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ORIGINAL ARTICLE

Does woody biochar mixed with liquid organic fertilizer reduce ammonia volatilization following field application?

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Abstract Given biochar's capacity for adsorbing ammonium, its addition to liquid organic fertilizer is a potential strategy to reduce ammonia (NH_3) volatilization. To investigate the effect of biochar produced from wood chips on NH_3 emissions from field-applied liquid organic fertilizers (cattle slurry and anaerobic digestate), NH_3 volatilization was measured at three different sites over two or four years using the dynamic chamber Dräger-Tube Method. The organic compound leonardite was also tested as additive to digestate for NH_3 emission reduction. To offset its alkalinity, biochar was also mixed with the acidic organic compound whey before addition to slurry. Biochar, whey, and leonardite were mixed with liquid organic fertilizers four weeks before application. Two biochar application rates (500 and 1000 kg ha^{-1} yr^{-1} biochar carbon) from two distinct pyrolysis processes were tested. The measurements were conducted in existing multi-year field trials on grassland and arable land reflecting typical biochar farm practices. Liquid organic fertilizers were applied by either soil incorporation, trailing hose, or trailing shoe. In the arable crop rotation, ammonia emissions from cattle slurry without additives over four years (6.7, 35.6, 45.1, and 4.1% of total ammoniacal nitrogen (TAN) applied) were significantly lower than respective emissions from treatments with biochar addition (13.8, 43.5, 48.7, 4.9 and 13.7, 43.9, 51.7, 6.2% of TAN applied). In the grassland, NH_3 volatilization from cattle slurry measured over two years (20.2 and 29.1% of TAN applied) was significantly increased in the second trial year due to biochar admixture (17.6 and 39.8% of TAN applied), irrespective of whey addition. Ammonia volatilization from digestate application to arable land (19.8 and 26.1% of TAN applied) was also significantly lower compared to digestate with biochar addition in the second trial year (19.3 and 29.8% of TAN applied). Leonardite addition to digestate had no effect on NH_3 losses following field application. Ammonia emissions were higher after biochar

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addition, irrespective of the biochar production process, mainly at high application rates. The data suggest that the reduced infiltration of slurry liquid into the soil, resulting from the elevated slurry dry matter content and viscosity, was the primary factor contributing to the observed increase in NH_3 volatilization.

Keywords Biochar · Slurry · Digestate · Leonardite · Ammonia volatilization · Field experiment

Introduction

Gaseous nitrogen (N) volatilization from organic fertilizers in the form of ammonia (NH_3) can result in eutrophication of nutrient-poor ecosystems (Cameron et al. 2013; Fangmeier et al. 1994), while simultaneously reducing fertilizer N use efficiency (Webb et al. 2013). In addition, the contribution of NH_3 to the formation of atmospheric particulate matter can affect human health (Wyer et al. 2022). Germany is committed to reduce its national NH_3 emissions by 29% by 2030 compared to 2005 (European Parliament 2016). In 2020, agriculture accounted for 94% of total NH_3 emissions in Europe (European Environment Agency 2022). The reduction of NH_3 emissions from liquid organic fertilizers such as slurries from livestock production and digestate from biogas plants is a crucial element of mitigation efforts. The application of liquid organic fertilizers accounts for a considerable proportion of total NH_3 emissions (Bussink and Oenema 1998; Sommer and Hutchings 2001). In Germany, this share amounted to 39.5% in 2020 (Vos et al. 2022).

Various methods have been devised to reduce NH_3 emissions during and after slurry application. However, these methods are often cost-intensive for farmers (application with trailing shoes or hoses or shallow soil injection instead of broadcast application (Häni et al. 2016; Seidel et al. 2017)), potentially dangerous to handle (acidification of slurry with sulphuric acid (Kai et al. 2008; ten Huf et al. 2023b)) or difficult to implement in agricultural practice (application at low temperatures and low wind speeds (Sommer et al. 2003)). The addition of slurry additives during storage (e.g. bacteria, wheat straw, sawdust, leonardite, clinoptilolite, and biochar) to mitigate NH_3 emissions is therefore also of interest to farmers and has also been investigated (Cao et al. 2022b; El Bied et al. 2023; McIlroy et al. 2019; Pereira et al. 2022b). Leonardite is an oxidized by-product of lignite, obtained from lignite mining (Broughton 1972; Kalaitzidis et al. 2003). Observations indicate that the volatilization of NH_3 from animal faeces is reduced with leonardite addition, attributable to its acidity and high cation exchange capacity (Chen et al. 2015; Sun et al. 2016).

Biochar is considered as one of the key carbon dioxide removal technologies due to the long-term persistence of carbon in biochar (Gross et al. 2024) and it is predominantly utilized in soil applications where it can offer several co-benefits, i.e., enhance soil fertility (Bhattacharyya et al. 2024; Woolf et al. 2010). Biochar has also been proposed to be used as a slurry additive to reduce odor or as a feed additive to improve animal health and barn hygiene (Joseph et al. 2015, 2021). Laboratory experiments have shown that the addition of biochar can significantly reduce NH_3 loss during slurry storage (Baral et al. 2023; Cao et al. 2022a; Covali et al. 2021). The reduction in NH_3 fluxes can be attributed to the sorption properties of biochar for cations as ammonium (NH_4^+), which can vary considerably depending on the production process, feedstock material, and particle size (Feng et al. 2024; Fidel et al. 2018; Kizito et al. 2016; Viaene et al. 2023a). Other studies have shown no effect of biochar admixture to slurries on NH_3 volatilization (Gronwald et al. 2018; Hung et al. 2022) or even an increase, which has been attributed to an increase in slurry pH resulting from the alkaline nature of many biochars (Kim et al. 2021; Schimmelpfennig et al. 2014; Subedi et al. 2015). Biochar acidification (e.g. using sulphuric acid) prior to the admixture to slurry may reduce NH_3 volatilization through the shift of the NH_4^+: NH_3 equilibrium in the slurry (Miranda et al. 2021). Whey, a by-product of cheese production (Mirabella et al. 2014), with a pH between 3.7 and 4.3 (Braos et al. 2020; Kuhnen et al. 2021), has been shown to reduce NH_3 losses during slurry storage (Jatana et al. 2023).

The impact of biochar on NH_3 fluxes from slurry has been studied during (i) slurry storage (Liu et al. 2021; Pereira et al. 2022a), (ii) after separation of the solid slurry components (Viaene et al. 2023b), or (iii) with the use of biochar as a floating cover (Baral et al. 2023). Ammonia emissions following slurry or digestate application can be notably higher than during storage (Viguria et al. 2015). Most studies on the effects of biochar following slurry and digestate application have often carried out under laboratory conditions (Brennan et al. 2015; Schimmelpfennig et al. 2014; Viaene et al. 2024). However, the results of laboratory and field trials can differ considerably because the environmental influences can be kept constant in the laboratory. Such differences render direct comparisons challenging (Huang et al. 2023; Schimmelpfennig et al. 2014). In order to analyze the effect on NH_3 volatilization when applying biochar with liquid organic fertilizers and to provide clear recommendations for farmers, field trials with economically feasible amounts of high-quality biochar that has been intensively mixed with slurry or digestate early before application are required (Jaufmann et al. 2024).

Ammonia volatilization from liquid organic fertilizers depends on soil texture and soil type, land use (grassland, arable land, permanent crops), the application technique used, the incorporation of slurry into the soil and the meteorological conditions (Misselbrook et al. 2002; Sommer et al. 2003; Sommer and Hutchings 2001; Wulf et al. 2002). Furthermore, biochar properties and thus its effect on NH_3 volatilization can widely differ depending on the pyrolysis process conditions, type of feedstock, and biochar application rate used (Janczak et al. 2017; Liu et al. 2018; Sha et al. 2019; Tomczyk et al. 2020; Wang and Wang 2019).

To investigate the effect of applying practice-relevant rates of biochar, whey, and leonardite to liquid organic fertilizers on NH_3 volatilization under field conditions, a multi-year, multi-site field experiment was conducted in southern Germany. The primary objective of this study was to assess the main question: does applying biochar at practical rates to liquid organic fertilizers reduce NH_3 emissions under real-world conditions? Additionally, this study examines the following hypotheses: (i) The acidification of biochar with whey leads to a significant reduction of NH_3 losses following field application compared to biochar admixture alone; (ii) the addition of leonardite to digestate can reduce NH_3 volatilization following field application.

Materials and methods

Biochar and leonardite

The two distinct types of biochar used in the field experiments were obtained from two commercial producers that use different pyrolysis processes. Pyrolysis biochar (PB) was produced from forest wood chips at a temperature of 700 °C. Pyrolysis + gasification biochar (PGB) was provided by Syncraft GmbH (Syncraft GmbH 2024). This biochar was produced in a process in which forest wood chips were first subjected to pyrolysis at 500 °C in a vertical reactor. Subsequently, the biochar is placed in a floating bed reactor (850 °C), from which the biochar particles can only be discharged when they fall below a specific particle size (Syncraft GmbH 2024). This special process makes the biochar much finer than other biochars, with up to 86.7% of the biochar particles having a smaller diameter than 1 mm (Table 1). Both biochars comply with the guidelines of European Biochar Certification (EBC), which ensures that the biochars are of high quality with a low pollutant content (Table 2).

The leonardite was obtained from the Rösl GmbH company (Regensburg, Germany) (Rösl 2024). Its particle size distribution was comparable with PB. The pH value was slightly acid, and the total organic carbon (TOC) was lower compared to both biochars.

Mixing process of organic fertilizers and additives

Biochar, leonardite and whey were mixed into the slurry or digestate four weeks before spreading. The long mixing period was essential to optimize the adsorption of organic compounds in the slurry or digestate to biochar surface, in order to overcome the inherent hydrophobicity of biochar, and to ensure uniform application. After mixing biochar with the organic fertilizers they were sealed, to exclude NH_3 emissions during the storage process. At all sites and on all dates, the slurry and digestate were sampled approximately one week after the addition of additives in order to calculate the fertilizer N application rates. Mixing was carried out again before sampling. The slurry and digestate nutrient contents and characteristics are presented in Table 3. In comparison to the cattle slurry, the digestate had higher contents of

Table 1 Biochar and leonardite characteristics

Parameter	Unit	Pyrolysis biochar (PB)			Pyrolysis + gasifier biochar (PGB)		Leonardite (L)
Year		2021	2022, 2023	2024	2023	2024	
Total organic carbon (TOC)	% DM	84.6	85.0	90.0	85.0	81.4	62.7
Specific surface area (SSA)	$m^2 g^{-1}$	300	377	–	276[a]	223	–
Highest temperature (HT)	°C	700	700	700	820	820	–
Dry matter (DM)	%	74.9	76.1	64.6	68.1	76.7	59.0
pH in $CaCl_2$		7.9	9.2	8.3	9.5	10.0	6.2
Total nitrogen (TN)	% DM	0.36	0.46	0.38	0.47	0.53	0.53
H/C_{org}		0.41	0.21	0.32	0.12	0.10	–
O/C_{org}		–	–	0.035	–	0.032	–
Particle size distribution							
< 1 mm	% (w/w)	45.6	45.6	24.7	68.0	86.7	59.5
1–2 mm	% (w/w)	12.4	23.0	20.7	21.1	6.6	18.6
2–4 mm	% (w/w)	19.3	15.9	23.2	6.4	3.7	17.1
4–6 mm	% (w/w)	13.1	10.7	19.4	3.3	1.9	3.6
> 6 mm	% (w/w)	9.6	4.8	12.0	1.2	1.2	1.2

[a]Value from an older batch; Analysis methods: Carbon, Hydrogen, and Nitrogen (DIN 51732); Oxygen (DIN 51733); Carbonate (DIN 51726); SSA (DIN ISO 9277 and DIN 66137);

pH (DIN ISO 10390); DM (DIN 51718);

H/C_{org} and O/C_{org} are indicators for the degree of carbonization and biochar quality (Schimmelpfennig and Glaser 2012)

dry matter (DM), total nitrogen (TN) and total ammoniacal nitrogen (TAN), along with a higher pH.

Study sites and experimental design

The measurements of NH_3 emissions were carried out in multi-annual field trials at three locations in southern Germany with different soil and climatic conditions (Table S1). The aim of these trials was to investigate the effects of biochar in combination with organic and mineral fertilizers on crop yield, soil properties, and environmental impacts in conventional and organic farming. However, the present study exclusively examined NH_3 volatilization.

Thalhausen field experiment

A plot trial with conventional arable farming was set up at the Thalhausen/Dürnast research station of the Technical University of Munich (48°26′14.1″ N, 11°39′ 43.2″ E), located 30 km North of Munich. The crops grown in the field experiment were silage maize (2021, 2024), winter wheat (2022), and winter barley (2023). Cattle slurry (S) was applied at a N rate of 100 kg ha^{-1} yr^{-1}. The biochar carbon (C) application rates were 500 kg ha^{-1} yr^{-1} (PB1) und 1000 kg ha^{-1} yr^{-1} (PB2). The field trial consisted of a total of 18 treatments (Fig. S1). In the treatments, cattle slurry without biochar (S), cattle slurry with 500 kg ha^{-1} yr^{-1} biochar C (S PB1), and cattle slurry with 1000 kg ha^{-1} yr^{-1} biochar C (S PB2) NH_3 losses were measured immediately after slurry application (Table 2). The cattle slurry was mixed with the biochar in slurry tankers with a capacity of 6–8 m^3. Mixing was carried out by recirculation inside the slurry tank. The mixing ratio (MR) ranged between 12.2 kg m^{-3} (S PB1 in 2024) and 31.7 kg m^{-3} (S PB2 in 2022). The MR was chosen depending on the slurry N and the biochar C content. Each year, the cattle slurry was applied using a plot slurry tanker with a trailing hose or trailing shoe. The cattle slurry was incorporated immediately after application with a cultivator (0.12 m) in April 2021 and a rotary harrow (0.08 m) in April 2024 prior to maize sowing and band applied to the soil surface in winter wheat in March 2022 and in winter barley in April 2023.

Burghausen field experiment

At the Burghausen site (48°08′28.2 "N, 12°46′27.0 "E), located 90 km east of Munich, the plot trial was conducted on a field of a local conventional farm, where winter wheat was grown in 2023, and winter barley in 2024. Biochar C was applied at rates

Table 2 Overview of all trials and the weather conditions during the measurement period

Year	2021	2022		2023			2024	
Study site	Thalhausen	Thalhausen	Baierlach	Thalhausen	Burghausen	Baierlach	Burghausen	Thalhausen
Crop	Silage maize	Winter wheat	Grassland	Winter barley	Winter wheat	Grassland	Winter barley	Silage maize
Application date	April, 15	March, 24	July, 12[b]	April, 05	March, 23	July, 11[b]	March, 27	April, 30
Application technology	Trailing hose	Trailing hose	Trailing hose	Trailing hose	Trailing hose	Trailing shoe	Trailing shoe	Trailing shoe
Soil incorporation	Cultivator	none	none	none	none	none	none	Rotary harrow
Organic fertilizer	Cattle slurry	Cattle slurry	Cattle slurry	Cattle slurry	Digestate	Cattle slurry	Digestate	Cattle slurry
Total N input [kg ha^{-1}]	100	100	34.5[c]	100	170	57[c]	170	100
Treatments analyzed	S	S	S	S	D	S	D	S
	S PB1	S PB1	S PB2	S PB1	D PGB1	S PB2	D PGB1	S PB1
	S PB2	S PB2	S PB2 W	S PB2	D L	S PB2 W	D L	S PB2
Organic fertilizer additive	Biochar	Biochar	Biochar, whey	Biochar	Biochar, leonardite	Biochar, whey	Biochar, leonardite	Biochar
Weather conditions[d]								
Total precipitation (mm)	8.8	2.8	3.8	0.5	10.6	20.5	8.6	0.0
Mean temperature (°C)	4.3	9.4	19.9	4.7	7.4	21.2	11.2	15.5
Mean wind speed ($m\ s^{-1}$)	1.2	2.0	0.2	2.3	3.4	0.2	2.2	1.4

[b]Application to the third growth of the meadow; [c]Slight differences between the treatments (see 2.3); [d]Details of the weather during the measurement period can be found in the supplementary material; S = Cattle slurry without any supplement; D = Biogas digestate without any supplement; PB1 = Pyrolysis biochar C (500 kg ha^{-1} yr^{-1}); PB2 = Pyrolysis biochar C (1000 kg ha^{-1} yr^{-1}); PGB1 = Pyrolysis + gasifier biochar C (500 kg ha^{-1} yr^{-1}); L = Leonardite C (500 kg ha^{-1} yr^{-1}); W = Whey (2 l kg^{-1} biochar)

Table 3 Properties of the mixtures from organic fertilizers with biochar, leonardite, and whey in Thalhausen, Burghausen, and Baierlach

Study site	Parameter	Unit	2021			2022			2023			2024		
Treatment			S[e]	S PB1	S PB2	S[e]	S PB1	S PB2	S[e]	S PB1	S PB2	S[e]	S PB1	S PB2
Thalhausen	DM	%	6.3	7.6	9.2	6.0	7.7	8.1	4.5	5.9	6.7	3.9	5.4	6.6
	TN	kg m^{-3}	2.6	3.0	3.1	2.5	2.5	2.6	2.3	2.4	2.5	1.9	2.0	2.0
	TAN	kg m^{-3}	1.4	1.4	1.4	1.3	1.2	1.2	1.1	1.1	1.1	1.0	1.0	0.9
	MR	kg m^{-3}		13.0	25.3		18.1	31.7		14.9	29.5		12.2	26.2
	pH											7.3	7.3	7.4
Treatment									D	D PGB1	D L	D	D PGB1	D L
Burghausen	DM	%							7.5	8.6	8.8	7.7	9.0	9.1
	TN	kg m^{-3}							4.9	4.8	4.8	5.2	5.1	5.0
	TAN	kg m^{-3}							2.8	2.7	2.6	3.1	3.0	2.9
	MR	kg m^{-3}								15.9	21.5		15.9	21.1
	pH											8.2	8.3	8.0
Treatment						S[f]	S PB2	S PB2 W	S[g]	S PB2	S PB2 W			
Baierlach	DM	%				6.5	9.1	9.0	6.6	8.8	8.4			
	TN	kg m^{-3}				2.7	2.8	2.8	3.2	3.3	3.3			
	TAN	kg m^{-3}				1.1	1.1	1.1	1.6	1.4	1.4			
	MR	kg m^{-3}					30.2	28.2		21.5	20.2			
	pH								6.9	6.9	6.8			

DM = Dry matter; TN = Total nitrogen; TAN = Total ammoniacal nitrogen; MR = mixing ratio between biochar DM weight and liquid organic fertilizer volume + (whey); S = Cattle slurry N ((100)[e], (34.5)[f], (57.0)[g] kg ha^{-1} yr^{-1})); PB1 = Pyrolysis biochar C (500 kg ha^{-1} yr^{-1});PB2 = Pyrolysis biochar C (1000 kg ha^{-1} yr^{-1}); D = Digestate N (170 kg ha^{-1} yr^{-1}); PGB1 = Pyrolysis + gasifier biochar C (500 kg ha^{-1} yr^{-1}); L = Leonardite C (500 kg ha^{-1} yr^{-1}); W = Whey (2 l kg^{-1} biochar); Analysis methods: DM (DIN 15934); TN (DIN 16168); TAN (DIN 38406–5-2); pH (DIN 15933) of 500 kg ha^{-1} yr^{-1} (PGB1) and 1000 kg ha^{-1} yr^{-1} (PGB2) in combination with anaerobic digestate. The digestate originated from a thermophilic biogas plant in which silage maize and cattle manure were used as feedstock for biogas production with a retention time of 194 days. For comparison with biochar, leonardite was used in one treatment at a C application rate of 500 kg ha^{-1} yr^{-1}. All digestate treatments were fertilized at a N rate of 170 kg ha^{-1} yr^{-1}. The field trial consisted of a total of 10 treatments (Figure S2). The NH_3 volatilization measurements were carried out in the digestate without biochar (D), digestate with 500 kg ha^{-1} yr^{-1} biochar C (D PGB1) and digestate with 500 kg ha^{-1} yr^{-1} leonardite C (D L) treatments (Table 2). The digestate was stored in 1000-L tanks. The biochar and leonardite were mixed into the digestate using an electrically operated hand mixer. The MR of leonardite (21.5 and 21.1 kg m^{-3}) was higher compared to biochar (15.9 kg m^{-3}) because of the lower C content of leonardite. In March 2023, the application of digestate was conducted to winter wheat using a trailing hose applicator. The digestate was applied to winter barley in March 2024 at the same application rate with a trailing shoe applicator.

Baierlach field experiment

A plot trial was established at the Baierlach site (47°50′59.2″ N, 11°24′34.1″ E), located 35 km south of Munich, on an organically managed permanent grassland with 8 treatments (Figure S3). The trial involved the application of cattle slurry with biochar C rates of 500 kg ha^{-1} yr^{-1} (PB1) and 1000 kg ha^{-1} yr^{-1} (PB2). Furthermore, whey was used in one treatment to lower the alkaline biochar pH. NH_3 loss was measured in the cattle slurry (S), cattle slurry with 1000 kg ha^{-1} yr^{-1} biochar C (S PB2), and cattle slurry with 1000 kg ha^{-1} yr^{-1} biochar C and whey (S PB2 W) treatments (Table 2). Slurry storage and mixing with biochar (MR 30.2 kg m^{-3} (2022) and 21.5 kg m^{-3} (2023)) were carried out in the same way as in Burghausen. To buffer the alkaline pyrolysis biochar pH, whey (pH = 4.8) was added to the biochar before it was mixed into the cattle slurry. Approximately 2 L of whey was added per kilogram of biochar, whereby the MR was reduced. In the grassland trial, the slurry was applied three times per year and the NH_3 volatilization measurements were conducted in both years at the third application, which occurred in July. In July 2022 a trailing hose applicator was used to apply 34.5 kg ha^{-1} (S), 35.9 kg ha^{-1} (S PB2) and 35.9 kg ha^{-1} (S PB2 W) of cattle slurry N. In July 2023, 57.0 kg ha^{-1} (S), 57.0 kg ha^{-1} (S PB2) and 59.0 kg ha^{-1} (S PB2 W) of cattle slurry N were applied with the trailing shoe applicator. It was observed during the application process that the flowability of the slurry was reduced by the biochar admixture.

NH_3 volatilization measurements and flux calculations

The measurement of the NH_3 volatilization was carried out with the dynamic chamber Dräger-Tube Method (DTM) described by Pacholski et al. (2006). Two of the four chambers (115 mm diameter) were placed on slurry bands, while the remaining two were positioned between the slurry bands. The chamber headspace air was sucked at a defined rate through adapted gas detector tubes, up to a defined sampling range for NH_3 concentrations.

The measurements were taken at two-hourly intervals between 9:00 and 17:00 throughout the day (five measurements per day). They were carried out in three fertilized treatments and in the unfertilized control, each with three repetitions at every site. Separate chambers were used for the unfertilized treatments to avoid contamination. In order to prevent air exchange between the sward and the chambers, soil rings were pressed into the ground in the permanent grassland (Baierlach) for the duration of the measurement period. In arable soils (Thalhausen, Burghausen) the chambers were cautiously pressed into the soil.

The measurements produce flux readings which require an empirical calibration to reflect NH_3 emissions under in situ conditions (Pacholski et al. 2006). In order to convert the NH_3 fluxes in μmol mol^{-1} (ppm) to kg ha^{-1}, the detected NH_3 volatilization quantities were calculated on the basis of the current time, air temperature, air pressure, wind speed and the number and duration of pump strokes. The NH_3 concentrations of the unfertilized control were assumed to reflect background concentration and subtracted from those of the fertilized treatments. The NH_3 losses were cumulated by trapezoidal rule individually for each plot in order to facilitate statistical analysis of the data and accounting for error propagation in

Table 4 Absolute and TAN-input related NH_3 volatilization (mean ± standard deviation) following cattle slurry application in 2021, 2022, 2023 and 2024 (Thalhausen); means with at least one identical letter were non-significant from each other

Year	Crop	Treatment	Slurry N input (kg ha^{-1})	Biochar C input (kg ha^{-1})	TAN applied (kg ha^{-1})	Absolute NH_3-N emissions (kg ha^{-1})	TAN-input related NH_3-N emissions (% of TAN applied)
2021	Silage maize[h]	S	100	0	53.8	3.6 ± 1.3 a	6.7 ± 2.3 a
		S PB1	100	500	46.7	6.5 ± 1.0 b	13.8 ± 2.1 b
		S PB2	100	1000	45.2	6.2 ± 0.1 b	13.7 ± 0.1 b
2022	Winter wheat	S	100	0	52.0	18.5 ± 0.2 a	35.6 ± 0.4 a
		S PB1	100	500	48.0	20.9 ± 1.8 a	43.5 ± 3.7 b
		S PB2	100	1000	46.2	20.3 ± 1.0 a	43.9 ± 2.2 b
2023	Winter barley	S	100	0	47.8	21.6 ± 1.0 a	45.1 ± 2.0 a
		S PB1	100	500	45.8	22.3 ± 0.2 a	48.7 ± 0.4 ab
		S PB2	100	1000	44.0	22.7 ± 1.5 a	51.7 ± 3.3 b
2024	Silage maize[h]	S	100	0	52.4	2.1 ± 0.6 a	4.1 ± 1.1 a
		S PB1	100	500	48.0	2.4 ± 0.3 a	4.9 ± 0.7 a
		S PB2	100	1000	46.5	2.9 ± 1.0 a	6.2 ± 2.2 a

[h]Incorporation; S = Cattle slurry N (100 kg ha^{-1} yr^{-1}); PB1 = Pyrolysis biochar C (500 kg ha^{-1} yr^{-1}); PB2 = Pyrolysis biochar C (1000 kg ha^{-1} yr^{-1}); TAN = Total ammoniacal nitrogen

each plot. Error bars in the graphs indicate standard deviation of cumulated emissions calculated for each measurement date.

In addition to the absolute cumulative NH_3-N emissions, the relative NH_3-N emissions of the fertilizer TAN input were calculated (Eq. 1).

$$TAN - input\ related\ NH_3 - N\ emission\ [\%\ of\ TAN\ applied] = \frac{absolute\ cumulative\ NH_3 - N\ emissions\ [kg\ ha^{-1}]}{TAN\ applied\ [kg\ ha^{-1}]} \quad (1)$$

For application of the calibration formula of NH_3 flux correction, data from the nearby Dürnast weather station (distance: 4 km) were used in 2021, 2022 and 2023 at the Thalhausen site. At the Baierlach, Burghausen, and Thalhausen (in 2024) sites, a mobile weather station (Onset 2024) was used. The mobile weather station was installed for the duration of the measurement period, providing data on temperature and wind speed 2 m above the ground. The air pressure data were obtained from nearby stations of the German Weather Service (DWD Climate Data Center 2024).

In order to validate the results from the DTM, we used the ALFM2 spreadsheet model (https://projects.au.dk/alfam) which is based on micrometeorological measurements and synthesizes empirical knowledge (Hafner 2024). Input data from the field trials (slurry type, dosage rate, DM etc.) were used to run the model for a time period identical with measurement duration. The discrepancy between DTM and ALFAM2 was analyzed in the context of the mean absolute error from the cross-validation method outlined in Hafner et al. (2025).

Statistical analysis

Statistics were carried out using the software R (R Core Team 2024). For the evaluation with linear mixed models, the packages "lme4" (Bates et al. 2024) and "lmerTest" (Kuznetsova et al. 2020) were used. Initially, the statistical analysis was carried out separately for each site and each year. The absolute and relative losses obtained in individual measurement campaigns were not directly comparable with each other because different additives, organic reference fertilizers, crops and application methods were examined, respectively. The treatment was included as a fixed effect in the model and the block as random effect. Equation (2) shows an example of the model used.

$$Response_{ij} = \beta_0 + \beta_1 treatment_i + u_i + \varepsilon_{ij} \quad (2)$$

where $Response_{ij}$ represents the lth observation in row i. The analysis of variance (ANOVA) and a post hoc (HSD) test were performed to determine which treatments differed significantly. Significant

Table 5 Absolute and TAN-input related NH_3 volatilization (mean ± standard deviation) following digestate application in 2023 and 2024 (Burghausen); means with at least one identical letter were non-significant from each other

Year	Crop	Treatment	Digestate N input (kg ha^{-1})	Biochar C input (kg ha^{-1})	TAN applied (kg ha^{-1})	Absolute NH_3-N emissions (kg ha^{-1})	TAN-input related NH_3-N emissions (% of TAN applied)
2023	Winter wheat	D	170	0	97.1	19.2 ± 0.6a	19.8 ± 0.6a
		D PGB1	170	500	95.6	18.4 ± 1.3a	19.3 ± 1.4a
		D L	170	500	92.1	18.5 ± 0.7a	20.1 ± 0.7a
2024	Winter barley	D	170	0	101.4	26.4 ± 1.2a	26.1 ± 1.2a
		D PGB1	170	500	100.2	29.9 ± 1.2b	29.8 ± 1.2b
		D L	170	500	98.6	25.8 ± 0.7a	26.1 ± 0.7a

D = Digestate N (170 kg ha^{-1} yr^{-1}); PGB1 = Pyrolysis + gasifier biochar C (500 kg ha^{-1} yr^{-1}); L = Leonardite C (500 kg ha^{-1} yr^{-1}); TAN = Total ammoniacal nitrogen

Table 6 Absolute and TAN-input related NH_3 volatilization (mean ± standard deviation) following cattle slurry application in 2022 and 2023 (Baierlach); means with at least one identical letter were non-significant from each other

Year	Treatment	Slurry N input (kg ha^{-1})	Biochar C input (kg ha^{-1})	TAN applied (kg ha^{-1})	Absolute NH_3-N emissions (kg ha^{-1})	TAN-input related NH_3-N emissions (% of TAN applied)
2022	S	34.5	0	14.3	2.9 ± 0.5 a	20.2 ± 3.3 a
	S PB2	35.9	1000	14.3	2.5 ± 0.2 a	17.6 ± 1.4 a
	S PB2 W	35.9	1000	14.3	2.6 ± 0.4 a	18.3 ± 2.8 a
2023	S	57.0	0	28.2	8.2 ± 0.4 a	29.1 ± 1.4 a
	S PB2	57.0	1000	24.5	9.8 ± 0.4 a	39.8 ± 1.6 b
	S PB2 W	59.0	1000	25.3	9.6 ± 1.2 a	37.8 ± 4.6 b

S = Cattle slurry, PB2 = Pyrolysis biochar C (1000 kg ha^{-1} yr^{-1}), W = Whey (2 l kg^{-1} biochar); TAN = Total ammoniacal nitrogen

differences are marked with different letters. The "emmeans" package (Lenth et al. 2024) and the "multcomp" package (Hothorn et al. 2023) were used for this purpose.

To provide a mechanistic interpretation of the results, an analysis of NH_3 losses across locations and years was necessary. For this purpose, the NH_3 emission factor (EF) (dimensionless) was calculated using the following formula:

$$EF = \frac{TAN - input\ related\ NH_3 - N\ emission\ [\%\ of\ TAN\ applied]}{TAN - input\ related\ NH_3 - N\ emission\ [\%\ of\ TAN\ applied]\ from\ the\ reference\ treatment} \tag{3}$$

In this case, the references were the NH_3-N loss from the treatment (slurry or digestate) without admixture of biochar, leonardite, or whey. This factor was calculated with mean values (Tables 4, 5, and 6) to avoid error summation. The organic fertilizer DM content factor (DMF) (dimensionless) was calculated identically.

Furthermore, a correlation analysis was carried out to investigate the effects of biochar quantity, MR, and DMF on the EF. Here due to a low level of replication, the treatments with leonardite and whey were excluded. The significance of Pearson correlation was indicated as follows: $*p < 0.05$, $**p < 0.01$, $***p < 0.001$, n.s. not significant. Subsequently, another linear mixed effects model was fitted to evaluate the effects of slurry properties (DMF and TAN), incorporation, crop and weather during the measurement period (temperature, wind speed, and precipitation) on the EF. The Akaike Information Criterion (AIC) was used for model selection.

The figures were created in R Studio using the package "ggplot2" (Wickham et al. 2024).

Results

Thalhausen, conventional arable farming, cattle slurry application

A four-year series of measurements was conducted at the Thalhausen site. In two trial years, a reduction in slurry TAN content was observed with biochar admixture (Table 3). This reduction was characterized by a lower TAN input despite the same TN fertilization. In the two trial years in which soil incorporation was conducted immediately following slurry application (2021, 2024), the TAN-input related NH_3-N emissions were observed to be low, ranging from 4.1 to 13.8%. In the absence of soil incorporation in the trial years 2022 and 2023, the TAN-input related NH_3-N emissions were notably higher, reaching 35.6 to 51.7% (Table 4). In 2021, the majority of the NH_3-N emissions observed occurred during the initial period after the application in the slurry treatment (Fig. 1a). In contrast, the biochar-amended treatments also exhibited NH_3-N emissions over the subsequent days, whereby the TAN-input related NH_3-N emissions were significantly higher compared to the S treatment (Table 4). In winter wheat (2022), the TAN-input related NH_3-N emissions were significantly increased with biochar compared to the cattle slurry treatment. In winter barley, the TAN-input related NH_3-N emissions were significantly higher in treatment S PB2 (51.7%) compared to the slurry treatment (45.1%). As illustrated in Fig. 1b, c, the highest NH_3-N losses occurred immediately following slurry application, although measurable fluxes were observed

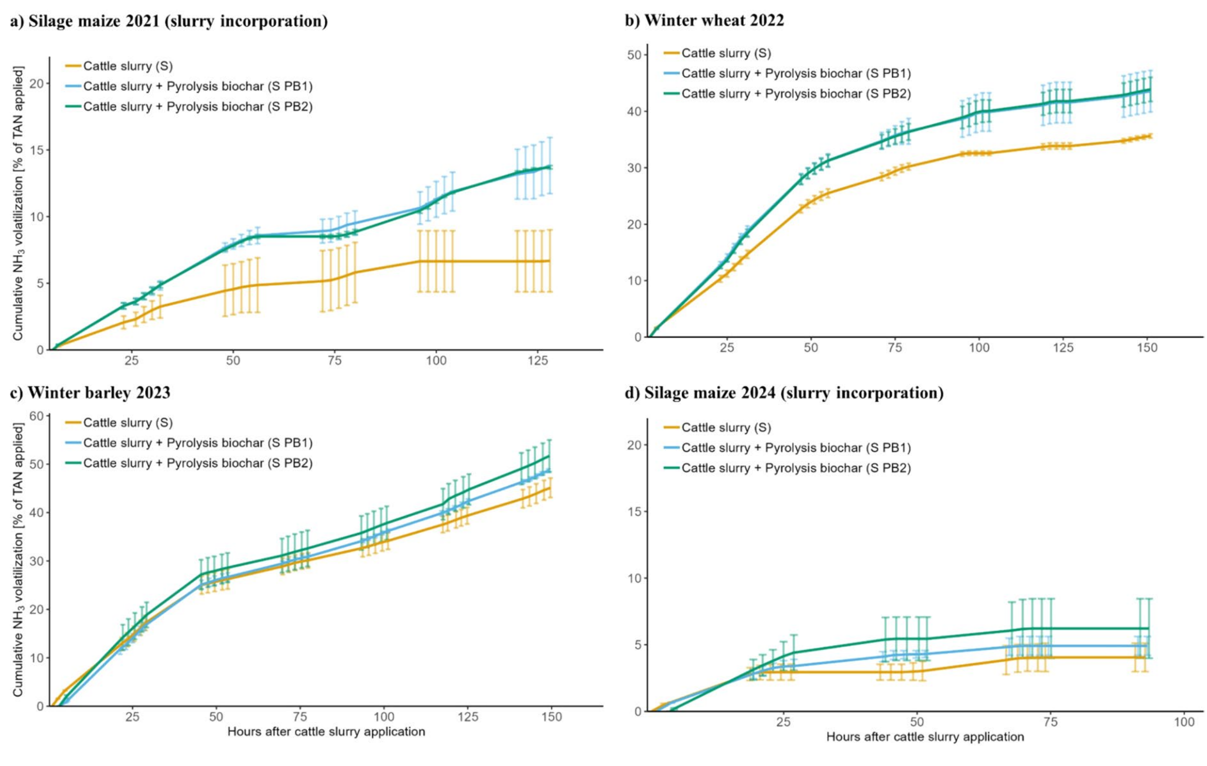

Fig. 1 Cumulative TAN-input related NH_3 volatilization in Thalhausen (2021, 2022, 2023, 2024); S = Cattle slurry N (100 kg ha^{-1} yr^{-1}); PB1 = Pyrolysis biochar C (500 kg ha^{-1} yr^{-1}); PB2 = Pyrolysis biochar C (1000 kg ha^{-1} yr^{-1}); soil incorporation 2021 with cultivator (0.12 m) and 2024 with rotary harrow (0.08 m); the error bars indicate the standard deviation calculated for each measurement date; TAN = Total ammoniacal nitrogen

for all treatments throughout the entire measurement period. In 2024, TAN-input related NH_3-N emissions did not differ significantly (Table 4). However, NH_3-N losses tended to be higher due to the addition of biochar to the cattle slurry. As illustrated in Fig. 1d, the losses were only minimal after the second day following the application of slurry.

The comparison with ALFAM2 indicated that the difference of TAN-input related NH_3-N emissions (DTM-ALFAM2) was low (−6.4 to 2.9%) when cattle slurry was incorporated (Table S20). However, when slurry was applied to the cereal canopy, the TAN-input related NH_3-N emissions measured with DTM were 14.7–30.8% higher compared to the values estimated by ALFAM2. However, the differences observed between the treatments with DTM were also observed with ALFAM2.

Burghausen, conventional arable farming, biogas digestate application

The addition of pyrolysis + gasifier biochar and leonardite also led to a decrease in the TAN content in the digestate, as can be seen from the quantities of TAN applied (Table 5). In 2023, the NH_3-N losses in the treatment D PGB1 were observed to be temporarily lower than in the other treatments (Figure S4a). However, the reduction was only statistically significant two days after the digestate application (Table S6). Over the subsequent days, the plots with biochar amendment exhibited a greater loss of NH_3-N, resulting in non-significant differences by the conclusion of the measurement period (Table 5). In 2024, the NH_3-N losses due to the addition of pyrolysis + gasifier biochar were significantly higher from the second day compared to the D L treatment. From the fourth day, the cumulative NH_3-N volatilization in the treatment D PGB1 was significantly higher than in the D treatment (Figure S4b, Table S7). After one week, the TAN-input related NH_3-N emissions with the addition of pyrolysis + gasifier biochar (29.8%) were significantly higher than in the D (26.1%) and D L (26.1%) treatments.

The comparison of DTM measurement results and ALFAM2 model output indicates that the TAN-input related NH_3-N emissions following digestate application were underestimated (Table S21). The mean difference of all treatments in 2023 and 2024 of TAN-input related NH_3-N emissions (DTM-ALFAM2) was −14.2%, with a significantly higher deviation in 2023 compared to 2024, where the differences between the treatments were also modelled very accurately.

Baierlach, organic permanent grassland, cattle slurry application

In the first trial year, no significant differences were observed between the treatments (Table 6). NH_3-N losses were only measurable on the day of application and the following day. There was no further volatilization on the subsequent days (Figure S5a). In the second trial year, a greater slurry TAN quantity was applied, resulting in elevated NH_3-N losses that persisted over several days (Figure S5b). The TAN-input related NH_3-N emissions were significantly higher in the S PB2 and S PB2 W treatments than in the S treatment (Table 6).

The comparison of DTM measurement and ALFAM2 for the grassland trial indicated that the TAN-input related NH_3-N emissions following cattle slurry application showed lower values for the model output (Table S22). The mean difference of all treatments in 2022 and 2023 of TAN-input related NH_3-N emissions (DTM-ALFAM2) was 20.4%. The much greater deviation in 2023 than in 2022 is noticeable. In 2022, the model output indicated higher TAN-input related NH_3-N emissions with biochar addition, which was not measured with DTM. In 2023, the ALFAM2 model output predicted an increase in TAN-input related NH_3-N emissions by 0.5% with biochar addition, while the effect measured with DTM was 10.7%.

Cross-year and cross-location analysis

The TAN-input related NH_3-N emissions for all locations were evaluated depending on the weather conditions (Fig. 2). The figure indicates that the incorporation of slurry was accompanied with a low level of NH_3 emissions (< 16%). However, the study does not show a clear effect regarding the impact of weather variables such as temperature, wind, and precipitation on NH_3 volatilization.

To gain a more profound understanding of the impact of biochar on NH_3 losses, the EF was calculated and statistically evaluated across all sites and years. The EF varied between 0.87 and 2.06; however, at most examination dates, the values were

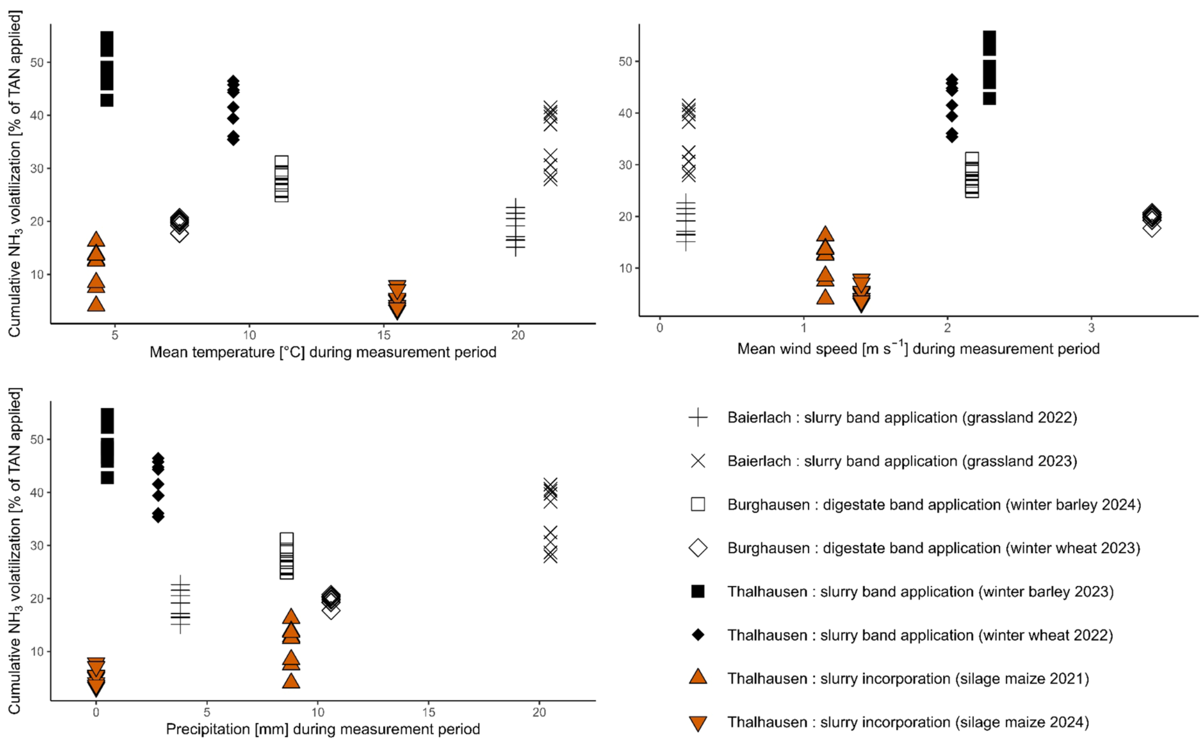

Fig. 2 Effect of weather (temperature, wind, and precipitation) during the measurement period on the TAN-input related NH_3-N emissions; each symbol shows a separate measuring period

Table 7 NH_3 emission factor (EF) (dimensionless) for all trial year and all sites

Year	Thalhausen		Burghausen		Baierlach	
	S PB1	S PB2	D PGB1	D L	S PB2	S PB2 W
2021	2.06	2.04				
2022	1.22	1.23			0.87	0.91
2023	1.08	1.15	0.97	1.02	1.37	1.30
2024	1.20	1.51	1.14	1.00		

EF was calculated according to Eq. 3; S = Cattle slurry; PB1 = Pyrolysis biochar C (500 kg ha^{-1} yr^{-1}); PB2 = Pyrolysis biochar C (1000 kg ha^{-1} yr^{-1}); W = Whey (2 l kg^{-1} biochar); D = Digestate; PGB1 = Pyrolysis + gasifier biochar C (500 kg ha^{-1} yr^{-1}); L = Leonardite C (500 kg ha^{-1} yr^{-1})

Table 8 Summary of the mixed effects model (Eq. 4)

	Estimate	Standard error	t value	*p* value	Significance
Intercept	1.061	0.438	2.422	0.028	*
DMF	0.633	0.272	2.329	0.034	*
Temperature (°C)	−0.033	0.013	−2.507	0.024	*
Wind speed (m s^{-1})	−0.163	0.093	−1.749	0.133	n.s.

The effects of relative organic fertilizer dry matter (DMF), mean temperature, and mean wind speed during the measurement period on the NH_3 emission factor (EF); Significance of factors were indicated as follows: *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$, n.s. not significant greater than 1.00. This indicated that the NH_3 volatilization was higher because of biochar, leonardite, or whey amendment. At the Thalhausen site, the EF was consistently greater than 1.00. In contrast, at the Burghausen and Baierlach sites, biochar addition increased emissions only in one of the two investigated years (Table 7).

The EF correlated weakly, but significantly with the biochar quantity (r = 0.48) and the DMF (r = 0.49) (Table S18). The DMF strongly correlated with biochar quantity and MR, respectively. Therefore, only the DMF was considered in the further statistical analysis. The DMF, temperature, and wind were integrated as fixed effects. The crop was set as the only random effect, which contains additional information about organic fertilizer incorporation. The DMF exhibited a substantial explanatory effect on EF, which was also statistically significant (Table 8). Air temperature and measurement campaign average wind speed (2 m height) had small negative parameter estimates; however, the effect of temperature was significant.

Discussion

NH_3 measurement methodology

The DTM has been widely used in field experiments in Germany and China with small plots to measure NH_3 fluxes (Gericke et al. 2011; Ni et al. 2015; Zhang et al. 2022). Nevertheless, for all results, it is important to note that the positioning of the chambers can influence the quality of the data, as the slurry bands have different shapes, and the chambers cover only a small area. In some cases, the DTM yielded lower NH_3 emissions than micrometeorological methods (Kamp et al. 2024; Vilms Pedersen et al. 2018), while it also showed good agreement in others (Gericke et al. 2011; Quakernack et al. 2012). This may be due to the fact that the method was only calibrated up to a wind speed of 4 m s^{-1}, so that the NH_3 losses are underestimated at higher wind speeds (ten Huf et al. 2023a; Vilms Pedersen et al. 2018). During the measurement periods investigated in this study, the wind speed was, with few exceptions, below 4 m s^{-1} (see supplementary material).

A large database was compiled to adapt the ALFAM2 model (Hafner et al. 2025). This database contains NH_3 measurements obtained by micrometeorological methods from various countries under different environmental conditions. Following the model fitting, a cross-validation was performed, where the apparent error of TAN-input related NH_3-N emissions varied mainly between 5 and 20%, with most values below 30%. The mean difference (DTM-ALFAM2) of each study site was within this range (Table S20-S22), with a tendency of higher fluxes compared to ALFAM2 calculations. However, the model mostly simulated the differences between treatments in the same way as measured by the DTM, mainly due to differences in DM content. Kelsch et al. (2025) discovered that the DTM underestimated the NH_3 concentration of the sampled air, with increasing uncertainty when the emissions were low. Consequently, they suggested a higher detection limit. In this field trial, the NH_3 volatilization was mostly much higher in the first 2 days after application, but from day 3 to 7, the detected NH_3 concentrations were often lower than the detection limit observed by Kelsch et al. (2025). The tendency of DTM measurements to result in higher cumulated losses than ALFAM2 calculations can also be explained by the timing of measurements. In the original method (Pacholski et al. 2006), it is strongly recommended to do measurements in the very early morning (shortly after daybreak) and later in the evening, as these are the times with low emissions. In the interpolation between fluxes obtained by the measurement protocol in this study (measurements between 9 am and 5 pm), these timings and low fluxes were not appropriately considered and resulted thus in higher cumulated emissions. Nevertheless, based on this discussion, measured NH_3 emissions in this study can be considered in the range of emissions observed by micrometeorological methods as reflected in the ALFAM2 model.

Effects of application technique, field crops, and weather conditions on NH_3 volatilization following field application

The application technique has a significant impact on NH_3 volatilization following liquid organic fertilizer application (Häni et al. 2016; Webb et al. 2010). Previous studies observed emission reductions by slurry incorporation of 40–81% (Emmerling et al. 2020; Hafner et al. 2019; Sanz-Cobena et al. 2019). Soil incorporation or injection minimizes the contact of the slurry with the atmosphere and the positively charged NH_4^+ ions are fixed to the exchangers of organic and mineral soil components, thereby preventing NH_3 volatilization (Pietzner et al. 2017; Vilarrasa-Nogué et al. 2020; Wang et al. 2022). Nevertheless, the slurry incorporation technique and depth can affect NH_3 emissions (Carozzi et al. 2013; Webb et al. 2010). In the year 2021, the slurry was incorporated with a cultivator (0.12 m), compared to the incorporation with a rotary harrow (0.08 m) in 2024. In contrast to the cultivator, which works in strips, the rotary harrow mixes the entire soil surface. Although measurements were taken in different years under varying conditions, and thus are not directly comparable, incorporation with a cultivator was associated with higher NH_3 emissions than incorporation with a rotary harrow.

However, losses of NH_3 from incorporated slurry may be challenging to detect with DTM due to the unknown distribution of the slurry in the soil (Ni et al. 2015).

The emission behavior in grassland differs significantly from that of arable land, as no incorporation into the soil is possible (except in the case of open slot injection). Additionally, there is no direct contact between the slurry strips and the soil due to the sward. Moreover, there is an increased surface between slurry and air, because slurry adheres to the grassland leaves (Thompson et al. 1990). In the first year of the trial, overall NH_3-N losses were low at the Baierlach site. High temperatures and drought resulted in rapid drying of the slurry bands on the sward (Table S16). The desiccation of the slurry strip induced the formation of a crust that prevented gas exchange (Braschkat et al. 1997; Salazar et al. 2014).

Biochar admixture to organic fertilizers increased NH_3 volatilization

The goal of this study was to evaluate whether biochar could mitigate NH_3 emissions following field application. The results showed that none of the two woody biochars tested caused significant reduction in NH_3 emissions in any of the test years, locations, or organic fertilizers (slurry vs. digestate) considered. Conversely, in some trial series, significant increases in NH_3 losses were observed, when considering cumulative TAN-input related NH_3 emissions. There are a few chemical and physical processes that could have contributed to this result, which are discussed below.

Biochars capacity to adsorb NH_4^+

The surface of biochar may carry a negative charge, creating an affinity for cations, including NH_4^+ (Fidel et al. 2018). The NH_4^+-N adsorption capacity of woody biochar produced at 700 °C has been reported to be of 2.2 mg g^{-1} (Feng et al. 2024), and one produced at 600 °C of 3.3 mg g^{-1} (Zheng et al. 2024). Keeping in mind these values, the biochar rates used in these experiment were well below those needed to mitigate NH_4^+ volatilization from slurry/digestate (Table S19). However, higher rates would result in excessively high DM content, which was identified as the main driver for elevated NH_3 emissions. Therefore, no higher biochar quantities, but biochars with a higher NH_4^+ adsorption capacity would be necessary. It must also be considered, that NH_4^+ bound to the biochar surfaces may have been released after a few days (Aghoghovwia et al. 2020; Cai et al. 2016; Viaene et al. 2023a) and possibly led to additional NH_3 losses (Thalhausen 2021; Burghausen 2023).

Biochar aging during co-composting was found to increase the NH_4^+ sorption capacity (Antonangelo et al. 2021; Nguyen et al. 2023); however, the effect of slurry/digestate and biochar storage period, which spans approximately four weeks in this experiment, on NH_3 volatilization after field application is unknown. Previous research has shown that biochar addition directly before pig slurry application increased NH_3 emissions (Schimmelpfennig et al. 2014), while biochar addition during anaerobic digestion had no effect (Viaene et al. 2024), and co-composting of biochar and pig manure reduced NH_3 volatilization following field application (Zhang et al. 2019).

Biochar admixture changed slurry and digestate properties

The DM content of all the slurry samples studied increased significantly with biochar addition (r = 0.92, Table S18). Previously, it was demonstrated that a higher DM content was negatively correlated with the infiltration rate of liquid slurry components into the soil, whereby the NH_3 fluxes were increased (Häni et al. 2016; Mkhabela et al. 2009; Pedersen et al. 2024). In some analysis, the slurry pH increased slightly due to biochar admixture (Table 3). Modifications in slurry pH alter the equilibrium between NH_4^+ and NH_3, resulted in elevated NH_3 losses (Langley-Randall et al. 2024; Qu and Zhang 2021). Additionally, the slurry strips exhibited a darker coloration due to biochar incorporation, a process that led to a reduction in the albedo (Usowicz et al. 2016). This may have led to temperature increases within the slurry bands during periods of cool, sunny weather. Ammonia fluxes from slurries increase with rising temperature from 5 °C to 14 °C; however, this is not clearly observed at higher temperatures (Pedersen et al. 2021). Another explanatory mechanism is based on our observations, which indicated that the slurry bands remained moist for a longer period of time with biochar amendment because biochar has the ability to retain water (Verheijen et al. 2019; Yu et al. 2013). The NH_3 losses from slightly moist slurry strips were higher because NH_3 and NH_4^+ are only in equilibrium when dissolved in water (Sigurdarson et al. 2018). Biochar typically contains low organic N concentrations in the range between 0.36–0.46 (PB) and 0.47–0.53% DM (PGB) (Table 1). E.g., with a MR in Baierlach 2023 of 30.2 kg m^{-3}, 0.14 kg m^{-3} biochar N were added to the slurry. This resulted in higher TN contents when PB was added to the cattle slurry. In contrast, the detected TAN in the digestate with PGB and, in some instances, in the cattle slurry with PB, were lower. Given the relatively low biochar NH_4^+ adsorption capacity (Table S19), this effect can be attributed to the dilution of mineral N in the organic fertilizer by the organic substance of the biochar. Furthermore, NH_3 emissions during the mixing before sampling cannot be completely excluded, as these were not measured. Thus, the combination of higher TN and/or lower TAN resulted in a lower quantity of applied TAN. Accordingly, despite similar total NH_3-N emissions, the proportion of emissions relative to the applied TAN was elevated.

Whey and Leonardite did not affect NH_3 volatilization

Whey had no effect on either cattle slurry pH or NH_3 volatilization which contrasts with previous studies in which biochar acidification reduced NH_3 emissions (Ro et al. 2015; Vieira Firmino et al. 2024). The whey pH (4.8) was not particularly low compared to other experiments (Braos et al. 2020; Jatana et al. 2023). Given that slurries are effective at buffering changes in pH (Overmeyer et al. 2020), a stronger acid or addition immediately before application may have had other impacts on NH_3 volatilization.

In contrast to the findings of Cao et al. (2022b), addition of leonardite did not reduce NH_3 volatilization in either of the trial years. The acidity provided by the leonardite was not sufficient to change the pH of the digestate in a relevant way. Concurrently, the DM content of the slurry increased with the addition of leonardite, slowing down the soil infiltration of the liquid slurry components (Sommer et al. 2006). If the leonardite had been incorporated shortly before application, the result might have been different, as the digestate pH rises again after acidification due to its buffering capacity (Finzi et al. 2024; Regueiro et al. 2016).

Conclusion

The results of the field trials indicate that biochar amendment did not reduce NH_3 emissions following the application of either cattle slurry or digestate. In some instances, NH_3 volatilization even increased. This effect was primarily attributable to the increase in slurry and digestate DM content resulting from the addition of biochar. Furthermore, the high biochar pH and the darker color of the slurry and digestate bands may have influenced the NH_3 losses. Therefore, it is not recommended to use woody biochar under the pedoclimatic conditions of this study as an additive by early admixture in slurry/digestate storage to reduce NH_3 losses from field applied slurry/digestate. This study did not investigate the impact of biochar on NH_3 emissions during slurry/digestate storage. Finally, when applying cattle slurry/digestate to agricultural land, it is important to stress the need to adhere to practices recommended by experts/policy makers for a specific pedoclimatic condition. To minimize NH_3 volatilization, this can include the application of slurry and digestate close to the soil surface with immediate soil incorporation on uncultivated fields, dilution with water, or acidification with sulphuric acid.

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Author contribution TS, EJ, FW, AP, HS, KJH: conceptualization and methodology. TS, EJ, FW, HS: investigation. TS, FW: formal analysis and data curation. AP: software and formal analysis. TS: wrote the original draft. EJ, FW, AP, KJH: writing-review and editing. KJH: supervision.

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Data availability We plan to make the data publicly available in the future. Until data and R scripts used will be made available on request.

Declarations

Conflict of interest The authors declare no competing interests.

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