on Mikkelsen et al. (2016), the average retention time in the housing was set at 20 days for cattle and 19 days for pigs. It was assumed that export for biogas treatment or transfer to a storage tank took place ten times during the year starting 1 June, and with emptying in April.

The temperature of slurry in barns was set at 13.8°C for cattle and 18.6°C for pigs (In the outdoor storage tank, the temperature of untreated slurry and digestate was calculated from monthly mean temperatures as described by Mikkelsen et al. (2016).

The amounts of VS exported from barns, corresponding to the average amount and composition of VS collected, was calculated from registrations on biogas plants. Using total VS exported from the house, or total VS in digestate, as starting value, the amount of remaining VS in the storage tank was then calculated with daily time steps. Estimation of VS degradation assumes that the ratio of methane to the total carbon loss, mainly as CH_4 and CO_2, is known. During anaerobic digestion, methanogenesis is the main degradation pathway, and the methane share is 55-65%. During outside storage the temperature is lower, and the share of methane is often considerably lower than 60%; Petersen et al. (2016) quoted studies reporting values of 10-30%. The present analysis assumed that 25% of the carbon derived from VS degradation in untreated slurry was emitted as methane, and 10% in digestate (Mikkelsen et al., 2016), and the carbon content in VS was assumed to be 0.45 kg $(kg)^{-1}$ (Petersen et al., 2016).

Table S4. The methane production potential values, *lnA'*, for digestate, cattle slurry and pig slurry were calculated based on information extracted from published studies about methane production rate, total VS and temperature. In the table, $\bar{x} \pm s.e.$ refers to average and standard errors.

Slurry type	Month	lnA'	Source
Digestate	March	28.3	Maldaner et al. (2018)
	April	27.4	Elsgaard et al. (2016)
	September	28.2	Maldaner et al. (2018)

	$\bar{x} \pm s.e.$	**27.9 ± 0.4**	
Cattle slurry	March	28.6	Maldaner et al. (2018)
	April	28.2	Husted (1994)
	April	30.1	Elsgaard et al. (2016)
	July	29.3	Husted (1994)
	September	29.6	Maldaner et al. (2018)
	$\bar{x} \pm s.e.$	**29.2 ± 0.1**	
Pig slurry	January	30.4	Sharpe et al. (2002)
	April	31.1	Husted (1994)
	May	29.2	Sharpe et al. (2002)
	July	30.8	Husted (1994)
	August	30.0	Sharpe et al. (2002)
	$\bar{x} \pm s.e.$	**30.3 ± 0.4**	

From AD production of biogas and units for upgrading the gas by reducing CO_2, the CH_4 emission is at about 1.1% (Nielsen, 2019). The emission is low because leakage of biogas has been reduced significantly since the start of 2000. Today, Denmark uses different types of upgrading plants from which CH_4 is leaking (Table 3.5. Amine and water scrubbing plants constitute the majority (86%) of the plants, whereas a smaller number of plants uses membranes (14%). The CH_4 loss varies from 0.05% from amine upgrading (56% of upgrading plants) to 1% emission from water scrubber (30% of upgrading plants). The total loss from upgrading plants in Denmark is estimated to be about 0.3%. (Communication Kvist T. Dansk Gasteknisk Center, 2020). Based on currently available knowledge, a total loss of 1% is estimated for the biogas plant and the upgrading plant as the plant with best practice.

3. NH_3 emission in reference and biogas scenarios

Ammonia is emitted from all stages of the manure or digestate management system, and when deposited on land or in water, some of the NH_3 will be transformed to N_2O and be emitted to the atmosphere, where it will contribute to climate warming. This indirect N_2O emission from N in the form of NH_3 is estimated using the equation (IPCC, 2006):

$$F_{N2O} = F_{NH3}*0.01\ 44/28$$

Where F_{N2O} is given in kg N_2O, F_{NH3} is given in kg NH_3-N emitted and 0.01 is a default factor given by the IPCC for calculation of the climate warming effect of emitted NH_3.

For stored liquid manure NH_3 emission factors (EF_{NH3}) are given as a percentage of total ammoniacal N (TAN=NH_3-N+NH_4^+-N) in the stored slurry. For stored slurry with the same composition the *EF* will differ as affected by depth of the slurry storage. In EMEP/EEA Air Pollutant Emission Inventory Guidebook it is assumed that storage depth are 3 m while in

Denmark it is 4 m, and as emission is related to surface area then the EF will be 25% lower in Denmark due the differences in area to volume ration. For Danish conditions with 4-metre-deep storage tanks the emission factor (Table S5) are, therefore lower. A recent review show a large variation in the measured reduction in NH_3 emission as affected by covering the slurry, due to the differences in composition of slurry, natural surface crust are not always accounted for in the reference to a treatment, scale of study, cover thickness and measuring method (Kupper et al. 2020). We are using the emission factors given by Hansen et al. (2008), which are within the range of measured emissions.
The main part of abattoir waste are stomach and intestinal parts of livestock, and it is assumed that NH_3 emission during storage is similar to emission from pig slurry with a floating cover, i.e., 2.5% of TAN.

Table S5. Ammonia emission factors for stored liquid and solid manure (Hansen et al. 2008) and organic food waste (Pardo et al. 2015).

Category	Cover	% of TAN	% of N-total
Digested slurry	None	27.3	
Digested slurry	Straw, floating clay granules etc.	5.2	
Digested slurry	PVC cover	2.6	
Cattle slurry	None	10.3	
Cattle slurry	Straw, floating clay granules etc.	3.4	
Cattle slurry	PVC cover	1.7	
Pig slurry	None	11.4	
Pig slurry	Straw, floating clay granules etc	2.5	
Pig slurry	PVC cover	1.3	
Cattle deep litter	None		10
Cattle deep litter	PVC cover		8
Organic food waste	None, composting		21

About 20% of Danish slurry is stored in concrete tanks coved by PVC tents and the remaining part covered by floating covers. It is know that on farms delivering slurry to biogas plants concrete tanks are new and on most of these the cover is most probably a PVC tent construction and the rest is covered by floating covers. Using this information we have calculated the following average emission factors for stored liquid manure:

- Digestate: EF = (5,2+2,6)/2 = 3,9% of TAN
- Cattle slurry: EF = (3,4*0,8 + 1,7*0,2) ≈ 3,1% of TAN
- Pig slurry: EF = (2,5*0,8 + 1,3*0,2) ≈ 2,3% of TAN

The SD for the estimated EF (NH_3 in % of TAN) for stored solid manure was large in a recent reviews of NH_3 emission from manure management (Sommer et al. 2019), the emission factor used here is 10% for cattle deep litter which is much lower than the 32% given for stored solid cattle manure in (Sommer et al. 2019). The reason is probably the higher content of straw in the deep litter that contribute to immobilisation of TAN and a reduce NH_3 emission potential. As with deep litter, considerable variation is also seen in NH_3 evaporation in connection with storage or composting of organic waste, depending on the material, composting method, moisture, C:N ratio, addition of straw or other organic material etc. (Pardo et al. 2015). The alternative to AD treatment of abattoir waste is composting. Incineration of organic waste is prohibited in Denmark, so the only hygienic safe alternative treatment to AD is composting. The NH_3 emission factor for composting of source separated organic waste given in table 6 is from the review of Pardo et al. (2015), i.e. 21% of total N and this is the emission factor used here. Ammonia emission factors for applied liquid manure given by Hansen et al. (2008) are used in this study. The factors were calculated using the ALFAM model (Søgaard et al. 2002) using average monthly weather conditions and average slurry compositions. In the past it was assumed (Hansen et al. 2008) that digestate infiltrated the soil faster and more efficiently than untreated slurry due to a lower dry matter content and viscosity. It was assumed that the fast infiltration counterbalanced the higher pH of digestate so that emission from applied digestate was similar to the emission from untreated slurry (Hansen et al. 2008). In the last ten years biomass co-digested with slurry has changed, and more fibrous material in form of straw, deep litter, etc. are added, which has resulted in a higher DM concentration and viscosity and stickiness of digestate. The consequence of which is that NH_3 emission from digestate applied on the soil is higher than that from untreated livestock slurries due to the higher pH and poor infiltration of digestate than of reference slurry (Sommer et al. 2006). In a resent unpublished study NH_3 emission from untreated cattle slurry applied on soil was lower than from digestate produced by AD treatment of the cattle slurry without adding waste or plant biomass. Dry matter concentration of digestate was 1.5% lower than in the untreated cattle slurry, and the pH value was 0.6 units higher. It was observed that the digested slurry was sticky and that this counterbalanced a lower DM causing infiltration into the soil to be slow and the hypothesis is that this caused an increased NH_3 emission potential. The stickiness may be related to transformation of organic particles and formation of larger particles and microbial flocs (filaments). These filaments can contribute to the adhesiveness and viscosity (Sommer et al. 2006), and together with the larger particles, this reduce the infiltration in the soil as the soil pores are blocked and the slurry in consequence are distributed over a larger surface, which in addition to pH effect and a longer time exposure will increase the NH_3 emission potential of digested slurry. Taking these new factors in consideration it is estimate that the NH_3 emission from digestate applied on soil with trailing hoses and injected in grass will be 15% higher than emission from untreated slurry. The emission factors of untreated slurry (Table S6) are from Hansen et al. (2008).

Table S6. Ammonia emission factors for cattle and pig slurry applied to soil (Hansen et al. 2008), and the novel estimates emission from digestate.

Livestock Category	Crop	Month	Application method	NH_3 emission cattle slurry	NH_3 emission digestate

				% of TAN	% of TAN
Cattle	Whole-crop barley w/second-layer crop	April	Injected into black soil	2	2
Cattle	Clover grass, height < 5 cm	March	Injected into grass	24	
Cattle	Clover grass, height < 5 cm	June	Trailing hose + acid*	32	-
Cattle	Clover grass, height < 5 cm	July	Trailing hose + acid*	32	-
Cattle	Clover grass, height < 5 cm	August	Trailing hose + acid*	32	-
Cattle	Clover grass, height < 5 cm	June	Injected into grass	-	37
Cattle	Clover grass, height < 5 cm	July	Injected into grass	-	37
Cattle	Clover grass, height < 5 cm	August	Injected into grass	-	37
Cattle	Whole-crop barley w/second-layer crop	April	Injected into black soil	2	2
Cattle	Maize	April	Injected into black soil	2	2
Pig	Winter oil seed rape, , before sowing	August	Injected into black soil	1	1
Pig	Winter wheat, height 20 cm	April	Trailing hose	15	17
Pig	Winter wheat, height 20 cm with catch crop	April	Trailing hose	15	17
Pig	Spring barley, height 25 cm	May	Trailing hose	13	15
Pig	Spring barley, before sowing	April	Injected into black soil	1	1
Pig	Winter rape, before sowing	August	Injected into black soil	1	1

*Field acidification to pH 6.4. Digestate is injected into grass without acidification because addition of acid is avoided due to extensive foaming and acid consumption.

**Biogas slurry: TS =45.3 g L^{-1}, TAN=2.85 g N kg^{-1}, pH= 7.73.

***Cattle slurry: DM=7.41TAN = 2.12, pH=6.97

***Pig slurry: TS = 38 g L^{-1}, TAN= 3.2g N kg^{-1}, pH= 7.3 (SEGES 2018).

It is assumed that 50% of the cattle slurry is injected into black soil before sowing maize and barley whole-crop in April, 25% is applied to grass in March before cutting and for the rest of the slurry application is evenly distributed between Jun, July and August. The weighted emission factor for

ammonia loss from cattle slurry applied to the field is calculated to be about 15% of TAN and for digested slurry 17%.
For pig slurry it is assumed that 15% is applied to winter rape, 60% to winter wheat, and 25% to spring barley before sowing and half of the slurry applied to black soil is injected into black soil. The weighted emission factor for NH_3 emission from pig slurry applied to fields is 11% of TAN. The emission factor for digestate is estimated to be 13% of TAN.
From applied deep litter, a large share of the ammonia content in the slurry evaporates (Hansen et al. 2008; Sommer et al. 2019).

4. Calculation of NO_3^- leaching in reference and biogas scenarios

AD processing transforms organic N to inorganic N, which is immediately available to the crop to which the digested fertilizer is applied. The increased N uptake by crops may reduce NO_3^- leaching. In addition the reduced content of organic N in the applied digestate may reduce NO_3- leaching following the crop growth season, as less organic N is available for transformation and leaching during periods without crop growth and N uptake.
Nitrate leaching from the applied fertilizer during the year of application is related to the amount of total N applied (Sørensen and Børgesen, 2015), therefore, AD treatment of livestock slurry is not assumed to affect NO_3 leaching in the first year as the same amount of N (including mineral fertilizer N) will be applied with and without digestion according to the Danish fertilizer regulation. In practice more N is taken up by the first crop after application of digestate, i.e. the mineral fertilizer equivalent is higher due to the increase in inorganic N (Sørensen and Pedersen, 2020). In this study the effect of digestion on NO_3^- leaching over a 10-year period was calculated using a model based on the principles applied by Sørensen and Børgesen (2015), using the following assumptions:

- The leaching from applied mineral N and organic N released during the year of application is the same as for N added with inorganic fertilizers and is based on the marginal leaching calculated using the NLES4 leaching model for a standard crop rotation on cattle farms under Danish conditions (Sørensen and Børgesen 2015). On sandy soils with high precipitation the leaching rate from inorganic N applied in spring is 0.23 kg N kg N^{-1} and on loamy soils with low precipitation the leaching rate is 0.13 kg N kg N^{-1}.
- The leaching from N mineralised due to transformation of organic-N added in digestate in year 2 and subsequent years is assumed to be twice as large as leaching from mineral N applied in the spring, because organic N transformations take place throughout the year (Sørensen and Børgesen, 2015; Sørensen et al. 2019). The leaching rate of mineralized N in the subsequent years is therefore 0.46 kg N kg N^{-1} on sandy soils with high precipitation and 0.26 kg N kg N^{-1} on loamy soils with low precipitation.
- A similar average N mineralization rate is assumed for organic N in all categories of organic manure (Figure 4.1).

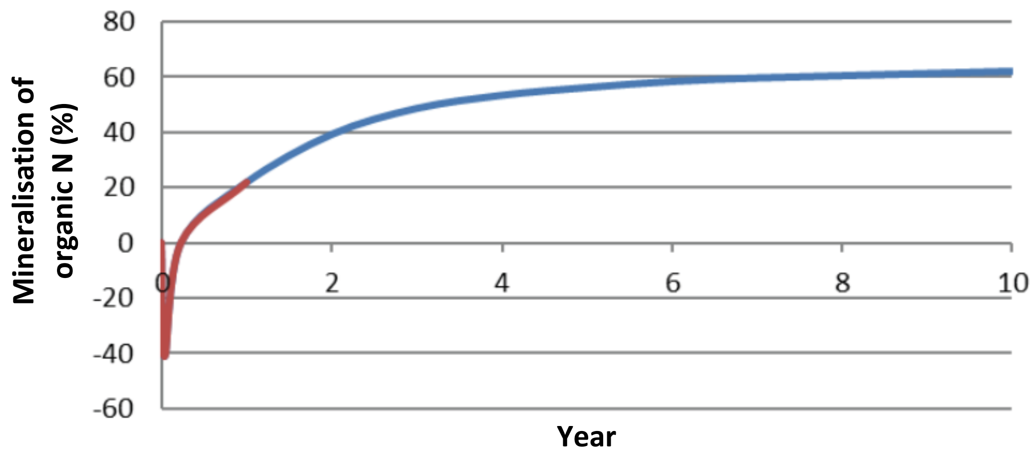

Figure S1. Average cumulative net N mineralisation from organic N applied in livestock manure over a 10-year period after application (Sørensen and Børgesen, 2015).

- The calculation of marginal N leaching was made for clayey soil with low average precipitation and for sandy soil with high average precipitation. An average effect has been calculated after weighting with 80% of the manure applied to sandy soil with high precipitation and 20% to clayey soil with low precipitation, according to the distribution of livestock manure in Denmark (Sørensen and Pedersen, 2020).
- Moreover, assumptions were made for the utilization requirements for the input of organic fertilizer according to Danish legislation and the share of ammonium N (or immediately plant-available N) in organic fertilizers with and without digestion (Table S7).
- The effect of changed ammonia loss occurring in connection with storage and application of fertilizer by conversion to AD were calculated separately.

Table S7. Assumptions about plant available N (NH_4^+-N) in biomasses before and after biogas treatment during the first crop growing season after application of manure, required N use efficiency for manures and organic wastes (by Danish legislation), and calculated reduction in NO_3^- leaching due to AD of manure - not accounting for changed NH_3 loss. The share of NH_4^+-N in manure is based on Sørensen and Børgesen (2015). Organic N expected to be transformed to $NH4^+$ within the first season is included in the NH_4^+-N share of total N.

Fertiliser type	NH_4^+ -N/total N		Utilisation requirement	Reduced NO_3 leaching by digestion	
	Untreated	Digested	%	% of total N	kg N t^{-1}
Cattle slurry	0.58	0.68	70	1.7	0.067
Pig slurry	0.79	0.90	75	1.8	0.105
Cattle deep litter	0.20	0.50	45	5.0	0.477
Poultry manure	0.50	0.80	45	5.0	0.957

Clover grass silage	0.35	0.60	40	5.0	0.440
Maize silage	0.25	0.60	40	-19.3	-0.757*
Straw	0.00	0.30	40	5.0	0.213
Slaughterhouse waste (Stomach and intestinal content)	0.25	0.60	40	5.9	0.229
Biowaste (source-separated organic household waste)	0.25	0.80	40	9.2	0.479

* Negative value indicates increased nitrate leaching by inclusion of maize as an energy crop.

The calculated effect of digestion on nitrate leaching for different categories of manure and organic fertilizer is given in Table S7. If an energy crop such as maize is used in co-digestion at biogas plants then an increased amount of N will be added to the system, which results in increased nitrate leaching (Table S7). The calculations do not include any effect on NO_3^- leaching from the actual cultivation of the energy crop, because this depends on an unknown substitution of alternative crops and the related crop rotation. Under Danish conditions maize cultivation is usually expected to contribute with a higher nitrate leaching than most other crops. With the current Danish fertilizer regulation farmers can apply slightly more mineral N fertilizers after anaerobic digestion due to the accounting system for organic fertilizers (Sørensen and Pedersen 2020). The effect of this is not accounted for in the present calculation of nitrate leaching.

In organic cropping systems with plant production, it is common to have grass-clover crops as a green manure, and it can be relevant to utilize the harvested grass-clover in a biogas plant and return the digestate as a fertilizer. If grass-clover is used in a biogas plant, it is here assumed that the alternative to the digestion of harvested grass-clover is to leave the grass-clover as cut green manure directly in the field. It is assumed that 35% of N in untreated grass-clover is available in the short term. De Notaris et al. (2018) compared the leaching measured in two organic systems where cutting of grass-clover/lucerne with re-application as green manure was compared to the harvesting of crops and addition of a similar amount of N in digested manure. They found slightly greater leaching when digested manure was applied (not statistically significant), but also a N balance indicating accumulation of organic N in soil during the period 2010-2014, when cut grass-clover was left in the field. During the subsequent period (2015-2017), slightly greater leaching was found in a system where grass-clover was left in the field with increased leaching of 4 kg N ha^{-1} compared to a system with applied digested manure. These measurements are subject to considerable uncertainty, but the measured values agree well with the calculated effect of digestion of grass-clover reducing leaching with 5.0% of total N (Table S7). That corresponds to a reduction in NO_3^- leaching of about 3-4 kg N ha^{-1} at an average removal and subsequent return of digested grass-clover of 70 kg N ha^{-1}, which was seen in the above study (De Notaris et al., 2018). It should be noted that, economically, it will usually be the best solution to use grass-clover as cattle feed and then subsequently digest the manure produced by the animals. However, not all organic arable farms are located close to cattle farms where grass-clover can be used and exchanged with manure.

Changes in ammonia emission have both direct and indirect effects on nitrate leaching. If ammonia emission after application is increased less N is available for leaching from the area (direct effect). Ammonia emission also influences deposition of N on agricultural land, forests, and nature areas and thereby has an indirect effect on nitrate leaching. It is estimated that about 30% of the evaporated NH_3 from agriculture is re-deposited on Danish soils (Hansen et al., 2008; Petersen and Sørensen, 2008). Olesen (2020) has estimated that an average of about 33% of the deposited nitrogen is leached. The increased NH_3 emission due to including AD in the systems thereby results in increased NO_3^- leaching corresponding to of 30% x 33% = 10% of the increased NH_3 emission. The total 10-year effect of an increased NH_3 loss is a reduced leaching of 22% (direct effect) less an increase of 10% (indirect effect), which results in a total reduction in NO_3^- leaching corresponding to 12% of the increased N loss with NH_3. This factor has been used to calculate the total reduction of nitrate leaching.

In connection with digestion, it is possible to achieve greater availability of N in slurry if, at the same time, increased ammonia losses can be avoided. The increased fertilizing effect has previously been assessed by Sørensen and Børgesen (2015) to correspond to 0.10-0.15 kg N kg N^{-1} during the year of application. The long-term residual effect will, however, be lower, so the long-term increase in fertilizer value only corresponds to 0.05 – 0.08 kg N kg^{-1} treated N. With the current fertilizer regulations, where economically optimal fertilizer standards can be used, the increased fertilizing effect is not expected to be utilized for increased yields but may potentially be utilized by the farmer to reduce the use of commercial fertilizers. With the current rules, such "voluntary" standard reduction can be substituted by reduced use of catch crops. Basically, this means that the reduced use of fertilizer is not translated into reduced NO_3^- leaching. The farmer can, however, save on costs of both establishing catch crops and buying N-fertilizers.

5. Carbon storage in soil in reference and biogas scenarios

The effect of biogas treatment of slurry and other livestock manure on soil carbon (C) storage is poorly understood. Therefore we base our estimates on a laboratory incubation study, where soil C storage as a fraction of the total amount of C added to AD was slightly lower than of C in untreated slurry manage, which is applied to soil (Thomsen et al. 2013). This study indicated that the amount of C digested in the biogas plant contributed to C storage in soil by 25% of the effect achieved from adding C in fresh plant material and straw, i.e., 0.25 × 15% = 3.75% of the C in the biogas would have been stored after a 20-year period, assuming soil storage of 15% of the carbon added to plant material over 20 years (Christensen, 2004). Corresponding to C storage that would have been seen had the organic material not been degraded in the biogas plant. This results in net CO_2 emissions from minor soil carbon storage of 37.5 kg C, corresponding to 138 kg CO_2 per tonne of C in the produced biogas (methane and CO_2), or, with a carbon content of 45% in VS, minor carbon storage of 62 kg CO_2 per tonne of VS degraded in the biogas plant. The estimates of the effect of biogas digestion on carbon storage are subject to uncertainty, but are in line with studies showing that a reduction in the amount of degradable C incl. carbohydrate added to the soil, will reduce C storage (Liang et al. 2018).

References

See reference list in the article.