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ORIGINAL ARTICLE

Ammonia emissions after trailing hose application of digestates and cattle slurry

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Abstract Anaerobic digestion (AD) is a promising technique for waste management, producing energy and contributing to nutrient recycling in agroecosystems. While digestates have higher plant-available nutrient contents, they may be prone to increased ammonia (NH_3) losses due to elevated pH values and ammonium contents. This study investigates NH_3 emissions from an agricultural digestate consisting of cattle slurry, solid manure and food processing waste (SLA) and a municipal organic waste digestate (LID) applied alongside untreated cattle slurry (SLU) as a reference to maize and cereals with a trailing hose. Values of dry matter, ammoniacal nitrogen (TAN) and pH were higher for SLA and LID than for SLU. Emissions were determined in five application events over three years with NH_3 concentration measurements using an impinger system combined with backward Lagrangian Stochastic dispersion modeling. On average, 31%, 42%, and 43% of applied TAN volatilized as NH_3 from SLU, SLA, and LID, respectively. Despite being higher from the treatments, these differences were not statistically significant. Therefore, it remains unclear whether digestates differ in NH_3 emissions from untreated slurry. This topic needs further investigation due to the increasing use of AD.

Keywords Ammonia · Agricultural digestate · Manure · Non-agricultural digestate

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Introduction

Anaerobic digestion (AD) in biogas plants offers a promising solution for recycling livestock manure and municipal organic waste. It produces energy, contributes to nutrient recycling and reduces CO_2 emissions by replacing fossil fuels (Weiland 2010). Digestates present a viable alternative to mineral nitrogen (N) fertilizers due to their relatively high total ammoniacal N (TAN) content, which often leads to greater N use efficiencies of digestates than for untreated slurries (Gutser et al. 2005). However, the increases in TAN and pH due to the degradation of organic matter during AD (Pedersen and Hafner 2023; Weiland 2010) enhance the potential of ammonia (NH_3) volatilization after field application of digestates. On the other hand, the degradation of organic matter during AD decreases the dry matter (DM) content of the digestate, which reduces its viscosity and increases soil infiltration. Enhanced soil infiltration diminishes emissions (Hafner et al. 2025) which counteracts a rise in emissions due to elevated values of TAN and pH (Pedersen and Hafner 2023).

Substantially fewer emission data from digestates after field application are available than for NH_3 losses from untreated slurry. There is a large variation in emission data due to differences in pedoclimatic conditions, crops and agricultural management (Hafner et al. 2018, 2024; Sintermann et al. 2012). Hafner et al. (2018, 2025) established the ALFAM2 database, which includes data on NH_3 emissions mainly from experiments in temperate climates. Data on emissions from digestates are underrepresented in this database. Pedersen and Hafner (2023) presented 20 datasets with emissions from the application of digestates whereof 8 exclusively consisted of livestock slurry and 12 of livestock slurries with co-substrates.

Digestion techniques, categorized as wet or dry digestion, affect the properties of digestates. Wet digestion, prevalent in agricultural biogas plants, uses continuously stirred reactors and is suitable for substrates with less than 10% dry matter. In contrast, dry digestion processes solid substrates with 15–35% dry matter and is commonly used for municipal or industrial waste (Weiland 2010). If substantial amounts of solid substrates are co-digested with slurry, contents of TAN and DM as well as pH are higher than in untreated and unamended slurry, which enhances the risk of NH_3 emissions after application (Hafner et al. 2018; Pedersen and Hafner 2023). The prediction of NH_3 emissions from digestates is challenging due to the large variation in feedstock and processing conditions (Pedersen and Hafner 2023). Variations can be due to different types of livestock slurry and solid manure from a number of livestock categories (e.g. cattle, pig, poultry) and various co-substrates which include many types of municipal or industrial organic waste. The types of feedstock materials and their percentage fed to an AD plant may vary largely. In addition, postprocessing of the digestates, such as solid–liquid separation, influences their characteristics and emission levels (Hjorth et al. 2010). Overall, there is a plethora of possible digestate types which may vary in emission after field application. In view of the wide variation in digestate properties, an improved understanding of their NH_3 emissions is needed to develop optimized application and emission reduction strategies. This is necessary to ensure their improved N use efficiency and reduce environmental impacts (Nkoa 2014).

The NH_3 emissions are further affected by various factors, especially the application method in the field (Webb et al. 2010) and weather conditions, specifically wind speed (Sommer et al. 1997; Maris et al. 2021) and temperature (Hafner et al. 2018). Banded application, e.g. using a trailing hose, reduces emissions by about 30% to 50% compared to broadcasting (Hafner et al. 2018, 2025; Webb et al. 2010).

The objective of this study was to compare NH_3 emissions from two different digestates with untreated cattle slurry as a reference applied to arable crops using a trailing hose and thus to increase the knowledge of emission levels occurring after field application of digestates.

Materials and methods

Study site

The research was carried out on two organically managed arable fields in Switzerland, located in Wallbach (47° 33′ 54.0″ N 7° 53′ 08.2″ E, 334 m a.s.l. and 47° 34′ 10.4″ N 7° 53′ 50.5″ E, 342 m a.s.l.). The soil was classified as Haplic Luvisol with a silt loam texture and a pH of 6. NH_3 emissions were monitored during three cropping seasons in the first days after each of the five application events in silage maize (2 applications), winter wheat (2 applications), and winter barley (1 application), with crop canopy height always below 45 cm.

A weather station (47° 34′ 10.8″ N 7° 53′ 54.4″ E, Agrometeo, Meteoschweiz) in the vicinity of the plots recorded air temperature, rainfall, and solar radiation on an hourly basis. Data for wind speed and wind direction were obtained from another nearby weather station (47°34′ 19.9″ N 7° 52′ 40.5″ E, SwissMetNet, Meteoschweiz). The average air temperature at 2 m height ranged between 7 and 20 °C over 54 h after the application events. It was about 5–10 °C higher during the applications in 2018 than in 2019 and 2020 (Supplementary material S1). Mean wind speeds ranged between 1.7 and 2.7 m s^{-1}, with a maximum of 4.1 m s^{-1} during the initial 2 h after each application. Limited rainfall occurred during two of the measurement periods, with 0.8 and 3.4 mm over 54 h recorded during the first application in 2018 and the second application in 2019, respectively (Supplementary material, S1). During the second application in 2018, cumulative rainfall was 24.4 mm over 54 h, with the first rain occurring 30 h after slurry spreading.

Experimental design

Untreated cattle slurry (SLU), agricultural digestate based mainly on anaerobically digested slurry (SLA) and municipal organic waste liquid digestate (LID) were used for the experiments. SLU was obtained from an organic farm (Aemethof, Densbüren, Switzerland) in 2018 and 2019. In 2020, slurry from a conventional farm (Paul Frey, Asp, Switzerland) was used because the slurry employed previously was highly diluted with rainwater. SLA was produced from cattle slurry, horse manure, and about 20% of food processing waste such as coffee grounds and herb extraction residues as co-substrates under thermophilic conditions (53 °C) for 60 days on average in an agricultural biogas plant using wet fermentation (Synfarms, Densbüren). LID was obtained from an industrial biogas plant (Biopower, Pratteln, Switzerland) utilizing a Kompogas® system, which employs dry fermentation of garden waste, municipal organic waste and food waste from retailers at 55 °C during an average of 15 days. A screw press for SLA and a centrifuge for LID were used for solid–liquid separation of the digestates. LID (10.4%) and SLA (5.4%) had a higher dry matter content than SLU (4.2%; Table 1). Digestates had a higher pH (7.8–7.9) than SLU (7.2). The N contents were more than twice as high for SLA (4.0 g/kg) and LID (5.0 g/kg) than for SLU (1.8 g/kg). Manure application rates were adjusted to an equal amount of total N applied with each type. Hence, the application rates of SLA and LID were less than half of the applied volumes of SLU.

Three plots measuring 30×30 m in 2018 and 20×20 m in 2019 and 2020 were defined within the arable fields. The slurry and the digestates were applied with a trailing hose directly onto the soil surface at target rates of 70 kg N ha^{-1} per application for maize and wheat, and 60 kg N ha^{-1} for barley. Spreading occurred on the same day, but with a slight time lag (<20 min) between spreading the different manures due to the need to load and transport the slurry between the field applications. Measurements

Table 1 Characteristics of untreated cattle slurry (SLU), agricultural digestate (SLA) and municipal organic waste digestate (LID) for each application event: dry matter (DM), pH and TAN (Total Ammoniacal Nitrogen)

Year	Application event	Date		DM (%)	pH	TAN (g/kg)
2018	1	04.05.2018	SLU	4.3	7.2	0.9
2018	1	04.05.2018	SLA	5.7	7.9	2.1
2018	1	04.05.2018	LID	11.0	7.8	2.7
2018	2	29.05.2018	SLU	4.4	7.4	0.8
2018	2	29.05.2018	SLA	5.1	7.8	2.0
2018	2	29.05.2018	LID	14.9	7.8	3.1
2019	1	26.03.2019	SLU	4.3	7.1	1.1
2019	1	26.03.2019	SLA	4.2	7.8	2.1
2019	1	26.03.2019	LID	8.0	7.8	2.7
2019	2	16.04.2019	SLU	2.5	7.3	0.6
2019	2	16.04.2019	SLA	6.2	8.0	2.7
2019	2	16.04.2019	LID	9.6	7.8	2.9
2020	2	03.04.2020	SLU	5.7	7.1	1.1
2020	2	03.04.2020	SLA	5.8	8.0	3.3
2020	2	03.04.2020	LID	8.7	7.8	2.8
Average			SLU	4.2	7.2	0.9
Average			SLA	5.4	7.9	2.4
Average			LID	10.4	7.8	2.8

More detailed characteristics of the slurry analyses are provided in Supplementary material, S2 of NH_3 emissions were initiated immediately after each application. The NH_3 emissions were measured in parallel with an assessment of nitrous oxide emissions in a nearby field experiment, using the same slurry and digestates, application dates and crop rotation as described in Efosa et al. (2023).

Measurement of NH_3 emissions

NH_3 concentrations were measured using a Low-Cost Impinger System (LOCI; Häni et al. 2016) placed in the center of each plot, with additional measurements positioned in the same field upwind of the manured plots to capture background concentrations. Each LOCI sequentially operated seven impinger positions that ran at two heights in parallel. Impingers, each with a volume of 22 ml, were supplied with 15 mL of a 0.01 M sodium acetate buffer solution (pH = 4) as an acid trap and a few drops of strongly diluted dichlormethane solution (0.5 ml dichloromethane per 1000 ml water) to inhibit microbial activity. Ambient air was drawn through the impingers at 0.7 L min^{-1} and NH_3 dissolved into TAN, which was later analyzed (spectrophotometer, DR 2800, HACH, Colorado, United States). The sampling intervals ranged between 2 and 24 h, increasing in duration with increasing time after manure application. The total duration of all measurements was up to 66 h but differed between the individual campaigns. Cumulative emissions were determined for the time period of 54 h. Between 54 h and the end of the campaigns the concentrations over the manured plots were similar to the background concentration.

Emissions were calculated using a dispersion model. For each concentration measurement at the plot center (C_{ctr}), the measured background concentration (C_{bgd}) as well as the sum of the modeled concentration increases related to any emissions from upwind plots (sum C_{plt}) were subtracted to obtain the net NH_3 concentration related to the emissions from the corresponding source plot.

The final emission (E) for each plot and each interval was calculated as:

$$E = \frac{\left(C_{ctr} - C_{bgd} - \sum C_{plt}\right)}{D_{bLs}}$$

where D_{bLs} is the modeled dispersion factor, i.e., the concentration-to-emission ratio related to the dispersion of a tracer gas between the source and the sensor (Flesch et al. 2004). The value of D_{bLs} was calculated using the R (R Development Core Team 2021) package *bLSmodelR* (Häni 2022), which is based on the backward Lagrangian Stochastic (bLS) dispersion model (Flesch et al. 2004). The turbulence statistics required to run the model were derived from wind measurements with an ultrasonic anemometer (WindMaster™Pro, Gill Instruments Limited, Lymington, UK), located in the center of the field.

For the campaigns in 2018, we compared NH_3 emissions based on micrometeorological data recorded by the ultrasonic anemometer with modeled data using WindTrax (Fig. 1). The NH_3 emission rates were slightly higher when utilizing the modeled micrometeorological data, i.e. on average 0.19 kg NH_3–N ha^{-1} h^{-1} which corresponds to a deviation of less than 2%. Consequently, the WindTrax model was employed to estimate emissions during periods when the ultrasonic anemometer data was unavailable due to technical issues with the device.

Statistical analysis

Data analysis employing a significance level of $p < 0.05$ was performed using R based on the applications of 2018 and 2019 where replicates are available. Cumulative NH_3 emissions over 54 h, expressed in kg ha^{-1} and percentages of NH_3 volatilized from total applied TAN were subjected to one-way ANOVA.

Results

NH_3 emissions measured in the initial two hours after the application of SLU, SLA and LID ranged from 0.8 to 5.7 kg NH_3–N ha^{-1}, decreasing to 0.1–1.9 kg NH_3–N ha^{-1} in the interval 2–6 h after application (Fig. 1). Cumulative emissions over 54 h were 31%, 42% and 43% of applied TAN for SLU, SLA and LID, respectively, with emissions of SLA and LID being respectively 26% and 28% higher than of SLU. However, the differences were not statistically significant ($df = 2$, $F = 0.909$, $p = 0.437$). Also, cumulative emissions over 54 h, expressed in kg NH_3–N ha^{-1} did not exhibit a statistically significant difference ($df = 2$, $F = 2.047$, $p = 0.185$).

Michaelis–Menten dynamics illustrated the mean time courses for NH_3 emissions, with half of the

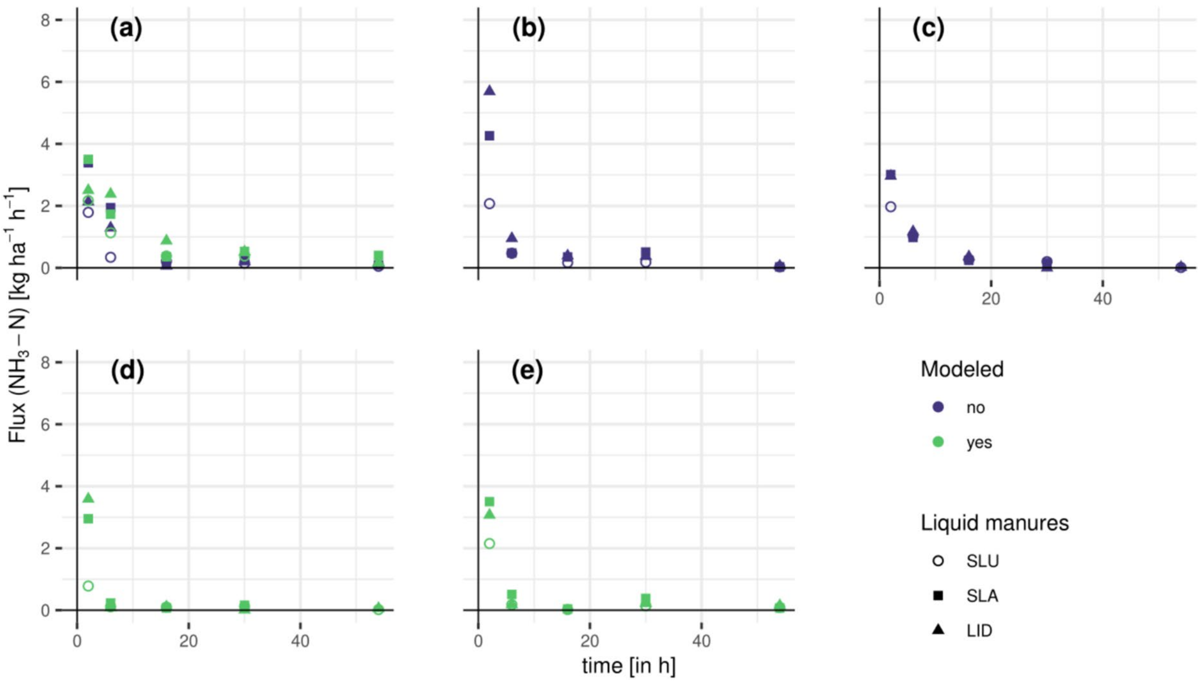

Fig. 1 NH_3–N emission rate (kg ha^{-1} h^{-1}) from cattle slurry (SLU), agricultural digestate (SLA) and municipal organic waste digestate (LID) calculated with atmospheric data from an ultrasonic anemometer (Modeled = "no") and with modeled atmospheric data (Modeled = "yes"). Panels represent one sampling campaign each and from three consecutive years: 2018: **a** 4.5.2018–6.5.2018, **b** 29.5.2018–31.5.2018, 2019: **c** 26.3.2019–28.3.2019, **d** 16.4.2019–18.4.2019 and 2020: **e** 3.4.2020–5.4.2020

maximum TAN loss occurring within 5.6 h for SLU, 4.0 h for SLA, and 3.2 h for LID (Fig. 2). The emission level over 54 h from SLU expressed as percent of TAN applied was higher by a factor of 2.0 on average (range: 1.3 to 2.7) than predicted by the ALFAM2 model (Hafner 2024). The observed emissions from SLA and LID were higher by a factor of 1.6 and 1.3, respectively, than predicted by ALFAM2. Exceptions were the first and last campaign for LID with comparable numbers for the predicted and observed emissions, and the second application in 2019, when the observed emissions from SLA and LID were lower

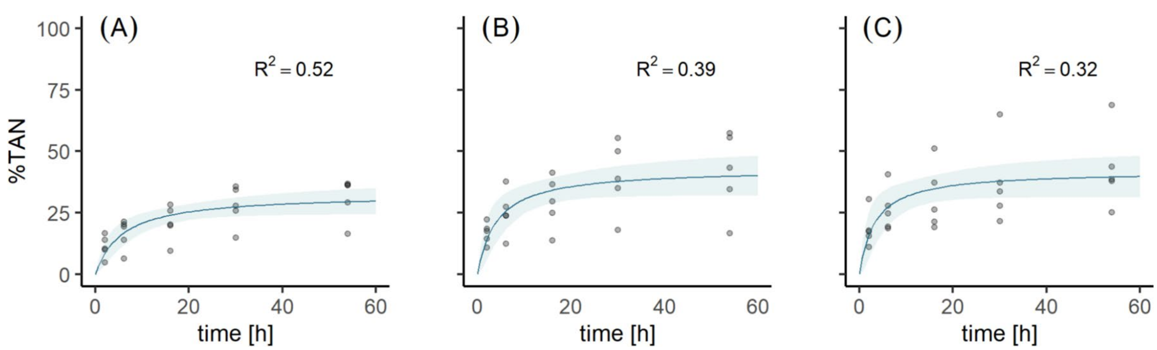

Fig. 2 Time courses of individual application events and predicted average cumulative NH_3 emission with 95% confidence intervals for **A** cattle slurry (SLU), **B** agricultural digestate (SLA), and **C** municipal organic waste digestate (LID) given in percent TAN over 54 h after manure application. Predicted average trajectories were calculated for each treatment based on Michaelis–Menten dynamics

by 40% and 30%, respectively, than the predicted NH_3 loss (Supplementary material, S1).

Discussion

Cumulative NH_3 emissions for SLU were at the higher end of literature values for cattle slurry applied by a trailing hose (Häni et al. 2016) but within the range reported by Hafner et al. (2025). Data obtained from the ALFAM2 database (Hafner et al. 2018) yielded approximately 20% of applied TAN for winter wheat and about 12% for maize based on micrometeorological measurement methods after the application of cattle slurry with a trailing hose. Observed emitted TAN was on average 26% for SLU for winter wheat and 33% for silage maize (Fig. 2).

Both digestates, SLA and LID, exhibited emissions of approximately 40% of applied TAN, which was higher than for SLU (Supplementary material, S1). Although the difference was not statistically significant, higher emissions of digestates would be expected due to the higher values for DM, pH and TAN of SLA and LID compared to SLU (Table 1). Higher emissions for digestates than for untreated slurry were also found in previous studies using surface banding (Ni et al. 2012; Nicholson et al. 2017), broadcasting (Eickenscheidt et al. 2014; Nicholson et al. 2017, 2018) or injection (Nicholson et al. 2018). Schmidhalter (2024) mimicked trailing hose application at laboratory-scale and also found higher emissions from digestates compared to untreated cattle slurry using a ^{15}N balance. Other studies found no significant differences between the emissions from digestates compared to untreated slurries applied with a trailing hose (Andersson et al. 2023; Lemes et al. 2023; Wulf et al. 2002; Wagner et al. 2021) or a trailing shoe (Nicholson et al. 2018). Each of these studies used digestates with DM contents of mostly less than 6%, and the untreated slurries used as a reference were higher in DM than the digestates. This could explain the absence of significant differences in emission levels between untreated slurries and digestates in these studies. The comparison of the absolute emission levels found in the present study with other literature data was challenging due to variations in temperatures ranges, digestate types and properties, and application techniques. All studies cited here except for the lab scale study of Schmidhalter (2024) used enclosures which generally provide only relative estimates of emissions (Hafner et al. 2018).

Pedersen and Hafner (2023) suggested that digestates with a high DM content and with high viscosity due to addition of solid co-substrates exhibit a risk of increased NH_3 emissions. The results of the present study do not clearly support this hypothesis, since the differences between the emissions of digestates and untreated slurry are not statistically significant. This shows that more measurements are necessary of NH_3 emissions after field application of digestates with representative characteristics which currently implies a high DM content. Similarly to Pedersen and Hafner (2023), we support the use of micrometeorological methods which provide absolute emission levels. It must be noted however, that a high emission level of digestates exhibiting a high DM content and thus a substantial share of solid co-substrates does not necessarily indicate higher emissions than if the substrates fed to the AD plants were field-applied untreated. To do this, emission measurements from the application of individual substrates fed to AD plants (e.g. slurry, solid manure, green waste) should be conducted and compared to emissions from the respective digestates. In addition, emissions from all applied fractions after AD must be considered. Our experiments only encompassed untreated slurry and the liquid fraction of the digestates, and thus did not comply with the requirements outlined above.

Conclusion

Due to the increasing use of digestates as fertilizers, this study compared NH_3 emissions from two types of liquid digestates with untreated slurry. Emissions from both digestates tended to be higher than from the untreated slurry but the differences were not statistically significant. Their potentially higher emission level requires additional action to reduce emissions using low emission techniques and management options.

AD is becoming more prevalent but there are still limited data on NH_3 emissions from digestates derived from different feedstock materials and with varying characteristics, especially with high dry matter contents. This highlights the need for additional measurements of these digestate types. Studies must consider the used feedstock materials, the previous stages of the manure cascade and the different fractions, if digestates are further treated (e.g. solid–liquid separation, composting of the solid fraction etc.). A better understanding on how the modified properties of the digestates change their NH_3 emissions compared to untreated manures is necessary for adapting and optimizing manure management.

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Author contributions C.H., N.E. and E.B. planned the experimental work, T.K.and C.H. developed the methodological approach; N.E. carried out the measurements; J.S., H.M.K., C.H. and E.B. guided and supervised the work; N.E. analyzed the data, N.E. and T.K. wrote the publication; all authors commented on the manuscript.

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Data availability No datasets were generated or analysed during the current study.

Declarations

Conflict of interest The authors declare no competing interests.

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