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Assessment of ammonia emission mitigation efficiency of established and novel field application techniques for anaerobically digested slurry

Johanna Pedersen [a,*], Sasha D. Hafner [a], Cristiane Romio [a], Andreas S. Pacholski [b]

[a] Aarhus University, Department of Biological and Chemical Engineering, Denmark
[b] Thünen Institute for Climate-Smart Agriculture, Braunschweig, Germany

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ABSTRACT

Biogas can be produced from anaerobic digestion of animal slurry. In Denmark, recalcitrant biomasses are often added along with slurry, resulting in a high dry matter digestate with a high potential for ammonia emission following field application. A total of 12 field trials were conducted to quantify the effects of two digestate treatments and four application techniques on ammonia emissions after field application. Digestate was treated by low dose acidification and separation and applied by trailing hose, trailing shoe, a novel combination of trailing shoe and harrowing tine, and open slot injection. Emissions were determined with a system of dynamic flux chambers and cavity ring-down spectroscopy. Measured ammonia emissions varied substantially among trials and were clearly related to digestate source. Acidification with 1 kg concentrated sulfuric acid per ton slurry effectively reduced emissions in some cases, but the effect was inconsistent and most likely related to dry matter content of the digestate. There were lower cumulative emissions from the liquid fractions after separation compared to the unseparated digestate in all cases, but the reduction effect varied considerably (33–83 %). There was no clear difference between trailing hose and trailing shoe. But the modified trailing shoe with harrowing tine significantly reduced cumulative emissions compared to the trailing shoe only (34–39 %). Based on the data, it is not feasible to recommend one universal application approach or treatment strategy for consistently reducing emission under all conditions. Future work should investigate the sources of variation in emission mitigation, to understand when specific measures are effective.

1. Introduction

Livestock production causes emission of ammonia (NH_3) and greenhouse gases throughout the management chain (Uwizeye et al., 2020; Yan et al., 2024). These emissions are harmful to the environment, human health, and climate (Sutton et al., 2011; Wyer et al., 2022). One of the primary emission sources is animal manure (Uwizeye et al., 2020). Researchers have investigated low-emission manure management options to reduce the environmental impact while conserving nitrogen (N) to meet the food requirements of a growing population worldwide (Ambrose et al., 2023; Beltran et al., 2021; Hassouna et al., 2023).

Liquid and solid animal manure have for decades been used for production of biogas, which can be a sustainable energy source (Khoshnevisan et al., 2021; Weiland, 2009). The residue from biogas production, digestate, can subsequently be applied in the field as organic fertilizer to utilize the nutrients originally present in the feedstock (Möller and Müller, 2012; Nkoa, 2013; Zilio et al., 2021). Digestate produced from animal slurry can be field applied using the same methods as raw slurry. With no co-substrates, digestion of manure reduces dry matter (DM) content by biodegradation (Möller and Müller, 2012; Möller, 2015) but increases pH through production of ammonia and consumption of organics acids (Georgacakis et al., 1982; Sommer and Husted, 1995a; 1995b). But in Denmark, the input into the biogas reactors has changed over recent decades. Addition of relatively recalcitrant biomass, such as straw and deep litter as co-substrates, is used to increase biogas production. The high dry matter (DM) (Romio et al., 2024) and high pH of resulting digestate could lead to much higher ammonia emission after field application (Pedersen and Hafner, 2023).

To lower ammonia emissions and improve the utilization of slurry and digestate as a fertilizer, different treatments and application techniques have been developed. Deep injection or immediate incorporation are the most effective application techniques for reducing emission

* Corresponding author. Aarhus University, Blichers Alle 20, 8830, Tjele, Denmark.
E-mail addresses: jp@bce.au.dk (J. Pedersen), sasha.hafner@bce.au.dk (S.D. Hafner), ane.romio@bce.au.dk (C. Romio), andreas.pacholski@thuenen.de (A.S. Pacholski).

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(Webb et al., 2010), but they are not suitable for growing crops. Open slot injection in grass can be very effective at reducing emission (Hansen et al., 2003; Huijsmans et al., 2001), but yield reductions have been reported (Rodhe and Etana, 2005). For surface application in growing crops, band spreading by trailing hose or trailing shoe reduces emissions compared to broadcast application (Hafner et al., 2025; Misselbrook et al., 2002). Generally similar emission ranges have been reported for trailing hose and trailing shoe, but substantial differences have also been observed in some studies (Misselbrook et al., 2002). Variability in emissions measured after application is high for all application methods (Hafner et al., 2025; Huijsmans et al., 2001; Webb et al., 2010), which can be ascribed to interactions between soil and slurry properties (Huijsmans et al., 2001; Häni et al., 2016; Misselbrook et al., 2002), differences in the design and use of specific application technologies (Webb et al., 2010), and measurement error (Kamp et al., 2024).

Available slurry and digestate treatments include solid-liquid separation, which alters physical properties and chemical composition. Separation produces two or more fractions, where dry matter (and typically nutrients) is concentrated in one or more of the fractions: typically, a high dry matter 'solid' fraction and low dry matter 'liquid' fraction. A recent literature review found that after application of the liquid fraction of animal slurry or digestate ammonia emissions are on average lower than after application of the unseparated slurry or digestate (Pedersen et al., 2022). However, variation in reduction efficiency is large, related in part to differences in dry matter separation efficiency, which can vary greatly (Romio et al., 2024). The total ammonia emissions from application and storage of both the liquid and the solid fractions must be considered when assessing separation as an ammonia reducing technology (Amon et al., 2006; Pedersen et al., 2022).

Slurry or digestate can be acidified before or during field application, and acidification has been found to be an efficient method to reduce ammonia emissions (Fangueiro et al., 2015). Lowering of pH ensures that more ammonia will be in the non-volatile ammonium form (NH_4^+), decreasing the emission potential. The emission mitigation efficiency of acidification depends on the buffer capacity of the slurry, type and amount of acid used, as well of the timing of acidification (Fangueiro et al., 2015). Concentrated sulfuric acid (H_2SO_4) is commonly used due to its low cost and effectiveness in lowering pH. To effectively lower ammonia emissions from high pH slurry types, such as digestate, large amounts of acid are required. Nyord et al. (2021) found that more than 10 kg 96 % sulfuric acid per ton digestate were needed to reach a pH of 6. These large acid demands are expensive and can cause extensive foaming of the slurry or digestate, making pumping and field application challenging (personal observation). To obtain the benefits of acidification without increasing costs too much or causing management challenges, it has been proposed to use small amounts (<2 kg 96 % sulfuric acid per ton digestate) of acid during field application and accept a lower emission reduction (Pedersen and Nyord, 2023).

Because of the increase in field application of slurry digestate and its high ammonia emission potential, exacerbated by a shift toward higher dry matter, there is a pressing need to find application techniques or digestate treatments that can ensure low ammonia emissions when digestate is applied in growing crops where injection is not practical. Separation and acidification are treatments known to reduce ammonia emissions after field application, but only limited data are available for slurry digestate, especially digestate with high dry matter (Pedersen and Hafner, 2023). Furthermore, banded digestate application (trailing hose or trailing shoe) still poses a high ammonia emission potential compared to untreated slurry. Increased knowledge about whether a general lower emission from trailing shoe applied digestate can be expected compared to trailing hose applied digestate is needed for management decisions. Typical emission studies with a small number of application events or single source of digestate cannot capture true variability in emission reductions that probably exists due to differences in digestate properties and interactions with soil and weather. Although it complicates efforts to reduce emissions, understanding this variability is important for tailoring treatments to specific situations for maximizing emission reductions. Inclusion of multiple digestate sources and many field trials over two years made the present study unusually informative in this regard.

The objective of this study was to assess the relative ammonia emission mitigation effect of both different application techniques compared to trailing hose application and digestate treatments by separation and low-dose acidification (1.84 kg 96 % sulfuric acid per ton digestate) using four digestate sources from within Denmark in a total of 12 field trials over two years. Differences in emissions between treatments were quantified with a system of dynamic flux chambers (DFC) and high time resolution measurements of the ammonia concentration. It was hypothesized that a) trailing shoe application of digestates results in lower emissions compared to trailing hose application, b) emissions can be further reduced if the trailing shoe is modified to increase the digestate-soil interaction by shallow soil surface cutting, and c) separation and low-dose acidification lead to lower emissions compared to untreated digestate, but with varying results depending on digestate properties.

2. Materials and methods

A total of 12 field trials were conducted in 2023 and 2024 to quantify ammonia emissions after field application of digestate (Table 1). Four trials with band spreading simulated by manual application of digestate were conducted to assess the mitigating effect of mechanical treatments and low-dose acidification (trials 1–4; Table 1). Eight trials were conducted with digestate application by plot-scale machinery for assessment of different application methods (trials 5–12, Table 1). Trials 5–8 tested the relative difference in emissions between four different application techniques: trailing hose, two different designs of trailing shoe, and trailing shoe with a harrowing tine (Table 1). Trials 9 and 10 were used to test the difference between emissions after application with trailing hose and trailing shoe at two different driving speeds during application (4 and 12 km h^{-1}). The emission results from application at 12 km h^{-1} were analysed with results from trials 5–8 (Table 1), whereas the comparison between the emissions at two different application rates are presented and discussed in (Pedersen et al., 2026). Trials 11 and 12 tested the effect of different application techniques: trailing hose, trailing shoe, and open slot injection.

Measured cumulative emissions should not be taken as an estimate of absolute emissions because the chambers modify the environment of the emitting surface compared to ambient conditions by changing air flow, precipitation, radiation, temperature, and soil conditions. But it was assumed that chamber measurements provided a reasonable estimate of relative effects, as substantiated by earlier comparisons with micrometeorological measurements (Hafner et al., 2024; Sommer and Misselbrook, 2015).

Digestate and soil properties and air temperature were measured in all trials (Table 2). All trials except 11 and 12 included slurry digestate from the biogas plant at Campus Viborg, Aarhus University (digestate A) applied by trailing hose, providing a reference to account for emission differences related to e.g., weather.

2.1. Field trial conditions

All trials except trials 11 and 12 were conducted on one field site at the experimental research station at Campus Viborg, Aarhus University, Denmark (56.49316 °N, 9.56128 °E). The field was a flat loamy sand (9 % clay, 24 % silt, 65 % sand) field with winter wheat sown in the preceding fall of both years. Prior to trial 4, the wheat was cut to a height of 10 cm (Table 2), as it was too high to properly install the flux chambers.

Trials 11 and 12 were performed at a commercial farm. Both fields had a coarse sand texture (4.2 % clay, 1.8 % silt, 90.3 % sand), with 1st year grass, cut in the vegetation period the day before digestate

Table 1
Overview of experiments performed.

Trial	Application technique[a]	Digestate treatment[b]	Field	Application start	Measurement duration (h)	Group for statistical test
1	TH manual	Sep-D, Dis, Acid	Viborg	12-04-2023 14:45	150	i
2	TH manual	Sep-S, Acid	Viborg	19-04-2023 14:20	150	i
3	TH manual	Sep-D, Sep-S	Viborg	10-05-2023 13:51	150	i
4	TH manual	Sep-D, Sep-S, MF	Viborg	03-06-2024 10:50	150	i
5	TH, TS1, TS2, TS3	None	Viborg	13-03-2024 16:51	85	ii
6	TH, TS1, TS2, TS3	None	Viborg	19-03-2024 14:51	150	ii
7	TH, TS1, TS2, TS3	None	Viborg	08-05-2024 15:31	150	ii
8	TH, TS1, TS2, TS3	None	Viborg	23-05-2024 13:38	150	ii
9	TH, TS1	None	Viborg	06-03-2024 16:25	150	ii
10	TH, TS1	None	Viborg	24-04-2024 15:30	150	ii
11	TH, TS1, OSI	None	Fåborg-1	25-06-2024 13:44	85	iii
12	TH, TS1, OSI	None	Fåborg-2	01-07-2024 15:16	85	iii

[a] TH manual: digestate was applied manually with a watering can fitted with a hose to mimic trailing hose application, TH: trailing hose, TS*X*: Trailing shoe, X = model 1, 2, or 3, OSI: open slot injection.
[b] Sep: separated, D: separated with decanter centrifuge, S: separated with screw press, Dis: disrupted, Acid: acidified with 1.84 kg H_2SO_4 (concentrated sulfuric acid) t^{-1} slurry, MF: microfiltration.

Table 2
Trial conditions. Soil-water content (± standard deviation, n = 3), dry bulk density (± standard deviation, n = 3), soil pH (n = 1), ambient air temperature at application (based on hourly averages) and average ambient air temperature during the measuring period with range in parentheses. Plot of air temperature for the whole measuring period can be found in Fig. S9.

Trial	Water content (g g^{-1})	Dry bulk density (g cm^{-3})	Soil pH	Crop height (cm)	Ambient air temperature (°C)	
					At application	Average over whole measuring period (min., max.)
1	0.20 ± 0.01	1.13 ± 0.04	5.3	12	8.66	8.3 (3.7, 13.7)
2	0.24 ± 0.01	1.07 ± 0.06	5.2	13	12.7	9.2 (2.9, 18.1)
3	0.14 ± 0.02	1.06 ± 0.09	5.6	8	17.6	13.0 (5.4, 21.2)
4	0.22 ± 0.01	1.33 ± 0.09	5.7	10	14.5	10.8 (5.6, 17.7)
5	0.24 ± 0.01	1.23 ± 0.04	5.6	7	8.39	5.4 (−2.4, 10.7)
6	0.25 ± 0.01	1.15 ± 0.03	5.5	8	5.36	5.4 (0.6, 10.5)
7	0.23 ± 0.01	1.15 ± 0.04	4.9	28	13.9	12.8 (5.9, 20.7)
8	0.10 ± 0.02	1.20 ± 0.02	5.6	38	16.4	14.8 (8.9, 20.4)
9	0.26 ± 0.01	1.31 ± 0.05	5.6	7	3.56	1.7 (−3.2, 5.8)
10	0.29 ± 0.05	1.17 ± 0.04	5.3	22	6.22	8.8 (2.1, 18.2)
11	0.17 ± 0.01	1.38 ± 0.06	5.8	8	22.6	17.9 (11.3, 26.8)
12	0.21 ± 0.01	1.31 ± 0.02	7.0	22	15.5	13.7 (11.8, 16.2)

application (Fåborg-1, 55.57461 °N, 8.78369 °E) and 2nd year grass, cut 8 days before digestate application (Fåborg-2, 55.57949 °N, 8.78632 °E) for trials 11 and 12, respectively.

On the day of digestate application, one top-soil sample was taken for measurement of 1:1 water pH and three 100 cm^3 undisturbed soil cores were taken from the top layer (0–5 cm) of the soil for determination of dry bulk density and gravimetric soil water content (measured after drying at 105 °C). Each trial required an area of approximately 20 m × 20 m, and soil homogeneity was assumed within this area. For each trial a new section of the field was used to avoid placement of chambers onto a spot where slurry was applied during a previous trial. Soil properties and crop height can be found in Table 2. Ambient air temperature at 2 m height was logged continuously with a weather station at a nearby field (<0.7 km) in 1 h intervals (Table 2 and Fig. S9).

2.2. Digestate properties and application

Digestates from a total of 4 different biogas plants were used during the trials. The plants varied in feedstock as well as hydraulic retention time, temperature of reactor(s), and other management practices, but all were primarily based on pig and cattle slurry as feedstock. Digestates A and B originated from biogas plants treating some straw, with more in B, although the quantity is unknown, while C and H had none. These differences in feedstock resulted in different physical and chemical characteristics of the digestates (Table 3), which could affect emissions and the relative reductions obtained by treatments or application techniques.

The total ammoniacal nitrogen (TAN) application rate (kg ha^{-1}) was fixed at 60 kg TAN ha^{-1} as this is a standard application rate for winter wheat in Denmark. Digestate application rate (m^3 ha^{-1}) (Table 3) was determined based on the total ammoniacal nitrogen concentration. The total ammoniacal nitrogen concentration was determined photometrically 1–2 days prior to application in a well-mixed digestate sample, using a commercial kit (Spectroquant Test kit No. 1.00683.0001, Merck KgaA, Darmstadt, Germany).

Subsamples of the digestates were also taken in the field for analysis. Digestate pH was measured in the field (Knick Portavo 902 pH meter) at the start of the application. pH was also measured in the laboratory in parallel with the other digestate analysis (Knick Portavo 902 pH meter) (Table 3). In the laboratory, digestate was gravimetrically analysed for dry matter and volatile solids by heating at 105 °C and calcination at 550 °C, respectively (APHA, 2017). Total nitrogen was analysed by low-field nuclear magnetic resonance spectrometry (Jensen et al., 2021) and total ammoniacal nitrogen was measured as above. Apparent viscosity was measured in triplicate with a rotational viscometer (Brookfield DV2T, AMETEK Brookfield, Middleboro, USA). Viscosity indicates the resistance of a fluid to flow, i.e., the higher the viscosity, the lower the fluid flow rate for the same applied driving force. Viscosity is defined as the ratio between shear stress (force per unit area) and shear rate (velocity gradient) (Schneider and Gerber, 2020). A volume of approximately 600 mL of sample at 20 ± 3 °C was used. The cylindrical spindles 61 LV, 66 LV-C, 67 LV-C, and 64 LV were employed, according to the apparent viscosity ranges of the samples. Apparent viscosities were measured at different shear rates (proportional to the spindle rotational speed) for 1 min, with shear rates increasing from 1 to 44 s^{-1}. As recommended by the viscometer manufacturer, viscosities values measured

Table 3
Digestate properties (mean ± standard deviation, $n = 2$). TAN: total ammoniacal nitrogen.

Trial	Digestate[a]	Treatment[b]	Dry matter (%)	Volatile solids (% dry matter)	TAN (g kg^{-1})	Total nitrogen (g kg^{-1})	pH[c]	pH[d]	Application rate (m^3 ha^{-1})	Application rate (kg TAN ha^{-1})
1	A	None	5.56 ± 0.01	76.62 ± 0.14	1.6 ± 0.01	2.7 ± 0.26	7.9 ± 0.02	8.3 ± 0.01	35	55
1	A	Sep-D	3.19 ± <0.01	66.59 ± 0.08	1.5 ± 0.12	2.9 ± 0.70	7.8 ± 0.01	7.9 ± NA	35	52
1	A	Sep-D + acid	3.29 ± <0.01	64.93 ± <0.01	1.5 ± 0.01	2.6 ± 0.15	7.8 ± 0.21	6.5 ± 0.12	35	54
1	A	Dis	5.33 ± <0.01	77.51 ± 0.14	1.5 ± 0.01	3.1 ± 0.04	7.8 ± 0.05	8.1 ± 0.11	35	51
1	A	Dis + acid	5.46 ± 0.01	76.73 ± 0.06	1.6 ± 0.03	2.8 ± 0.74	7.4 ± 0.01	6.5 ± 0.03	35	56
2	A	None	5.44 ± 0.05	76.14 ± 0.04	1.5 ± 0.26	3.0 ± 0.38	8.0 ± 0.01	8.4 ± 0.01	32	47
2	A	Acid	5.60 ± 0.06	75.30 ± 0.13	1.8 ± 0.32	3.7 ± 0.24	7.6 ± 0.07	7.3 ± 0.04	32	57
2	B	None	9.18 ± 0.04	74.48 ± 0.06	2.5 ± 0.09	4.9 ± 0.25	7.8 ± 0.02	8.1 ± 0.01	23	58
2	B	Acid	9.30 ± 0.07	74.33 ± 0.19	2.3 ± 0.13	5.6 ± 0.18	7.5 ± 0.01	7.7 ± 0.01	23	53
2	B	Sep-S + acid	7.33 ± <0.01	68.91 ± 0.09	2.6 ± 0.05	5.2 ± 0.03	7.7 ± 0.02	7.5 ± 0.01	20	51
3	A	None	5.53 ± 0.01	75.73 ± 0.06	1.2 ± 0.29	3.4 ± 0.17	8.0 ± 0.42	7.9 ± 0.01	38	45
3	B	None	9.24 ± 0.02	74.23 ± 0.02	2.1 ± 0.29	4.4 ± 0.08	7.8 ± 0.03	8.1 ± 0.01	23	48
3	B	Sep-S	7.03 ± 0.01	68.52 ± 0.07	2.4 ± 0.17	4.7 ± 0.26	7.7 ± 0.06	7.9 ± 0.01	19	46
3	C	None	4.34 ± 0.04	66.02 ± 0.06	2.6 ± 0.27	3.9 ± 0.14	8.0 ± 0.02	8.2 ± 0.01	21	55
3	C	Sep-D	2.52 ± <0.01	58.57 ± 0.10	3.4 ± 0.48	5.0 ± 0.054	7.9 ± 0.04	8.0 ± 0.01	19	64
4	A	None	4.16 ± 0.15	73.31 ± 0.40	2.0 ± 0.09	3.0 ± 0.01	7.7 ± 0.03	7.8 ± 0.01	30	60
4	A	Sep-D	2.98 ± 0.01	67.48 ± 0.19	1.8 ± 0.09	3.0 ± 0.12	7.7 ± 0.02	7.8 ± 0.01	30	53
4	A	Sep-S	3.3 ± <0.01	68.05 ± 0.07	2.1 ± 0.01	3.1 ± 0.01	7.7 ± 0.01	7.8 ± 0.01	30	63
4	A	MF thin	3.42 ± <0.01	67.62 ± 0.04	2.0 ± 0.04	3.1 ± 0.19	7.8 ± 0.02	7.9 ± 0.01	30	59
4	A	MF thick	7.82 ± 0.07	76.83 ± 0.23	2.1 ± 0.03	2.8 ± 0.38	7.4 ± 0.01	7.8 ± 0.01	37	78
5	A	None	3.66 ± <0.01	73.63 ± 0.32	1.5 ± 0.03	1.9 ± 0.02	7.6 ± 0.03	8.1 ± 0.01	32	48
6	A	None	3.71 ± 0.07	73.94 ± 0.24	1.5 ± 0.02	1.9 ± 0.28	7.8 ± 0.01	8.0 ± 0.01	32	49
7	A	None	3.47 ± 0.09	74.16 ± 1.03	1.5 ± 0.16	2.2 ± 0.18	7.9 ± 0.02	8.1 ± 0.01	32	49
8	A	None	2.95 ± 0.13	72.18 ± 0.47	1.5 ± 0.06	2.7 ± 0.34	7.9 ± 0.01	8.1 ± 0.01	32	48
9	A	None	3.35 ± 0.005	71.87 ± 0.08	1.5 ± 0.07	2.0 ± 0.05	7.7 ± 0.05	8.2 ± 0.01	32	48
10	A	None	3.74 ± <0.01	75.23 ± 0.34	1.4 ± 0.03	1.9 ± 0.04	7.9 ± 0.03	8.1 ± 0.01	32	45
11	H	None	6.72 ± 0.11	68.33 ± 0.21	3.1 ± 0.01	4.7 ± 0.58	7.6 ± 0.01	7.3 ± 0.02	20	62
12	H	None	6.66 ± 0.05	65.59 ± 0.57	3.0 ± 0.04	4.8 ± 0.04	7.6 ± 0.03	7.7 ± 0.06	20	59

[a] Digestate source/biogas plan ID.
[b] Sep: separated, D: separated with decanter centrifuge, S: separated with screw press, Dis: disrupted; acid: acidified with 1 kg H2SO4 t-1 slurry, MF: microfiltration.
[c] Measured in the laboratory.
[d] Measured in the field immediately prior to application.

at shear rates resulting in torques lower than 10 % were discarded. Fluids presenting constant apparent viscosities at increasing shear rates are known as Newtonian fluids, while fluids presenting decreasing and increasing viscosities at increasing shear rates are called shear-thinning and shear-thickening fluids, respectively. The viscosity behaviour at varying shear rates is often described by the power law model (Equation (1)), which encompasses two constants, the consistency coefficient K (cP s^n) and the flow behaviour index n (dimensionless). The higher the consistency coefficient, the higher the apparent viscosity for a same shear rate and flow behaviour index. A flow behaviour index of 1 indicates that a fluid behaves as a Newtonian fluid, while flow behaviour indexes smaller and larger than 1 indicate that a fluid is shear thining and shear thickening, respecitvely (Schneider and Gerber, 2020; Schneider and Gerber, 2020)

$$\eta = K\,\gamma^{n-1}. \qquad \text{(Equation 1)}$$

Values of K and n were estimated by minimizing the sum of squares between measured η and the values calculated with Equation (1) using Excel's Solver tool.

Particle size distribution was measured in duplicate by a combination of gravimetric and laser diffraction methods since laser diffraction required that particles were smaller than 2 mm. It is expected that samples with larger particles impair slurry infiltration rate more than samples with smaller particles for the same dry matter content due to a more extensive blockage of soil pores. Approximately 30 g of sample was filtered in a sieve with 2 mm openings and washed with approximately 500 mL deionized water. The sieves were dried at 105 °C for 24 h to determine the fraction of dry matter larger than 2 mm. The filtrate particle size distribution was determined by laser diffraction (Malvern Mastersizer, 2000; Malvern Panalytical, Malvern, United Kingdom) after dispersion with water using the Fraunhofer scattering model.

2.2.1. *Digestate treatment trials*

For trials 1–4 (Table 1), digestate fractions were collected 2–4 weeks prior to application and stored outside under a roof in closed buckets until the time of application. The digestate was applied manually with a watering can fitted with a hose to mimic trailing hose application. Earlier comparisons have shown that variability is lower with manual application than machine application (Pedersen et al., 2024), so it was used in order to allow for fewer replicates and therefore more treatments in each trial. The predetermined volume of digestate was evenly distributed in two narrow bands at the soil surface in the area that the dynamic flux chamber would subsequently cover. The volumes were determined based on a total ammoniacal nitrogen analysis prior to the trials. The true application rate was later calculated based on the volume applied, and subsequent total ammoniacal nitrogen analysis of samples collected at the time of application (Table 3).

Digestates were separated with different separation techniques. Digestate A was separated with a decanter centrifuge (GEA Westfalia Separator, GEA Industry GmbH, LA, USA) for trials 1 and 4 and with a screw press separator (SEPCOW260-3, Saveco, IL, USA) for trial 4. Only the liquid fractions from these separations were used for field application trials. The liquid fraction of screw press separated digestate A was further separated with a micro filter (MFT260, Saveco, IL, USA) for trial 4, yielding a thick and a thin liquid fraction, both used in the application trials (Table 3). For trial 1, digestate A was also treated with a mechanical disintegrator (DisRuptor, Vogelsang GmbH & Co. KG, Essen (Oldenburg), Germany) intended to reduce particle size and viscosity. Digestate C was also separated with a decanter centrifuge (GEA Westfalia Separator, GEA Industry GmbH, LA, USA), whereas digestate B was separated with a screw press separator (Boerger, LLC, Chanhassen, MN, USA). From both separations, only the liquid fraction was used for application trials (Table 3).

For slurry acidification, a fixed amount of 1.84 kg concentrated sulfuric acid (H_2SO_4, 96 %) per m^3 digestate was used (corresponding to 1 L acid per m^3 digestate). To mimic field acidification, the corresponding amount of acid was added to the watering can and mixed with the digestate by stirring immediately prior to application.

2.2.2. *Digestate application technique trials*

For trials 5–10 (Table 1), digestate was collected 4 weeks prior to the first application in a concrete storage tank with cover. Before each application the digestate was stirred in the tank for proper mixing. The slurry was applied with a 9-m experimental boom allowing application with three different techniques simultaneously (see details in supporting materials section S1). Four different application techniques were tested: trailing hose (TH); two trailing shoe products abbreviate as TS1 (Samson Agro A/S, Viborg, DK) and TS2 (Bomech, Albergen, NL); and TS3, a novel technique combining trailing shoe with a harrowing tine, which is a further development of the prototype tested in McCollough et al. (2022). TS3 was produced by adding a harrowing tine to the TS1 design (for pictures, see supporting materials section S1). The design of TS3 was made with the purpose of enabling it to interact with the soil to create a mix of soil and slurry after application, even during conditions where the soil has a surface crust, which is common in Denmark on non-sandy soils with winter cereal crops (personal observation) due to the mudding impact of rain on soil aggregates (Lado et al., 2007; Morin et al., 1981).

The distance between the hoses or shoes on the booms was 250 mm. The tractor and slurry tanker had a driving speed of 12 km h^{-1} during the applications, and the hoses or shoes were always in contact with the soil surface during the application.

Digestate for trials 11 and 12 was stored in a covered concrete tank for one month prior to application. The application during trials 11 and 12 was done with a 3-m boom designed for 3-m trial plots. The driving speed was 5.5 km h^{-1}, and the distance between the hoses and shoes was 240 mm, whereas the distance of the open slot injection discs was 180 mm.

Pictures of the digestate application for trials 5–12 can be found in supporting materials S2.

2.2.3. *Trials assessing the experimental setup*

Some trials included measurements with a dynamic flux chamber on bare soil without any fertilizer application, and these results can be used to assess the background emission of the soil and crops. See the supporting materials section S3 for results and discussion of these measurements.

Temperature loggers were placed at the soil surface inside and outside the chambers to check how the chambers affected the soil surface temperature. See the supporting materials section S5 for details.

2.3. *Ammonia emissions*

2.3.1. *Measuring system*

Ammonia emissions were measured with a system of 16 dynamic flux chambers connected to a cavity ring-down spectrometer (CRDS) (model G2103, Picarro Inc., Santa Clara, CA, USA) for concentration measurements. A detailed description of the system and its validation can be found in Pedersen et al. (2024).

Each flux chamber consisted of an open-bottom polyethylene cylinder, with a diameter at the bottom of 700 mm, giving a chamber base area of 0.38 m^2. The total height of the chamber was 392 mm, and a plywood deflector plate was located 92 mm from the top, with 30 mm between the edge of the plate and the sides of the chambers (Fig. S5 and S6). The deflector plate ensures even distribution of the inflow air above the emitting surface. The air intake into the chamber consists of three galvanized-steel pipes that are evenly distributed on top of the chamber (1 m height, 80 mm diameter). In the middle of the chamber, air is drawn in with a fan, and an iris diaphragm is located between the chamber and the fan to control and measure the volumetric air exchange rate (m^3 of air flow per m^3 chamber volume per time, AER (min^{-1})). During all the trials the AER was kept constant throughout the trials at

15 min^{-1}.

From each chamber and from three background ports positioned at the air inlet of three of the chambers, air was drawn through a heated and insulated polyvinylidene fluoride (PVDF) tube (OD: 6.35 mm, ID: 4.76 mm) at 1.5 L min^{-1} to a rotary valve controlled by the CRDS instrument. Air from the valve was analysed by the CRDS instrument for 8 min at each valve position in order to ensure a representative recovery of ammonia concentration with the instrument (Pedersen et al., 2024). The CRDS measured the ammonia concentration every 2 s. The instrument has been evaluated for potential interferences in an agricultural environment (García et al., 2024; Kamp et al., 2019). The mean of the last 30 s of measurements per measurement cycle (8 min) was used for further calculations, as this period has been found to yield stable readings (Pedersen et al., 2024).

The flux (F, mg N m^{-2} min^{-1}) was calculated from the background corrected concentration of ammonia (C, mg N m^{-3}), the airflow in the emission chamber (q, m^3 min^{-1}), and the area of soil surface covered by the tunnel (A, m^2) (Equation (2)).

$$F = \frac{C \cdot q}{A} \quad \text{(Equation 2)}$$

Cumulative emission was calculated from the flux using the trapezoidal rule (Simmons, 1996).

For trials with manual digestate application (trials 1–4), three replicates were used per treatment, whereas four replicates were used per treatment for trials with machine application (trials 5–12) in order to account for the higher expected variability with machine application (Pedersen et al., 2024).

Tests have been performed to assess the effect of different parameters of the application with the 9 m boom on emission to ensure conditions as close to real practice on farms as possible. This includes tests of the digestate distribution in the slurry boom, effect of driving speed on emission, effect of distance between slurry bands and effect of daily changing the position of the dynamic flux chambers to a new location. The results and discussion of these tests can be found in the accompanying technical note (Pedersen et al., 2026).

2.3.2. Statistics

For all statistical analysis, a single flux chamber was used as the unit of analysis, and cumulative ammonia emission after 150 h (trials 1–4 and 6–10) or 85 h (trials 5 and 11–12) as a fraction of applied total ammoniacal nitrogen was the response variable. The grouping of the statistical analysis can be found in Table 1. Trials were grouped based on whether they assessed digestate source and treatment (trials 1–4, group i in Table 1) or application technique (trials 5–12), where winter wheat (trials 5–10, group ii in Table 1) and grass (trials 11 and 12, group iii in Table 1) were further separated. There may be an interaction between crop and application technique, especially between a grass and cereal crop, because cereal crops allow for trailing shoes to get in contact with the soil whereas a grass sward can hinder this, but it would be difficult to separate crop effects from other factors associated with trials with the experimental design used in this work. Even though trial 5 was carried out for a shorter period (85 h) compared to trials 6–10 (150 h) they were included in the same analysis. From Fig. S10 it is evident that there are only very low emissions after 85 h in any of the trials, and there is no effect of application technique. Because soil, weather, and other factors that were not controlled varied among trials and undoubtedly affected emission, trial ID was included as a random component in mixed-effects models. For trials 1–4 (group i in Table 1), the treatment digestate origin (e.g. A or B) and physical treatment (e.g. none or Sep-D) were fixed factors. For trials 5–12 (group ii and iii in Table 1), application method was a fixed factor. The lmer () function from the lme4 package v1.1–35.5 was used for mixed-effects models (Bates et al., 2015). Because of the lack of balance in all three subsets used in the analyses (trials 1–4, 5–10, and 11–12), marginal means were calculated using the emmeans () function from the emmeans package in R (Lenth, 2024). Tukey's test (confidence level of 95 %) was used to compare treatments using the pairs () function from the emmeans package, with the Kenward-Roger method for estimating degrees of freedom (Lenth, 2024).

3. Results and discussion

3.1. Ammonia emissions

The highest ammonia emissions generally occurred right after application, with a large range in fluxes (Fig. 1), which was expected due to the differences in soil and digestate properties as well as application technique and digestate treatment. Generally, clear diurnal patterns were observed, but patterns were less pronounced for the liquid fraction of separated digestates than for the unseparated digestates (Fig. 1, trials 1–4), which is most likely due to the lower dry matter of the liquid fractions (Table 3), leaving less digestate at the soil surface where emission can occur in the days following application. In most of the trials, clear differences between the flux of the various treatments were observed in the first 50 h after application (Fig. 1), but after approximately 50 h only minor differences were measured (Fig. S10).

Emission varied substantially among trials (Fig. 1 and Table 4) and standard deviation related to the random trial variable was estimated as 6.0 % of applied total ammoniacal nitrogen for the trials 1–4 subset and 6.5 % for trials 5–10. All the trials except 11 and 12 were performed on the same field with the same crop, therefore, this variation does not include contributions from effects of soil type and crop, which could be substantial. Results from the present study may only be applicable to sandy soils. The variation within one treatment was relatively low in all the trials. Some treatments showed a high variation, which was always in connection to a very low mean cumulative emission (coefficient of variance >60 %, cumulative emission <5 % of TAN). This is in line with previous work (Pedersen et al., 2024) evaluating the measuring system. Based on the variation and the mean emissions across trials for a certain treatment, it is hypothesized that an increased number of replicates (chambers) per treatment would not have given more significant results. The cumulative emissions from the manually applied digestate A (trials 1–4) were generally higher than the machine applied digestate A (trials 5–10) (Table 4), likely due to the higher dry matter of digestate A in trials 1–4 (average of 4.99 %, Table 3) compared to digestate A in trials 5–10 (average of 3.48 %, Table 3).

Emission was clearly related to digestate source (p-value $<10^{-9}$ from a likelihood ratio test comparing mixed-effects models with and without digestate source) with an estimated standard deviation of about 12 % of applied total ammoniacal nitrogen. Clearly, not all digestates behave the same (Table 4). At least some of this variability appeared to be related to digestate dry matter; emission was positively correlated with dry matter, with a slope of 4 % of applied total ammoniacal nitrogen per % change in dry matter (p-value = 0.014 from a linear model t-test). Consideration of more than 3 digestate sources could strengthen this conclusion. A negative correlation with digestate pH (p-value = 0.045 from a linear model t-test) is unexpected and likely spurious, casting some doubt on the dry matter correlation results.

The coefficient of variation among replicate dynamic flux chambers (CV, standard deviation divided by the mean) varied both between and within trials. For machine application of digestate (trials 5–12), the average coefficient of variation was 20 % (n = 4, range 6–45 %), which

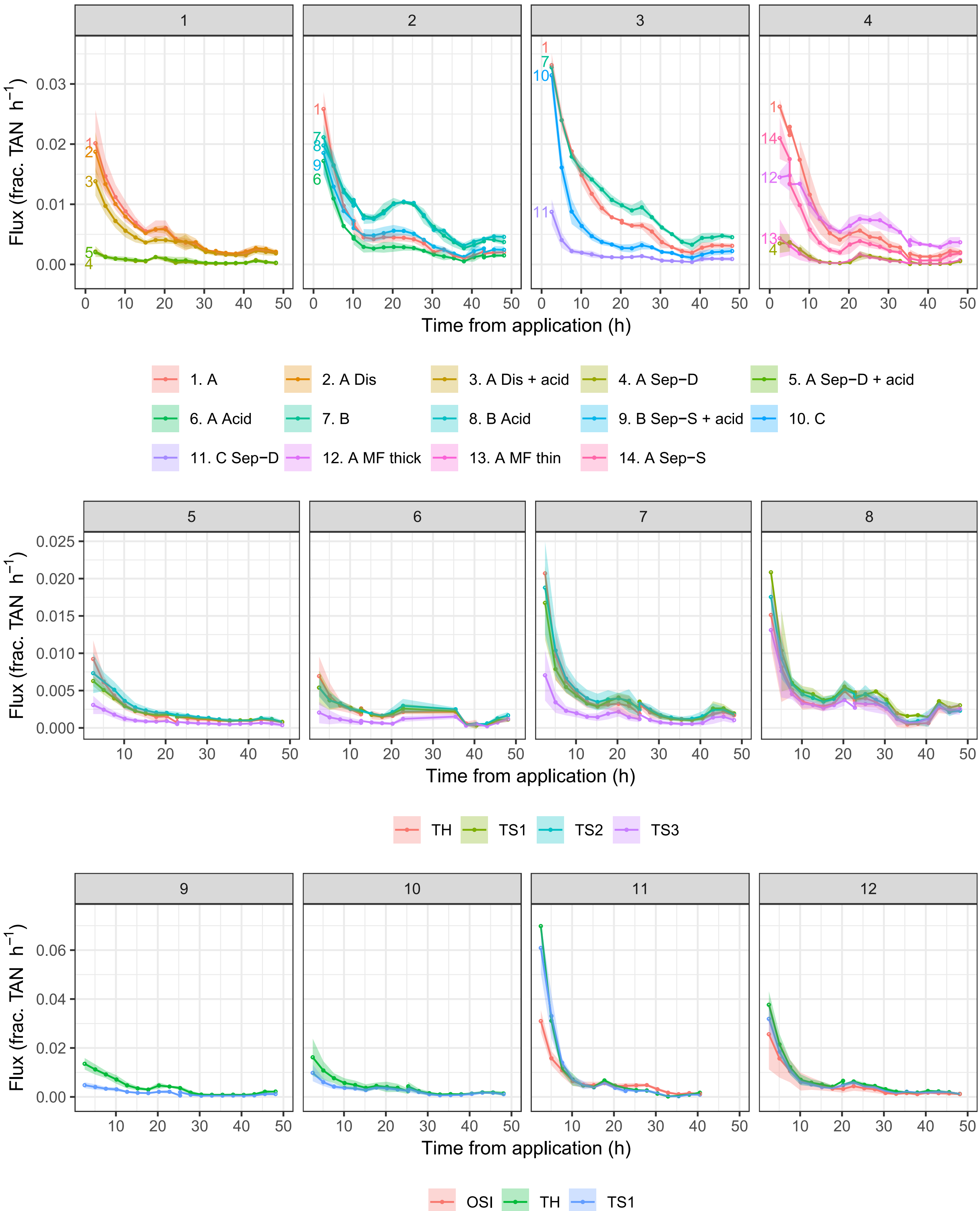

Fig. 1. Ammonia emission rate (as a fraction of applied total ammoniacal nitrogen per hour) after application of digestate. Plot titles show trial ID. Standard deviation are displayed in bands (n = 3 for trials 1–4 and n = 4 for trials 5–12). See Fig. S10 for ammonia flux during the whole trial (150 or 85 h). TAN: total ammoniacal nitrogen; A, B, C: digestate source/biogas plan ID, Sep: separated, D: separated with decanter centrifuge, S: separated with screw press, Dis: disrupted, acid: acidified with 1 kg H_2SO_4 t^{-1} digestate, MF: microfiltration; TH: trailing hose, TSX: Trailing shoe boom (X = model 1, 2 or 3), OSI: open slot injection.

is between the coefficient of variations of 14 % (n = 6) and 55 % (n = 7) observed with the same dynamic flux chamber system in a previous study for application with trailing hose and injection, respectively (Pedersen et al., 2024). For the experiments with manual application (trials 1–4), the average coefficient of variation was 19 % (n = 3, range 2.0–65 %), which is much higher than the 5 % (n = 9) observed in the previous study (Pedersen et al., 2024). The treatments with the highest coefficient of variation in trials 1–4 were the liquid fraction separated digestates where the cumulative emission generally was very low (Table 4). When the separated digestates are removed from the data set,

Table 4
Mean cumulative ammonia loss (% of applied total ammoniacal nitrogen, TAN) (mean ± standard deviation) after application of different untreated and treated digestates with different application techniques. Replication was 3 chambers in group i and 4 in groups ii and iii.

Trial	Treatment[1]	Emission (% TAN)	Trial	Treatment[2]	Emission (% TAN)	Trial	Treatment[2]	Emission (% TAN)
Group i			Group ii			Group iii		
1	A	35.2 ± 4.2	5	TH	12.0 ± 2.4	11	TH	42.6 ± 5.8
1	A Dis	36.2 ± 4.3	5	TS1	11.5 ± 0.7	11	TS1	37.1 ± 5.8
1	A Dis + acid	28.9 ± 5.9	5	TS2	13.2 ± 2.7	11	OSI	27.4 ± 3.5
1	A Sep-D	3.7 ± 2.4	5	TS3	5.6 ± 1.6	12	TH	36.1 ± 6.6
1	A Sep-D + acid	4.1 ± 2.6	6	TH	14.1 ± 2.8	12	TS1	31.0 ± 3.6
2	A	32.7 ± 4.5	6	TS1	13.4 ± 1.6	12	OSI	26.6 ± 12.1
2	A Acid	21.5 ± 2.5	6	TS2	15.0 ± 1.5			
2	B	47.9 ± 2.6	6	TS3	7.1 ± 1.6			
2	B Acid	52.2 ± 1.0	7	TH	25.2 ± 6.6			
2	B Sep-S + acid	34.4 ± 3.6	7	TS1	25.6 ± 4.5			
3	A	49.9 ± 3.2	7	TS2	26.3 ± 6.4			
3	B	59.1 ± 3.7	7	TS3	13.1 ± 3.8			
3	B Sep-S	40.5 ± 7.7	8	TH	24.8 ± 4.7			
3	C	32.3 ± 5.7	8	TS1	32.9 ± 5.0			
3	C Sep-D	11.7 ± 3.1	8	TS2	27.3 ± 2.2			
4	A	39.9 ± 6.3	8	TS3	23.4 ± 4.2			
4	A Sep-D	6.2 ± 1.2	9	TH	24.5 ± 4.6			
4	A Sep-S	26.5 ± 4.9	9	TS1	11.1 ± 4.1			
4	A MF thin	6.4 ± 2.2	10	TH	28.4 ± 9.9			
4	A MF thick	45.3 ± 2.7	10	TS1	21.2 ± 5.4			

[1]A, B, C: digestate source/biogas plan ID; Sep: separated, D: separated with decanter centrifuge, S: separated with screw press, Dis: disrupted, acid: acidified with 1 kg H_2SO_4 t^{-1} digestate, MF: microfiltration. All digestates in group i was applied manually with a watering can fitted with a hose to mimic trailing hose application.

[2]TH: trailing hose, TS*X*: Trailing shoe boom (*X* = model 1, 2 or 3), OSI: open slot injection. All digestate in group ii and iii was applied with an experimental tanker, as described in detail in the supporting material section S1.

the average coefficient of variation in trials 1–4 was 11 %, whereas the average coefficient of variation of the separated digestates in trials 1–4 was 31 %. As a fraction of applied total ammoniacal nitrogen, average standard deviation among replicates for separated digestates was 3.5 %, similar to the overall average of 4.1 %, despite much lower emission. With nearly constant total ammoniacal nitrogen application, this result implies that random error in dynamic flux chamber emission measurements is mostly independent of the magnitude of emission, at least partially explaining the high coefficient of variation values. Also, a larger variability in the exposed surface area after digestate application probably contributed, as slurry and digestate with a low dry matter have been found in previous studies to potentially spread out more on the soil surface compared to higher dry matter slurry (Pedersen et al., 2021).

3.2. Separation and acidification

Acidification with 1 kg H_2SO_4 t^{-1} digestate was only effective at reducing emission in some cases. Acidification effectively lowered the cumulative ammonia emissions from digestate A (by 33 % relative to untreated, p-value = 0.012 from Tukey's test), but there was no clear effect of acidification for disrupted digestate A (mean emission 18 % lower relative to disrupted A, p-value = 0.68) (Tables 4 and 5). The pH reductions measured in the field due to the acidification of these two treatments were 0.9 for digestate A and 1.6 for disrupted digestate A (Table 3). There was no reduction detected in cumulative ammonia emissions from the acidified separated digestate A, even though pH dropped by 1.4 units (mean emission 12 % lower relative to separated digestate A, p-value = 1.0). Emissions from the liquid fraction of separated digestate A was already low, presumably due to low dry matter content (Table 3), so any additional reduction from acidification was difficult to detect given the number of replicates and sensitivity of the chamber measurements. Acidification of digestate B or the liquid fraction of digestate B did not lower emissions at the given sulfuric acid application rate (mean emission 10 % higher emissions from acidified digestate B relative to untreated digestate B, p-value = 0.91) (Tables 4 and 5), likely due to a high buffer capacity of the digestate. Acidification of digestate B only lowered pH by 0.4 units, leaving the pH of the acidified digestate B at 7.7 (Table 3). Results do not support the low-dose acidification hypothesis c), as there were cases with both high and low dry matter where the low dose of acid did not result in any emission reduction. The lack of reductions due to acidification in low dry matter digestates are consistent with the findings by Wagner et al. (2021) who did not find any significant effect of acidification for low dry matter digestates, even when using higher amounts of concentrated sulfuric

Table 5
Marginal means of ammonia emissions by digestate source × digestate treatment or application method as percentage of applied total ammoniacal nitrogen. Treatments not sharing a letter within a column differ according to Tukey's HSD test (p-value <0.05). Standard error estimates were 3.3–4.2 % for 1–4, 2.9–3.0 % for 5–10, and 2.4 % of applied total ammoniacal nitrogen for trials 11–12. Replication was 3 chambers in group i and 4 in groups ii and iii.

Trials 1-4 Treatment[1]		Trials 5-10 Treatment[2]		Trials 11-12 Treatment[2]	
Group i		Group ii		Group iii	
A	39.4 d	TH	21.5 a	TH	39.4 a
A Acid	26.5 b	TS1	19.3 a	TS1	34.0 ab
A Dis	39.5 bcde	TS2	20.8 a	OSI	27.0 b
A Dis + acid	32.2 bcd	TS3	12.7 b		
A Sep-D	6.6 a				
A Sep-D + acid	7.4 a				
A Sep-S	26.4 bc				
A MF thin	6.3 a				
A MF thick	45.3 def				
B	51.9 ef				
B Acid	57.3 f				
B Sep-S	32.2 bcd				
B Sep-S + acid	39.4 cd				
C	24.1 b				
C Sep-D	3.5 a				

[1]A, B, C: digestate source/biogas plan ID; Sep: separated, D: separated with decanter centrifuge, S: separated with screw press, Dis: disrupted, acid: acidified with 1 kg H_2SO_4 t^{-1} digestate, MF: microfiltration. All digestates in group i was applied manually with a watering can fitted with a hose to mimic trailing hose application.

[2]TH: trailing hose, TS*X*: Trailing shoe boom (*X* = model 1, 2 or 3), OSI: open slot injection. All digestate in group ii and iii was applied with an experimental tanker, as described in detail in the supporting material section S1.

acid than used in the present study.

Reductions in pH and emission due to acidification were within reported values (Fangueiro et al., 2015; Pedersen and Hafner, 2023). Within literature, there is a large variation in reduction efficiencies of acidification, which is attributed to differences in acid type, amount used, timing of acidification compared to application, and digestate or slurry properties. Nyord et al. (2021) observed a larger reduction in digestate pH by a fixed acid amount compared to raw slurry, presumably due to the initial pH being farther from the pK_a value of the H_2CO_3/HCO_3^- equilibrium ($pK_a = 6.4$) where the buffer capacity is expected to be highest.

The dry matter reduction in the liquid fractions compared with the unseparated digestates was lower (38 % for decanter centrifuge and 22 % for screw press, Table 3) than average literature values for digestate (50 % for decanter centrifuge and 29 % for screw press (Pedersen et al., 2022)). The reason for this is unknown, but it is assumed that it is due to the generally large variation in separation efficiency (Pedersen et al., 2022), and low number of digestates that were separated in the present study as well as the high average dry matter of these digestates. All separations resulted in a lower viscosity consistency coefficient and higher flow behaviour index, closer to the behaviour of a Newtonian fluid (Table S1). This trend is explained by the reduction in dry matter, since less interactions among particles occur when less particles are present, resulting in a lower flow resistance for the fluid (Schneider and Gerber, 2020). Separated digestate also had lower amount of large (>2 mm) particles in the liquid fraction compared to the unseparated digestate (Table S1).

There were significantly lower cumulative ammonia emissions from the liquid fraction after separation compared to the unseparated digestate in all cases (p-value = <0.0001–0.0041), but the emission reduction efficiency varied greatly. Separation by screw press yielded the lowest reductions (33 % and 38 %), whereas decanter centrifuge yielded the highest cumulative ammonia emission reductions (83 % and 86 %) (Table 5), which is in line with data from literature (Pedersen et al., 2022). A second separation step of the liquid fraction from screw press in a micro filter (trial 4) further reduced the emission significantly, both compared to the unseparated digestate A (p-value <0.0001) and the liquid fraction from the screw press (p-value <0.0001). However, the overall reduction from unseparated digestate A to the thin fraction after micro filtration (84 %) is equal to the reduction obtained by separation of digestate A by decanter centrifuge alone (83 %) (Table 5). The separation hypothesis c) is supported by these results; a reduction was observed on average, but with varying efficiencies.

A recent literature review with a model calculation showed that the total ammonia emissions (sum of emissions from liquid and solid fraction both stored and applied) were reduced for high dry matter digestate after separation compared to the unseparated digestate if the liquid fraction was surface applied and the solid fraction was incorporated within 4 h after application (Pedersen et al., 2022). However, it needs to be emphasized that more experimental data is required for an overall assessment of the total ammonia reduction potential after separation of digestate. There is a lack of measurements of ammonia emissions from storage and application of the solid fraction.

3.3. Application techniques

There were no significant differences between the cumulative emission after application with trailing hose or the two conventional designs of trailing shoe (TS1 and TS2) (p-value = 0.44–0.98) (Table 5), so hypothesis a) is not supported. Generally, application by trailing shoe is reported to provide a lower cumulative emission than trailing hose on average, but the results from individual trials vary greatly, and in some individual trials higher emission from trailing shoe were measured compared to trailing hose (Andersson et al., 2023; Häni et al., 2016; Pedersen et al., 2020; Smith et al., 2000; Webb et al., 2010). The variation in the relative difference between trailing hose and trailing shoe are attributed to soil conditions (Andersson et al., 2023; Pedersen et al., 2020; Smith et al., 2000), as trailing shoe performs best if soil conditions allow for the creation of a furrow to contain the slurry or digestate. In two individual trials (9 and 10), the cumulative emissions were 55 and 25 % lower from one design of trailing shoe (TS1) compared to trailing hose, whereas the differences were negligible and in both directions in the other trials (Table 4). The reason for this is unknown, as both the digestate properties and soil conditions for trials 9 and 10 were within the conditions of the other trials with the same two application techniques (trials 5–8 and 11–12) (Tables 2 and 3). It is speculated that it was caused by a deeper furrow being created by the trailing shoe in these trials, perhaps due to slight differences in soil hardness, as the initial fluxes are lower than trailing hose (Fig. 1, trials 9 and 10) compared to the other trials with both techniques (Fig. 1, trials 5–8).

In contrast to the two conventional models of trailing shoe (TS1 and TS2), application with the modified trailing shoe (TS3) clearly reduced cumulative emission compared to trailing hose but also compared to the two other trailing shoe designs (TS1 and TS2) (34–41 %, p-value <0.0001–0.0023) (Table 5), supporting hypothesis b). The harrowing tine of the trailing shoe (TS3) mixed the soil and digestate (supporting materials section S2) and left a lower amount of liquid digestate exposed to the air after application, which is most likely the cause of the lower emissions, as evidenced by lower flux immediately after application (Fig. 1). The overall ammonia reduction by application with the modified trailing shoe (TS3) compared to the other three band application methods (TH, TS1, and TS2) is a large improvement compared to the former prototype tested in McCollough et al. (2022) where no average reduction was observed. It is important to note that the soils in McCollough et al. (2022) were heavly crusted, which was not the case in the present study. For trials 5–7, the cumulative emissions from digestate applied by the modified trailing shoe (TS3) were reduced by a minimum of 46 % compared to the three other band application techniques, whereas the reductions were lower in trial 8 (6–29 %) (Table 4). The high crop height in trial 8 (Table 2) may have reduced the effect of the harrowing tine by decreased contact with the soil (Table 2 and supporting materials S2). Furthermore, the soil-water content during trial 8 was lower (0.10 g g^{-1}) compared to trials 5–7 (0.23–0.25 g g^{-1}) (Table 2), which might have reduced the effect of the harrowing tine. To assess whether the new design should be used, the extra draft force (and thereby energy consumption) as well as any crop damage should be assessed.

In trials 11 and 12, open slot injection reduced emissions by 31 % compared to trailing hose (p-value = 0.0037, Table 5). There was no significant difference between trailing shoe (TS1) and trailing hose (p-value = 0.27) or trailing shoe (TS1) and open slot injection (p-value = 0.11) in these trials (Table 5). The reported differences between trailing hose and open slot injection varied greatly in literature; some studies found higher reductions (Huijsmans et al., 2001; Misselbrook et al., 2002; Nyord et al., 2008), whereas others reported similar results as the present work (Hansen et al., 2003; Pedersen and Nyord, 2023). It has been shown that the emissions after application by open slot injection are highly dependent on the depth and width of the furrow created in the soil (Hansen et al., 2003; Nyord et al., 2008). Pictures taken after application show that the slits did not contain all the applied digestate in the present study (supporting materials section S2). Interestingly, the two studies with similar differences between trailing hose and open slot injection (Hansen et al., 2003; Pedersen and Nyord, 2023) as the present study are both Danish studies, hence it can be speculated if the performance of the open slot injection also depends on region where the trials are performed due to e.g., soil type effects, management practices, and difference in models of machinery. In trial 12, the open slot injection has a standard deviation higher than any other treatments in the present study (Table 4), probably related to inconsistent and non-ideal open slot injection, i.e., the slit created could not contain all the digestate, leaving more at the soil surface in at least some locations, giving a higher emission potential (Fig. S11). Open slot injection can efficiently reduce ammonia emissions compared to surface applied digestate or slurry, but other studies have in some trials found higher nitrous oxide emissions from open slot injection compared to trailing hose (e.g., Maris et al., 2021; Nyameasem et al., 2023). If ammonia mitigation leads to a reduced input of mineral nitrogen to the field this can to some extend compensate for an increased nitrous oxide emission potential from open slot injection. Both the potential risks and benefits should be considered when selecting the best application method.

4. Conclusions

Field trials assessing different low ammonia emission treatments and application techniques for digestate showed a lack of consistency in mitigation effects even within a limited range of soil texture and crop canopy types. Both separation and acidification reduced emissions in some cases, whereas in other cases no effects were observed. On average, application by trailing shoe yielded similar results as application by trailing hose, although in individual trials lower emissions were observed from trailing shoe application. In contrast, digestate application by a new design of trailing shoes combined with a harrowing tine consistently lowered ammonia emissions in four trials. It is important to further investigate the sources of variation in reduction efficiency to understand when a reduction can be expected and when using a certain treatment or application technique will not yield any benefits. The optimal application technique and possible digestate treatment must be assessed based on the digestate properties and soil conditions at the time of application; it is not possible to recommend one universal approach that can be applied for all application scenarios. The present work focused only on ammonia emissions, but other factors need to be considered when deciding on the optimal management for a specific situation, such as possible trade-offs with other emitting gases (e.g., nitrous oxide and volatile organic compounds), cost of treatment or application technique as well as width of machinery and power consumption.

CRediT authorship contribution statement

Johanna Pedersen: Writing – review & editing, Writing – original draft, Visualization, Validation, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation. **Sasha D. Hafner:** Writing – review & editing, Visualization, Supervision, Methodology, Formal analysis, Data curation. **Cristiane Romio:** Writing – review & editing, Investigation, Formal analysis. **Andreas S. Pacholski:** Writing – review & editing.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.aeaoa.2026.100415.

Data availability

Most data and associated scripts can be found in the following GitHub release 10.5281/zenodo.14849239 for reproducibility.

References

Ambrose, H.W., Dalby, F.R., Feilberg, A., Kofoed, M.V.W., 2023. Additives and methods for the mitigation of methane emission from stored liquid manure. Biosyst. Eng. 229, 209–245. https://doi.org/10.1016/j.biosystemseng.2023.03.015.

Amon, B., Kryvoruchko, V., Amon, T., Zechmeister-Boltenstern, S., 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agric. Ecosyst. Environ. 112 (2–3), 153–162. https://doi.org/10.1016/j.agee.2005.08.030.

Andersson, K., Delin, S., Pedersen, J., Hafner, S.D., Nyord, T., 2023. Ammonia emissions from untreated, separated and digested cattle slurry – effects of slurry type and application strategy on a Swedish clay soil. Biosyst. Eng. 226, 194–208. https://doi.org/10.1016/j.biosystemseng.2023.01.012.

APHA, 2017. Standard Methods for the Examination of Water and Wastewater, 23rd ed. American Public Health Association.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Software 67 (1). https://doi.org/10.18637/jss.v067.i01.

Beltran, I., van der Weerden, T.J., Alfaro, M.A., Amon, B., de Klein, C.A.M., Grace, P., Hafner, S., Hassouna, M., Hutchings, N., Krol, D.J., Leytem, A.B., Noble, A., Salazar, F., Thorman, R.E., Velthof, G.L., 2021. DATAMAN: a global database of nitrous oxide and ammonia emission factors for excreta deposited by livestock and land-applied manure. J. Environ. Qual. 50 (2), 513–527. https://doi.org/10.1002/jeq2.20186.

Fangueiro, D., Hjorth, M., Gioelli, F., 2015. Acidification of animal slurry–a review. J. Environ. Manag. 149, 46–56. https://doi.org/10.1016/j.jenvman.2014.10.001.

García, P., Støckler, A.H., Feilberg, A., Kamp, J.N., 2024. Investigation of non-target gas interferences on a multi-gas cavity ring-down spectrometer. Atmos. Environ. X 22. https://doi.org/10.1016/j.aeaoa.2024.100258.

Georgacakis, D., Sievers, D.M., Iannotti, E.L., 1982. Buffer stability in manure digesters. Agric. Wastes 4, 427–441. https://doi.org/10.1016/0141-4607(82)90038-5.

Hafner, S.D., Kamp, J.N., Pedersen, J., 2024. Experimental and model-based comparison of wind tunnel and inverse dispersion model measurement of ammonia emission from field-applied animal slurry. Agric. For. Meteorol. 344. https://doi.org/10.1016/j.agrformet.2023.109790.

Hafner, S.D., Pedersen, J., Fuß, R., Kamp, J.N., Dalby, F.R., Amon, B., Pacholski, A., Adamsen, A.P.S., Sommer, S.G., 2025. Improved tools for estimation of ammonia emission from field-applied animal slurry: refinement of the ALFAM2 model and database. Atmos. Environ. 340. https://doi.org/10.1016/j.atmosenv.2024.120910.

Hansen, M.N., Sommer, S.G., Madsen, N.P., 2003. Reduction of ammonia emission by shallow slurry injection injection efficiency and additional energy demand. J. Environ. Qual. 32, 1099–1104.

Hassouna, M., van der Weerden, T.J., Beltran, I., Amon, B., Alfaro, M.A., Anestis, V., Cinar, G., Dragoni, F., Hutchings, N.J., Leytem, A., Maeda, K., Maragou, A., Misselbrook, T., Noble, A., Rychla, A., Salazar, F., Simon, P., 2023. DATAMAN: a global database of methane, nitrous oxide, and ammonia emission factors for livestock housing and outdoor storage of manure. J. Environ. Qual. 52 (1), 207–223. https://doi.org/10.1002/jeq2.20430.

Huijsmans, J.F.M., Hol, J.M.G., Hendriks, M.M.W.B., 2001. Effect of application technique, manure characteristics, weather and field conditions on ammonia volatilization from manure applied to grassland. NJAS - Wageningen J. Life Sci. 49 (4), 323–342. https://doi.org/10.1016/s1573-5214(01)80021-x.

Häni, C., Sintermann, J., Kupper, T., Jocher, M., Neftel, A., 2016. Ammonia emission after slurry application to grassland in Switzerland. Atmos. Environ. 125, 92–99. https://doi.org/10.1016/j.atmosenv.2015.10.069.

Jensen, O.N., Beyer, M., Sorensen, M.K., Kreimeyer, M., Nielsen, N.C., 2021. Fast and accurate quantification of nitrogen and phosphorus constituents in animal slurries using NMR sensor technology. ACS Omega 6 (27), 17335–17341. https://doi.org/10.1021/acsomega.1c01441.

Kamp, J.N., Chowdhury, A., Adamsen, A.P.S., Feilberg, A., 2019. Negligible influence of livestock contaminants and sampling system on ammonia measurements with cavity ring-down spectroscopy. Atmos. Meas. Tech. 12 (5), 2837–2850. https://doi.org/10.5194/amt-12-2837-2019.

Kamp, J.N., Hafner, S.D., Huijsmans, J., van Boheemen, K., Götze, H., Pacholski, A., Pedersen, J., 2024. Comparison of two micrometeorological and three enclosure methods for measuring ammonia emission after slurry application in two field experiments. Agric. For. Meteorol. 354. https://doi.org/10.1016/j.agrformet.2024.110077.

Khoshnevisan, B., Duan, N., Tsapekos, P., Awasthi, M.K., Liu, Z., Mohammadi, A., Angelidaki, I., Tsang, D.C.W., Zhang, Z., Pan, J., Ma, L., Aghbashlo, M., Tabatabaei, M., Liu, H., 2021. A critical review on livestock manure biorefinery technologies: sustainability, challenges, and future perspectives. Renew. Sustain. Energy Rev. 135. https://doi.org/10.1016/j.rser.2020.110033.

Lado, M., Ben-Hur, M., Shainberg, I., 2007. Clay mineralogy, ionic composition, and pH effects on hydraulic properties of depositional seals. Soil Sci. Soc. Am. J. 71 (2), 314–321. https://doi.org/10.2136/sssaj2006.0169.

Lenth, R.V., 2024. Emmeans: estimated marginal means, aka lest-squares means. In: R Package. https://doi.org/10.32614/CRAN.package.emmeans version 1.10.6.

Maris, S.C., Abalos, D., Capra, F., Moscatelli, G., Scaglia, F., Cely Reyes, G.E., Ardenti, F., Boselli, R., Ferrarini, A., Mantovi, P., Tabaglio, V., Fiorini, A., 2021. Strong potential of slurry application timing and method to reduce N losses in a permanent grassland. Agric. Ecosyst. Environ. 311. https://doi.org/10.1016/j.agee.2021.107329.

McCollough, M.R., Pedersen, J., Nyord, T., Sørensen, P., Melander, B., 2022. Ammonia emissions, exposed surface area, and crop and weed responses resulting from three post-emergence slurry application strategies in cereals. Agronomy 12 (2441). https://doi.org/10.3390/agronomy12102441.

Misselbrook, T.H., Smith, K.A., Johnson, R.A., Pain, B.F., 2002. Slurry application techniques to reduce ammonia emissions: results of some UK field-scale experiments. Biosyst. Eng. 81 (3), 313–321. https://doi.org/10.1006/bioe.2001.0017.

Morin, J., Benyamini, Y., Michaeli, A., 1981. The effect of raindrop impact on the dynamics of soil surface crusting and water movement in the profile. Jounral of Hydrology 52, 321–335.

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

Möller, K., Müller, T., 2012. Effects of anaerobic digestion on digestate nutrient availability and crop growth: a review. Eng. Life Sci. 12 (3), 242–257. https://doi.org/10.1002/elsc.201100085.

Nkoa, R., 2013. Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. Agron. Sustain. Dev. 34 (2), 473–492. https://doi.org/10.1007/s13593-013-0196-z.

Nyameasem, J.K., Ruser, R., Kluß, C., Essich, C., Zutz, M., ten Huf, M., Buchen-Tschiskale, C., Flessa, H., Olfs, H.W., Taube, F., Reinsch, T., 2023. Effect of slurry application techniques on nitrous oxide emission from temperate grassland under varying soil and climatic conditions. Grass Forage Sci. 78 (3), 338–358. https://doi.org/10.1111/gfs.12612.

Nyord, T., Hafner, S., Adamsen, A.P.S., Sommer, S.G., 2021. *Ammoniakfordampning Fra Forsuret Gylle Ved Udbringing Med Slæbeslange (EN: Ammonia Emission from Acidified Slurry Applied with Trailing Hoses)* (Advisory Report from DCA - Danish Centre for Food and Agriculture. Aarhus University, submitted, 15-02-2021). https://pure.au.dk/portal/files/211563336/Forsuring_150221.pdf.

Nyord, T., Søgaard, H.T., Hansen, M.N., Jensen, L.S., 2008. Injection methods to reduce ammonia emission from volatile liquid fertilisers applied to growing crops. Biosyst. Eng. 100 (2), 235–244. https://doi.org/10.1016/j.biosystemseng.2008.01.013.

Pedersen, J., Andersson, K., Feilberg, A., Delin, S., Hafner, S., Nyord, T., 2021. Effect of exposed surface area on ammonia emissions from untreated, separated, and digested cattle manure. Biosyst. Eng. 202, 66–78. https://doi.org/10.1016/j.biosystemseng.2020.12.005.

Pedersen, J., Feilberg, A., Kamp, J.N., Hafner, S., Nyord, T., 2020. Ammonia emission measurement with an online wind tunnel system for evaluation of manure application techniques. Atmos. Environ. 230. https://doi.org/10.1016/j.atmosenv.2020.117562.

Pedersen, J., Hafner, S.D., 2023. Ammonia emissions after field application of anaerobically digested animal slurry: literature review and perspectives. Agric. Ecosyst. Environ. 357. https://doi.org/10.1016/j.agee.2023.108697.

Pedersen, J., Hafner, S.D., Adamsen, A.P.S., 2022. Effectiveness of mechanical separation for reducing ammonia loss from field-applied slurry: assessment through literature review and model calculations. J. Environ. Manag. 323, 116196. https://doi.org/10.1016/j.jenvman.2022.116196.

Pedersen, J., Hafner, S.D., Pacholski, A., Karlsson, V.I., Rong, L., Labouriau, R., Kamp, J.N., 2024. Evaluation of optimized flux chamber design for measurement of ammonia emission after field application of slurry with full-scale farm machinery. Atmos. Meas. Tech. 17 (14), 4493–4505. https://doi.org/10.5194/amt-17-4493-2024.

Pedersen, J., Hafner, S.D., Pacholski, A.S., 2026. Methodological factors affecting ammonia emission measurement with flux chambers from field-applied biogas digestate slurry (technical note). Atmos. Environ. X 29. https://doi.org/10.1016/j.aeaoa.2025.100408.

Pedersen, J., Nyord, T., 2023. Effect of low-dose acidification of slurry digestate on ammonia emissions after field application (Short communication). Atmos. Environ. X 17. https://doi.org/10.1016/j.aeaoa.2023.100205.

Rodhe, L., Etana, A., 2005. Performance of slurry injectors compared with band spreading on three Swedish soils with ley. Biosyst. Eng. 92 (1), 107–118. https://doi.org/10.1016/j.biosystemseng.2005.05.017.

Romio, C., Ward, A.J., Møller, H.B., 2024. Characterization and valorization of biogas digestate and derived organic fertilizer products from separation processes. Front. Sustain. Food Syst. 8. https://doi.org/10.3389/fsufs.2024.1415508.

Schneider, N., Gerber, M., 2020. Rheological properties of digestate from agricultural biogas plants: an overview of measurement techniques and influencing factors. Renew. Sustain. Energy Rev. 121. https://doi.org/10.1016/j.rser.2020.109709.

Simmons, G.F., 1996. Calculus with Analytic Geometry. McGraw-Hill Education.

Smith, K.A., Jackson, D.R., Misselbrook, T.H., Pain, B.F., Johnson, R.A., 2000. PA—Precision agriculture. J. Agric. Eng. Res. 77 (3), 277–287. https://doi.org/10.1006/jaer.2000.0604.

Sommer, S.G., Misselbrook, T.H., 2015. A review of ammonia emission measured using wind tunnels compared with micrometeorological techniques. Soil Use Manag. 32 (S1), 101–108. https://doi.org/10.1111/sum.12209.

Sutton, M.A., Oenema, O., Erisman, J.W., Leip, A.H.v.G., Winiwater, W., 2011. Too much of a good thing. Nature Comment 472.

Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F., Casu, F., Rulli, M., Galloway, J.N., Leip, A., Erisman, J.W., Robinson, T.P., Steinfeld, H., Gerber, P.J., 2020. Nitrogen emissions along global livestock supply chains. Nat. Food 1 (7), 437–446. https://doi.org/10.1038/s43016-020-0113-y.

Wagner, C., Nyord, T., Vestergaard, A.V., Hafner, S.D., Pacholski, A.S., 2021. Acidification effects on in situ ammonia emissions and cereal yields depending on slurry type and application method. Agriculture 11 (11). https://doi.org/10.3390/agriculture11111053.

Webb, J., Pain, B., Bittman, S., Morgan, J., 2010. The impacts of manure application methods on emissions of ammonia, nitrous oxide and on crop response—A review. Agric. Ecosyst. Environ. 137 (1–2), 39–46. https://doi.org/10.1016/j.agee.2010.01.001.

Weiland, P., 2009. Biogas production: current state and perspectives. Appl. Microbiol. Biotechnol. 85 (4), 849–860. https://doi.org/10.1007/s00253-009-2246-7.

Wyer, K.E., Kelleghan, D.B., Blanes-Vidal, V., Schauberger, G., Curran, T.P., 2022. Ammonia emissions from agriculture and their contribution to fine particulate matter: a review of implications for human health. J. Environ. Manag. 323, 116285. https://doi.org/10.1016/j.jenvman.2022.116285.

Yan, X., Ying, Y., Li, K., Zhang, Q., Wang, K., 2024. A review of mitigation technologies and management strategies for greenhouse gas and air pollutant emissions in livestock production. J. Environ. Manag. 352, 120028. https://doi.org/10.1016/j.jenvman.2024.120028.

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