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More nitrogen to the crop!

To increase the nitrogen fertiliser value of cattle slurry and reduce its ammonia emissions

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Cover: Separated cattle slurry applied with trailing shoes in a winter wheat crop on clay soil. Photo: Karin Andersson.

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Abstract

Nitrogen (N) is an essential nutrient in crop production. However, it is also a serious environmental pollutant. Therefore, maximising crop utilisation of applied fertiliser N, with minimal N losses to the environment, is crucial. The N fertiliser effect from cattle slurry, which accounts for 75 percent of all manure spread in Sweden, is generally low. Its high dry matter (DM) content leads to slow infiltration and this poses a risk of large ammonia losses after slurry application. A high DM content also means a high carbon/nitrogen (C/N) ratio, which reduces the plant availability of nitrogen. The focus of this PhD project was to study measures to increase the nitrogen fertiliser value of cattle slurry, through treatments that reduce the slurry's C/N ratio and subsequently increase nitrogen availability to crops, combined with measures to reduce ammonia losses after slurry application. Crop utilisation of slurry N, expressed as Mineral Fertiliser Equivalents (MFE), was measured through harvest measurements in field trials and a pot experiment, and ammonia losses were measured in separate field experiments. Nitrogen availability depending on slurry and bedding material properties was investigated in two soil incubation experiments. Both solid-liquid separation and anaerobic digestion of cattle slurry, which reduces the slurry's C/N ratio, increased slurry MFE. The relationship between slurry C/N ratio and nitrogen availability was also confirmed in the two incubation studies. Acidification of slurry with sulphuric acid effectively reduced ammonia losses from all slurry types. For untreated and digested slurry, this was also reflected in increased MFE values, but not for separated slurry. Slurry application with trailing shoes did in most cases not increase slurry MFE compared to trailing hoses. The study shows that there is potential to increase the N fertiliser value of cattle slurry, however, there are no simple solutions, and to achieve maximum effect, different measures must be combined.

Keywords: cattle slurry, slurry separation, biogas digestate, trailing hose, trailing shoe, acidification, ammonia emissions, nitrogen fertiliser value, C/N ratio

Mer kväve till grödan! Att öka kvävegödselvärdet hos nötflytgödsel och minska dess ammoniakutsläpp

Sammanfattning

Kväve är nödvändigt i all växtproduktion och är även det växtnäringsämne som tillförs i störst mängd. Samtidigt är det ett ämne som, om det inte tas upp av grödan, riskerar att orsaka allvarliga negativa effekter på miljö, klimat och hälsa. Att det kväve som tillförs i växtproduktionen utnyttjas maximalt, med minimala förluster, är därför av största vikt. Kvävegödselvärdet hos nötflytgödsel, som utgör 75 procent av den stallgödsel som sprids i Sverige, är generellt lågt. Dess höga TS-halt leder till långsam infiltration i samband med gödselspridning och därmed risk för stora ammoniakförluster. En hög TS-halt innebär också en hög C/N-kvot vilket minskar kvävets växttillgänglighet. Fokus för detta doktorandprojekt har varit att studera åtgärder för att öka kvävegödselvärdet hos nötflytgödsel, genom olika behandlingar som sänker gödselns C/N-kvot och därmed ökar kvävets växttillgänglighet, kombinerat med åtgärder för att minska ammoniakförlusterna i samband med spridning. Kvävegödselvärdet mättes genom skördemätningar i fältförsök och i ett krukförsök och ammoniakförlusterna mättes i separata fältförsök. Kvävets tillgänglighet beroende på egenskaperna hos gödsel och strömaterial undersöktes i två inkubationsförsök. Både separering och rötning av nötflytgödsel, vilket sänker C/N-kvoten hos gödseln, visade sig kunna öka dess kvävegödselvärde. Sambandet mellan C/N-kvot och kvävetillgänglighet bekräftades även i de två inkubationsstudierna. Surgörning av gödseln med svavelsyra minskade effektivt ammoniakförlusterna från alla gödseltyperna. Detta avspeglade sig även i ett ökat kvävegödselvärde för obehandlad och rötad gödsel, dock inte för separerad. Gödselspridning med släpskor förbättrade inte nämnvärt kvävegödselvärdet jämfört med släpslangar. Studien visar att det finns potential att öka kvävegödselvärdet hos nötflytgödsel. Det finns dock inga enkla lösningar, utan för att uppnå maximal effekt behöver olika åtgärder kombineras.

Sökord: nötflytgödsel, gödselseparering, rötrest, släpslang, släpsko, surgörning, ammoniakförluster, kvävegödselvärde, C/N-kvot

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I. Andersson, K., Dahlin, A.S., Sørensen, P. & Delin, S. (2024). Bedding material properties and slurry C/N ratio affect the availability of nitrogen in cattle slurry applied to soil. Frontiers in Sustainable Food Systems, 8. https://doi.org/10.3389/fsufs.2024.1393674

II. 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. Biosystems Engineering, 226, pp. 194-208. https://doi.org/10.1016/j.biosystemseng.2023.01.012

III. Andersson, K. & Delin, S. Increased nitrogen fertiliser value of cattle slurry by solid-liquid separation, anaerobic digestion, and slurry acidification. Manuscript.

Additional paper

Karin Andersson also contributed to the following paper that was not included in this thesis:

I. 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. Biosystems Engineering, 202, pp. 66-78. https://doi.org/10.1016/j.biosystemseng.2020.12.005

All published papers are published open access.

The contribution of Karin Andersson to the papers included in this thesis was as follows:

I. Planned the study together with the co-authors. Carried out laboratory work, data analysis, and manuscript preparation with assistance from the co-authors.

II. Planned the study together with the co-authors. Carried out field work, data analysis, and manuscript preparation with assistance from the co-authors.

III. Planned the study together with the co-authors. Participated in field work. Carried out data analysis and manuscript preparation with assistance from the co-authors.

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Abbreviations

ANR	Apparent nitrogen recovery rate
BD	Biogas digestate
C	Carbon
CH_4	Methane
CO_2	Carbon dioxide
CS	Untreated cattle slurry
DM	Dry matter
DW	Dry weight
K	Potassium
LF	Liquid fraction from solid-liquid separation of slurry
MFE	Mineral fertiliser equivalents
N	Nitrogen
N_2O	Nitrous oxide
NH_3	Ammonia
NH_4	Ammonium
NO_3	Nitrate
P	Phosphorus
SF	Solid fraction from solid-liquid separation of slurry
TAN	Total ammoniacal nitrogen

1. Introduction

Nitrogen (N) is vital for all living organisms, as it is a key component for the synthesis of cell constituents such as proteins and nucleic acids. In an agricultural context, N is one of the macronutrients that is essential for plant growth. It is usually the most growth limiting nutrient, and is important for the quality of agricultural products. Nitrogen in its most common form, dinitrogen gas (N_2), which makes up almost 80 percent of the earth's atmosphere, is unavailable for most plant species, and the main N uptake occurs as a reactive N form such as ammonium (NH_4^+) or nitrate (NO_3^-) (Kuypers *et al.*, 2018). Biological N fixation is important in the transformation of N_2 to reactive N species, and occurs in both marine and terrestrial ecosystems (Galloway *et al.*, 2004). The invention of the Haber-Bosch process in the early 20th century, and the subsequent production of mineral N fertilisers, has drastically increased the amount of reactive N globally. Indeed, anthropogenic N input to the global land area is now exceeding those from natural sources (Schlesinger, 2009).

Although the increased N input has had an enormous impact on the potential for food production worldwide it is also causing severe environmental and human health problems. This includes increased emissions of the potent greenhouse gas nitrous oxide (N_2O), leaching of NO_3^- causing eutrophication of lakes and streams, and ammonia (NH_3) losses to the atmosphere contributing to acidification of soil and water when deposited to downwind ecosystems (Robertson & Vitousek, 2009). Ammonia also contributes to increased levels of fine particulate matter in the air leading to respiratory illnesses and premature deaths (Wyer *et al.*, 2022).

Within the National Emission reduction Commitments Directive, EU member states have set national commitments regarding the reduction of air pollutants, and Sweden is among the countries that need to further reduce its NH_3 emissions to meet the targets (European Enviroment Agency, 2022). Optimal use of the N that is applied to crops is key to improving both the agricultural and economical outcomes of crop production, and to minimise its negative environmental impacts. In order to reduce the input of mineral fertilisers, the crop N utilisation from fertilisation with animal manure and other recirculated nutrient sources needs to be improved.

When N is applied in the form of animal manure or other organic fertilisers, the fraction of N taken up by the crop is often considerably lower compared to mineral fertilisers. One reason for this is that part of the N is present in organic form, which is mainly unavailable for plant uptake and only slowly mineralised (Webb *et al.*, 2013). For the mineral N fraction, there is the risk of losses through gaseous emissions of HN_3, N_2O, and dinitrogen gas (N_2), and through leaching and runoff of NO_3^-. There is also competition for the mineral N between soil microorganisms and plants, and the addition of organic fertilisers with a high proportion of carbon (C) to N, i.e. a high C/N ratio, leads to net N immobilisation and hence a lower manure fertiliser value (Pedersen *et al.*, 2020a; Delin *et al.*, 2012; Sørensen *et al.*, 2003).

Cattle slurry generally has a rather high dry matter (DM) content relative to other animal slurries and a low ammonium N content (Webb *et al.*, 2013), which implies a low crop N uptake from the total slurry N applied. Treatments that reduce the slurry's C/N ratio and hence increase crop availability of slurry N include solid-liquid separation and anaerobic digestion. A high slurry DM content also implies a higher risk of NH_3 losses after slurry application due to a slower slurry infiltration rate (Pedersen *et al.*, 2022). In Sweden, the agricultural sector contributes 90% of the country's total NH_3 emissions, and of this, 31% originates from the spreading of animal manure (Statistics Sweden, 2023a). To reduce NH_3 emissions from animal manure fertilisation and increase its fertiliser value it is important to explore in what ways manure treatment and application could be optimised. This study has focused on cattle slurry, since it constitutes the largest part of the animal manure applied to agricultural crops in Sweden (Statistics Sweden, 2023b).

Both slurry separation (Sindhöj & Rhode, 2013; Møller *et al.*, 2000), slurry acidification and low-emission slurry application techniques such as slurry injection (Hansen *et al.*, 2003) comes with an additional cost for the farmer. This has to be taken into consideration, since it will influence the implementation rate of the different management practices.

2. Aims and objectives

The overall aim of this doctoral thesis was to evaluate different strategies to increase the N fertiliser value of cattle slurry. These strategies consisted of pre-treatments reducing slurry C/N ratio, combined with measures to reduce NH_3 emissions after slurry application.

The specific objectives of the study were to:

- Quantify the NH_3 emissions from land application of cattle slurry with different pre-treatments and application strategies.
- Evaluate the importance of slurry properties and C/N ratio on slurry N availability after spreading.
- Obtain an overall picture of the differences in N fertiliser value between different slurry treatments and application strategies under varying conditions (different crops, soil types and years) through field trials.
- Analyse the economic implications of the different strategies included in the thesis, based on the increases in nitrogen fertiliser value.

3. Background

3.1 Nitrogen forms, transformations, and losses

Nitrogen is found in many different forms, and the transformations between them are mainly performed by microorganisms that alter the oxidation stage of N (Kuypers *et al.*, 2018). In animal manure, N is present in both organic and inorganic form. Though there may be some plant uptake of simple organic N such as amino acids (Näsholm *et al.*, 2009; Schimel & Bennett, 2004), N is predominantly taken up in inorganic forms. Mineralisation of organic N in manure is a slow process, ranging from months to years (Webb *et al.*, 2013), and the contribution of plant available N from organic N in manure is therefore limited. Contrastingly the inorganic N in manure, which mainly consists of ammonium ions (NH_4^+) and dissolved ammonia (NH_3), together called total ammoniacal nitrogen (TAN), is immediately available for plant uptake (Webb *et al.*, 2013). After soil application, NH_4^+ that is not taken up by plants is either oxidized by microbes to NO_3^-, through the nitrification process, or assimilated into microbial biomass (Stein & Klotz, 2016). Nitrogen assimilated by microbes is incorporated into the soil's organic N pool and only slowly re-mineralised (Sørensen, 2004). Just like NH_4^+, NO_3^- is readily available for plant uptake.

In all the processes and transformations, from the slurry spreader to plant uptake, there is a risk of N losses, which reduces the fertiliser value of the slurry. Total gaseous N losses after slurry application has been estimated to be between 10-50% of total N applied, of which NH_3 emissions constitute the largest part (Oenema *et al.*, 2008). Ammonia emissions after slurry application is dependent on many factors such as air temperature (Pedersen *et al.*, 2021b; Balsari *et al.*, 2008), crop and soil properties (Misselbrook *et al.*, 2002), and application technique (Webb *et al.*, 2010), and can, in some cases, be as high as >90% of applied TAN (Hafner *et al.*, 2018; Pfluke *et al.*, 2011). Nitrate can be lost through either surface runoff or leaching through the soil profile (Wang & Li, 2019), or reduced to nitrite (NO_2^-), nitric oxide (NO), nitrous oxide (N_2O), and dinitrogen (N_2) in the denitrification process (Stein & Klotz, 2016). Both the nitrification and denitrification process carry a risk of gaseous N losses in the form of N_2O and NO (Firestone & Davidson, 1989). Ammonia emissions and NO_3^- leaching losses are also a cause of indirect N_2O emissions, which contribute to climate change (Tian *et al.*, 2019).

3.2 Low nitrogen fertiliser effect from cattle slurry

The nitrogen fertiliser value of cattle slurry is relatively low compared to other organic fertilisers (Webb *et al.*, 2013). This can be linked to the two slurry properties: DM content and C/N ratio. Since slurry C is largely part of the fibre fraction, a high DM content also implies a high slurry C/N ratio. Cattle slurry often has a high DM content, and is on average higher in Sweden with a standard value of 9% (Andersson *et al.*, 2023), compared to 6.7% in Europe as a whole (Webb *et al.*, 2013). A high DM content leads to slower slurry infiltration and hence a risk of increased NH_3 emissions after slurry application (Pedersen *et al.*, 2021a).

A high C/N ratio increases N immobilisation after slurry application (Peters & Jensen, 2011), and immobilised N is only slowly re-mineralised (Sørensen, 2004), leading to reduced plant availability of slurry originating N. A negative linear relationship between C/N ratio of different organic amendments and their effect on crop growth has been shown by Delin *et al.* (2012) and Gutser *et al.* (2005), and more specifically for animal slurries by Pedersen *et al.* (2020a), Reijs *et al.* (2007), and Sørensen *et al.* (2003).

3.3 Ways to increase the nitrogen fertiliser value of cattle slurry

3.3.1 Reducing slurry DM content and C/N ratio

Within animal slurries, over 90% of the C is present in organic form as a constituent of the complex carbohydrates in the fibre fraction (Fangueiro *et al.*, 2013; Fangueiro *et al.*, 2009), while most of the N (50-70%) is in inorganic form (Webb *et al.*, 2013). Solid-liquid separation of slurry removes C-rich dry matter from the slurry, resulting in a liquid fraction with a lower C/N ratio, and a solid fraction high in organic matter and P, therefore enabling the possibility to relocate P from areas with a high animal density and P load (Hjorth *et al.*, 2010). The solid fraction can also be suitable as bedding material in housing systems for dairy and beef cattle production (Fournel *et al.*, 2019). For the liquid fraction, the lower C/N ratio leads to less N immobilisation and hence an increased N fertiliser value (Pedersen *et al.*, 2020a). The lower DM content increases infiltration rate and reduces the risk of NH_3 emissions after slurry application (Pedersen *et al.*, 2022).

Anaerobic digestion is another process that reduces slurry DM content and C/N ratio, where organic matter via several steps is converted to methane and CO_2 by anaerobic microorganisms. Cattle manure has a high content of lignocellulose, which is resistant to degradation, leading to inhibition of the first step of the process (hydrolysis) and hence a low biogas yield (Song *et al.*, 2023), and is therefore frequently co-digested with other substrates. The digestion process also changes the manure composition in other ways, including both an increased pH and fraction of TAN, and a lowered content of volatile fatty acids (Risberg *et al.*, 2017; Möller & Müller, 2012). Volatile fatty acids are an easily degradable C source which are rapidly consumed by soil microorganisms during the first days after slurry application, leading to N immobilisation (Kirchmann & Lundvall, 1993).

3.3.2 Mitigating ammonia emissions

Reducing NH_3 emissions results in more N remaining in the soil and is thus a potential way to increase the utilisation of slurry N in crop production. Ammonia emissions are affected by many different factors including soil- and slurry properties, application technique, and weather conditions (Hafner *et al.*, 2018). In addition to adjusting the timing of slurry application to favourable weather conditions, a range of other measures can be used to mitigate NH_3 emissions, e.g. reduce slurry DM to enhance infiltration rate (Pedersen *et al.*, 2022; Sommer *et al.*, 2006) and slurry acidification with different types of acids (Fangueiro *et al.*, 2015). Other alternatives include reducing the exposed surface area of the slurry through application techniques such as trailing hoses and trailing shoes, where slurry is applied in narrow bands (Webb *et al.*, 2010), and increasing the soil-slurry contact through slurry incorporation or injection (Nyord *et al.*, 2008; Sommer & Hutchings, 2001). The main difference between trailing hoses and trailing shoes is that there is pressure applied on the trailing shoes, and they are designed to make furrows in the soil surface (Figure 1). This results in narrower slurry bands and slurry placement directly on the soil surface, beneath the crop, while trailing hoses apply slurry both on the soil and on the crop.

Figure 1. Slurry spreader (Zunhammer) used in the fertilisation field experiments, equipped with trailing hoses (front row) and trailing shoes (back row).

3.4 Measurement and evaluation methods

3.4.1 Evaluating nitrogen fertiliser effect

There are different terms used to describe how efficiently the N added to a system is used to produce yield. This thesis uses terms and definitions as described by Jensen (2013). In the context of crop production, the term Apparent Nitrogen Recovery rate (ANR) describes the additional amount of N taken up by the plant from the fertiliser. It is calculated by comparing the increase in N uptake between two different N fertilisation rates (Equation 1) and is expressed as a percentage of additional applied N (Jensen, 2013).

$$ANR = \left(\frac{N_{upt2} - N_{upt1}}{N_{appl2} - N_{appl1}}\right) \times 100 \qquad \text{(Equation 1)}$$

Where N_{upt} is the amount of N taken up, in aboveground biomass or harvested product (e.g. in grains), and N_{appl} is the amount of N applied. When the N fertiliser value of organic fertilisers is evaluated, the two interchangeable terms Mineral Fertiliser Equivalent (MFE) and Nitrogen

Fertiliser Replacement Value (NFRV) are often used, and MFE will be used hereafter in this thesis. The MFE value describes how much mineral fertiliser N is needed to replace the amount of manure N applied. It is calculated according to Equation 2, and expressed as a percentage of applied manure N.

$$MFE = \frac{ANR_{man}}{ANR_{fert}} \times 100 \qquad \text{(Equation 2)}$$

Since the N uptake response is more or less linear for a wide range of N application levels (Jensen, 2013), MFE can also be calculated directly from the equation for a mineral N response curve (Equation 3). As illustrated in Figure 2, this is done by identifying the corresponding mineral N fertiliser rate (x_1) for the N uptake obtained with manure (y_1), and dividing it by the amount of manure N applied (x_{man}).

$$MFE = \frac{x_1}{x_{man}} \times 100 \qquad \text{(Equation 3)}$$

The MFE values are typically calculated based on the total amount of N applied with slurries, MFE_{totN}. However, in some cases MFE can also be calculated based on the amount of TAN applied, MFE_{TAN}, for example in this study, where the application rates were based on slurry TAN content.

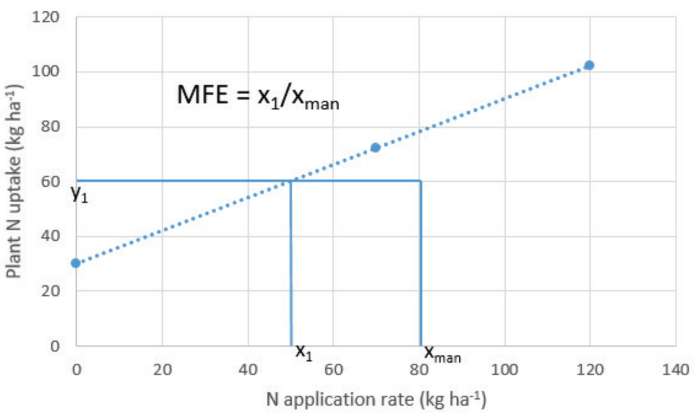

Figure 2. Calculation of Mineral Fertiliser Equivalents (MFE) for an organic fertiliser, based on the nitrogen response curve from mineral fertiliser nitrogen application.

3.4.2 To measure ammonia emissions

The techniques available for measuring NH_3 emissions in field experiments can be divided in two main categories: enclosure methods for small plot measurements and micrometeorological methods for larger areas up to field scale (Kamp *et al.*, 2024). Which method is most suitable depends on the aim of the study, resources, and available space. Among the micrometeorological methods, the Integrated Horizontal Flux (IHF) method is reliable and used as a reference technique for quantification of NH_3 emissions from field application of manure (Scotto di Perta *et al.*, 2020). For comparative studies with small-plot measurements, different types of chambers or wind tunnels are used. However, it should be noted that non-vented (static) chambers typically underestimate NH_3 emissions while the results from vented (dynamic) chambers are affected by the air velocity within the chamber (Scotto di Perta *et al.*, 2020). In previous Swedish NH_3 emissions experiments, dynamic chambers combined with passive diffusion samplers (Svensson, 1994) have been frequently used. In this study, wind tunnels were used in combination with online NH_3 measurements with a cavity ring-down spectrometer (CRDS) (Paper III).

3.5 Delimitation of the project

The focus of the project was on the effect of different slurry treatments and fertilisation strategies on short-term N availability and crop N uptake in the first growing season after slurry application. Quantification of N losses were limited to NH_3, therefore no measurements of nitrous oxide and dinitrogen emissions or NO_3^- leaching from the various fertilisation strategies was made. The slurry application strategies tested for NH_3 emission abatement was limited to slurry acidification with sulphuric acid and slurry application with trailing shoes instead of trailing hoses. No studies of the fertiliser value of the solid fraction from slurry separation was included, as the solid fraction produced was assumed to be re-circulated within the farm as bedding material for dairy cows. A simple calculation of the economy of the different slurry treatments was included, as the outcome of such an analysis could indicate how likely they are to be implemented by farmers.

4. Materials and Methods

The results reported in this thesis are based on a total of nine one-year fertilisation field experiments, one pot-experiment, five separate field experiments for measurement of NH_3 emissions and two laboratory incubation experiments.

4.1 Experimental sites, soils and manures

All fertilisation field experiments were carried out at two of SLU's field stations, Götala (58°22'N, 13°29'E) and Lanna (58°20'N, 13°7'E) in southwest Sweden. Soil for both the pot experiment and the two soil incubations was collected from Götala. The first incubation also included soil from Lanna, and all NH_3 emission experiments took place at Lanna. The soil at Götala is a sandy loam, while the soil at Lanna is a silty clay with over 40% clay content, and the variation in soil texture is relatively low between the fields within each site (Table 1). The crop rotation at Götala includes 3–4-year forage grass/clover leys and annual crops, and frequent fertilisation with cattle slurry and farmyard manure from cattle and sheep. The crop rotation at Lanna is cereal dominated and there has been no addition of animal manure during the last 50 years.

Table 1. Soil characteristics for the sites where fertilisation experiments were performed.

	Götala (n=4)		**Lanna (n=5)**	
	mean	**range**	**mean**	**range**
Sand (%)	69	67-71	11	7-14
Silt (%)	17	15-18	47	43-49
Clay (%)	14	14-15	42	41-45
Org matter (%)	3.4	2.9-4	2.9	2.6-3.3
pH (1:5 H_2O)	6.1	5.7-6.5	7.0	6.8-7.1
P_AL (mg 100 g^{-1} DW soil)	20.3	11-34	3.6	3.3-4.4
K_AL (mg 100 g^{-1} DW soil)	17	12-29	13	10-16
Mg_AL (mg 100 g^{-1} DW soil)	8.7	4.5-13	26.6	20-36
P_HCl (mg 100 g^{-1} DW soil)	163	130-190	43	37-49
K_HCl (mg 100 g^{-1} DW soil)	185	160-220	284	250-320

The sandy loam soil at Götala represents the most common soil type in Sweden (28% of the samples in a national soil inventory), followed by loam and silt loam (Djodjic, 2015). The soil type at Lanna, with over 40% clay content, is not as common, but is an appropriate representative for the most intensely farmed plain areas in Sweden.

The slurry types used were untreated cattle slurry (CS), the liquid fraction (LF) from solid-liquid separation of cattle slurry, also referred to as separated slurry in this thesis, and biogas digestate (BD). Untreated and separated slurry for the field experiments was collected from the same organic dairy farm once a year, in early spring before the first fertilisations, and stored in 1,000-litre plastic tanks in an unheated barn until use. Biogas digestate was collected from a commercial biogas plant at the same time of the year as the cattle slurries and stored in the same way. In the biogas plant, cattle slurry was co-digested in a substrate mix containing 65% cattle slurry, 20% pig slurry, and smaller fractions (less than 7% each) of chicken manure, food waste, and waste products from slaughterhouses and the food industry. Slurry characteristics are summarised in Table 2.

Table 2. Characteristics of the slurry types used in fertilisation field experiments.

	Untreated cattle slurry (CS)		**Separated cattle slurry (LF)**		**Biogas digestate (BD)**	
Experiments (n)	7		7		9	
	mean	range	mean	range	mean	range
DM (%)	8.5	7.3-9.8	5.3	4.6-6.1	5.3	4.8-5.9
Tot N (kg ton^{-1})	3.5	2.7-4.0	3.7	2.4-4.5	5.0	4.5-5.3
TAN (kg ton^{-1})	1.8	1.3-2.2	1.9	1.3-2.4	3.3	3.1-3.7
TAN / Tot N	0.50	0.44-0.54	0.51	0.44-0.57	0.64	0.46-0.69
Tot C / Tot N	10.7	9.8-12.4	6.4	5.4-8.1	4.3	3.7-4.8
Tot C / Org N	21.4	17.6-23.0	13.2	11.0-17.4	12.8	11.4-14.6
pH	7.1	6.8-7.7	7.0	6.9-7.2	8.0	7.6-8.4
pH acidified	5.9	5.3-6.4	6.1	5.8-6.4	6.7	6.2-7.1
H_2SO_4 (96%) added (kg ton^{-1})	6.7	5.5-9.2	5.0	3.7-7.2	16.5	14.7-19.3

4.2 Pot experiment with ryegrass

To test the effect of slurry C/N ratio on slurry N fertiliser effect, 34 cattle slurry samples collected on-farm were used in a 13-week pot experiment with ryegrass. Most of the samples originated from dairy farms, but there were also slurry samples from beef cattle. The slurry C/N ratios ranged from 7.5 to 13.9. The slurries were tested in triplicates, and three levels of mineral N application (0, 50 and 100 kg ha^{-1}) were included as reference. Each experimental unit (a plastic pot) was first filled with 3,000 g of soil, then a layer of cattle slurry containing 0.25 g total N, corresponding to 100 kg total N ha^{-1} was added, after which another 300 g of soil was added on top, to prevent NH_3 losses. The fraction of TAN varied between slurry samples, from 35% to 56% total N, with a mean of 50%. The soil was a sandy loam soil collected from Götala experimental farm (58°22' N, 13°29' E) with soil properties comparable to those for Götala in Table 1, and with a water-holding capacity of 350 g kg^{-1} dry soil (Jansson, 1958). In each pot, 50 seeds of ryegrass (*Lolium perenne*) were sown and covered with a thin layer of vermiculite, to keep the seeds moist until germination. The experiment was performed during September-December and the pots were kept in a greenhouse with extra light from mid-October and extra heating when needed to ensure the temperature was above 15°C (Figure 3). The pots were weighed and watered 2-3 times per week, to keep the soil moisture between 30 and 70 % of water-holding capacity. To avoid nutrient deficiencies, the pots were also fertilised with a nutrient solution containing a sufficient amount of macro- and micronutrients, except N, once a week. The grass was cut three times, after 5, 10, and 13 weeks. The grass samples were dried at 55°C and analysed for dry matter and total N content. A linear function was fitted to the N response curve from the three levels of reference mineral fertiliser application. Based on the three accumulated cuts, Mineral Fertilizer Equivalent values based on total N application with slurries (MFE_{totN}) were calculated, as described by Delin *et al.* (2012). Similarly, MFE values were calculated for the first grass cut, and for the first and second cut together.

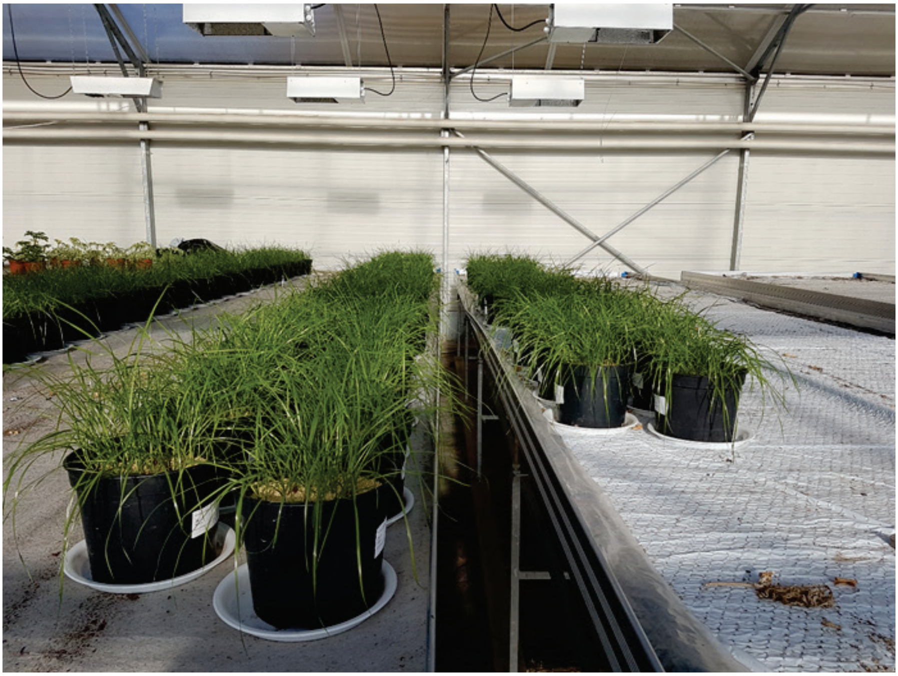

Figure 3. Pot experiment with ryegrass. The experiment included 34 different cattle slurries with three replicates.

4.3 Incubation experiments related to N availability

The results from the pot experiment highlighted the need to study slurry N availability in relation to slurry characteristics closer and under more controlled conditions in a laboratory incubation.

In a system where the solid fraction from slurry separation is used as bedding material, the manure fibre fraction is predominantly recirculated within the housing system. The most easily degradable C compounds are degraded first, which raised the question whether differences in the availability of C in different bedding materials could have any effect on the N turnover after slurry application. Would a more decomposed bedding material such as the solid fraction from slurry separation mean that the amount of readily available C becomes limiting, therefore reducing the immobilisation of N, and increasing the amount of plant-available N compared with e.g. straw? These questions were subsequently addressed in the two soil incubation experiments.

4.3.1 Incubation I – N availability depending on manure C/N ratio, bedding material, and soil type (Unpublished)

In this incubation, the soil N availability 28 days after manure application was studied for manure C/N ratios ranging from 6 to 18. To obtain the desired manure C/N ratios, different proportions of LF and bedding materials were added to the soil. The bedding materials used were either chopped wheat straw or the solid fraction from solid-liquid separation of cattle slurry (SF). One silty clay soil (Lanna) and one sandy loam soil (Götala) were compared. At the silty clay soil, intact soil cores were collected, using metal cylinders (height 50 mm, diameter 72 mm). At the start of the experiment, the soil cores were removed, cut in two, then LF and bedding material was added as a layer in the middle, to prevent NH_3 losses. Compared with thorough mixing of slurry and soil, this also better reflects the distribution of slurry in a field application situation with incorporation of slurry by ploughing or injection. Thereafter the soil cores were placed back into the metal cylinder. For the sandy loam, topsoil was collected from the field and then filled into the same type of metal cylinders that were used for the silty clay soil, also with LF and bedding material added as a layer in the middle. The cylinders were placed in plastic boxes in a climate chamber and incubated at 15°C for 28 days. The boxes were aerated and weighed regularly, and water was added when needed to keep the weight constant. Due to differences in soil bulk density, the amount of incubated soil was 460 g for the clay soil and 250 g for the sandy loam soil. The amount of total N added with manure was 100 and 120 mg kg^{-1} dry weight (DW) soil for the clay soil and the sandy loam soil, respectively.

At the end of the experiment, all soil samples were frozen and later analysed for mineral N content (NH_4^+ and NO_3^-). For the slurry treatments, Net mineral N release, expressed as a percent of added total N, was calculated according to Equation 4:

$$Net\ mineral\ N\ release = \frac{N_{min}\ in\ soil\ with\ slurry - N_{min}\ in\ control\ soil}{N_{tot}\ added\ with\ slurry}$$

(Equation 4)

4.3.2 Incubation II – N and C turnover depending on slurry C/N ratio and type of bedding material (Paper I)

The potential effects of differences in the fibre composition and degree of degradation of various bedding materials were studied in greater detail in a second incubation. The soil used for this was a sandy loam soil from the same site (Götala) as the first incubation, and the LF and bedding materials were also added as a layer in the middle of the soil. This time, each experimental unit consisted of a plastic flowerpot, containing 115 g of soil and 102 mg total N kg^{-1}, small enough to fit into the glass jar used for the CO_2 incubation. DW soil added with manure. For the N incubation, the same plastic boxes as in the first incubation were used, to maintain a relatively constant soil moisture content. The boxes were kept in the same climate chamber and temperature as in the first incubation, and on days 3, 7, 14, and 28, samples were taken out, frozen, and later analysed for mineral N content (NH_4^+ and NO_3^-). The samples for CO_2 measurements were kept in glass jars with a volume of 1.66 litres and with metal screw-lids, together with a CO_2 trap containing potassium hydroxide (KOH). Carbon dioxide production was measured on days 2, 5, 10, 15, 21, and 28. After the incubations, the fibre composition (neutral detergent fibre, acid detergent fibre and acid detergent lignin) of the bedding materials was analysed on non-incubated samples, according to the method described by Van Soest *et al.* (1991).

4.4 Ammonia emission field experiments

Field experiments with measurements of NH_3 emissions from application of different slurry types and different slurry application strategies were carried out after the first harvest in a grass ley on silty clay soil, at Lanna research station (Paper II). For the measurements of gaseous NH_3 emissions in this paper, a new system combining wind tunnels with continuous online NH_3 measurements with a cavity ring-down spectrometer (CRDS) (G2103 NH3 Concentration Analyzer, Picarro, CA, USA) was used (Pedersen *et al.*, 2020b) (Figure 4). This system has proved to be suitable for comparisons of different low-emission application techniques, due to its high measurement frequency and low variability. The setup consisted of nine wind tunnels, which allowed for three treatments with three replicated in each experiment. A total of five experiments were conducted. With trailing hose application of non-acidified slurry as reference, the two low-emission alternatives trailing shoe application and application of acidified slurry with trailing hoses were evaluated. This was done in three different experiments, one each for the three slurry types untreated cattle slurry, separated cattle slurry and biogas digestate. A fourth experiment compared NH_3 emissions from the three different slurry types, all applied with the reference technique trailing hoses. In the fifth experiment, the effect of slurry injection at two different depths was investigated. Slurry injection to a satisfactory depth of 5 cm was compared with an injection depth of 2 cm, reflecting conditions with a dry and hard soil surface. When the soil surface is hard, which is relatively common when spreading manure on clay soil in summer after the first grass harvest, the discs fail in penetrating to the desired depth.

Figure 4. Setup for the ammonia emission experiments. The nine wind tunnels were connected to online ammonia measurements with a cavity ring-down spectrometer.

4.5 Fertilisation field experiments

To test how different pre-treatments of cattle slurry and different low-emission application strategies affected the crop response, a total of nine field experiments were conducted in 2019-2021. Four experiments were carried out in winter wheat, one of which was at Götala and three at Lanna, and three experiments were conducted in grass ley, two of which were at Götala and one at Lanna. All these experiments included untreated cattle slurry, separated cattle slurry, and biogas digestate, and three levels of mineral fertiliser N as reference. Each of the slurry types were spread 1) non-acidified with trailing hoses, 2) non-acidified with trailing shoes, and 3) acidified with trailing hoses. The acidification was done by adding sulphuric acid to the slurries, resulting in a lowered slurry pH and thereby reduced NH_3 emissions. In the first year (2019) the trailing shoes performed poorly on the autumn-tilled, often hard soils in both grass ley and winter wheat. Therefore, two smaller experiments were carried out in 2020 with spring oats to test the trailing shoes on less dense, spring-tilled soil. These experiments included only one slurry type (biogas digestate) and application with trailing hoses and trailing shoes at two different application rates.

In winter wheat and spring oats, slurry application was performed at a late tillering stage, just before the beginning of stem elongation, in late Aril or early May for winter wheat and in late May or early June for spring oats. The grass ley experiments were fertilised after the first grass harvest of the year, in late May or Early June. Winter wheat and spring oats were harvested in August, while the grass was harvested twice after the slurry application, in July and September. In winter wheat and oats experiments, grain yield and N-yield, i.e. the amount of N found in the harvested product, was measured. In grass ley, grass DM yield and N-yield was measured for the two grass harvests following slurry application. To evaluate the crop response, ANR values were calculated for both mineral N and slurry treatments and MFE values were calculated for the slurry treatments, according to Equation 1 and Equation 3, respectively.

4.6 Economic evaluation

The economic calculations aimed at evaluating how large the cost of different slurry treatments could be, without exceeding the increase in slurry N fertiliser value. The MFE values for separated slurry and biogas digestate obtained from the fertilisation experiments (Paper III) were compared with the MFE value for untreated cattle slurry. The same calculation was also made for acidification of each of the three slurry types. The increase in N fertiliser value could then be used to evaluate the profitability of the different treatments. In the calculations, average values for the DM and nutrient content from the analyses of the different slurry types (n=7) and SF (n=2) were used.

The starting point for the calculations was 1,000 kg of untreated cattle slurry, with solid-liquid separation resulting in 883 kg liquid fraction and 117 kg solid fraction. For biogas digestate, the assumption was made that the same amount of total N was received back with the digestate as delivered with the slurry to the biogas plant. In this case, where other N-rich substrates were added at the biogas plant, this means that less biogas digestate was received back, and 1,000 kg of untreated cattle slurry corresponded to 718 kg of biogas digestate.

For the calculations related to slurry acidification, an example with an application of 60 kg TAN ha^{-1} was used, which was the same as the application rate in the grass ley experiments. The economic value of the solid fraction from slurry separation was calculated based on either its P and K content or its value as an alternative to sawdust as bedding material. The prices for N, P, and K was taken from the national Swedish fertilisation recommendations for 2024 (Andersson *et al.*, 2023).

5. Results and Discussion

5.1 Effect of slurry properties and slurry treatments

5.1.1 Pot experiment with different cattle slurries (Unpublished results)

In the pot experiment, ryegrass was fertilised with cattle slurries collected from different farms, to test the hypothesis that slurry N fertiliser value increases with a decreasing slurry C/N ratio. As expected, the results showed declining MFE-values with an increasing slurry C/N ratio, however, the correlation was weak both when looking at the first grass cut alone (r^2=0.075), and for two and three cuts (r^2=0.151 and r^2=0.145, respectively) (Figure 5). The variation between replicates was relatively low, with the coefficient of variation (CV) being less than 10% for all slurry treatments. The correlation between MFE and the amount of TAN (fraction of total N) applied was stronger (r^2=0.364) in the first grass cut but declined with the number of grass cuts (Figure 5). This is considerably lower than the study by Sørensen *et al.* (2003), who showed r^2-values of 0.67 for the correlation with C/N ratio and 0.53 for the correlation with the proportion of TAN in slurry.

As an average of all 34 cattle slurries included in the pot experiment, the MFE value of 52 % and the slurry C/N ratio of 9.5 corresponded well with the result for cattle slurry in a previous Swedish study with different organic fertilisers by Delin *et al.* (2012). However, within the group of slurry samples in this experiment, the negative linear correlation between slurry C/N ratio and MFE value was considerably weaker than expected.

Properties such as feed quality and bedding material, which potentially impact slurry N availability (Sørensen *et al.*, 2003; Sørensen, 1998), were not recorded for these slurries. Since the pot experiment was done with saved slurry samples from a previous project, and no new slurry analyses were made, one potential explanation for the poor correlation could be differences between the analysed slurry samples and the samples used in the pot experiment. This pot experiment showed a large variability within slurries from the same animal species, indicating that the correlation between C/N ratio and MFE, and between the fraction of TAN in slurry and MFE, should be seen as approximations rather than exact predictions.

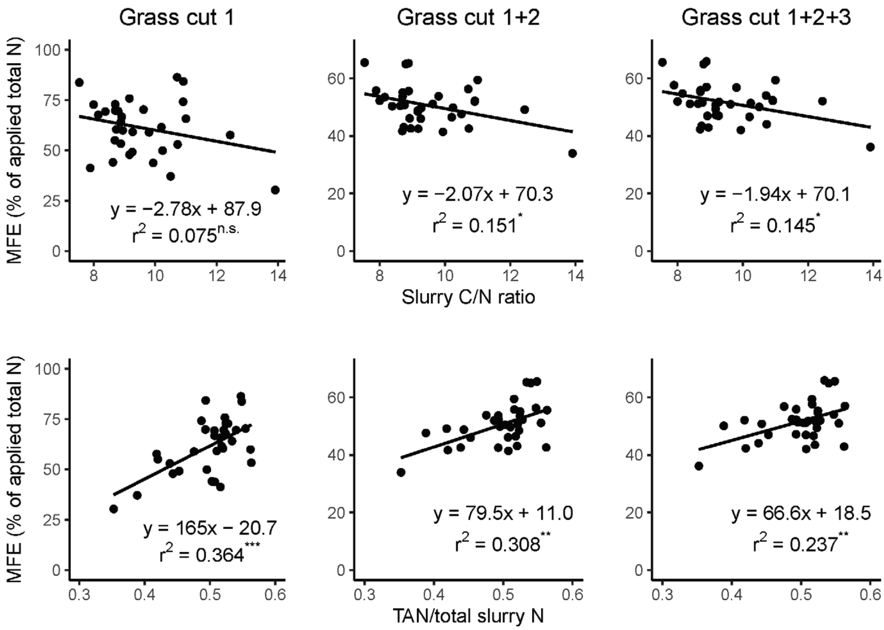

Figure 5. Mineral Fertiliser Equivalents (MFE) related to slurry C/N ratio (top row) and the fraction of TAN applied (bottom row) in the pot experiment with ryegrass.

5.1.2 Slurry C/N ratio - effect on N availability in different soils

The incubation studies included experimental slurries with different C/N ratios, consisting of mixes of LF and bedding material, either straw or SF. At experiment end, a negative linear relationship between slurry C/N ratio and plant available N, expressed as Net mineral N release (Equation 4), was hypothesised. It was also hypothesised that straw would provide a steeper slope of the relationship than SF, due to a higher content of non-degraded and easily available C, and thereby larger N immobilisation. In the first incubation (unpublished) both the silty clay soil and the sandy loam soil were used, whereas in the second incubation (Paper I) only the sandy loam was used. As expected, the results from the two incubations showed a negative linear relationship between slurry C/N ratio and Net mineral N release after 28 days (Figure 6). This is in line with what was hypothesised, and similar negative linear relationships between slurry C/N ratio and N fertiliser value have been shown by (Pedersen *et al.*, 2020a; Sørensen *et al.*, 2003). However, the incubation results showed no difference between straw and SF (Paper I).

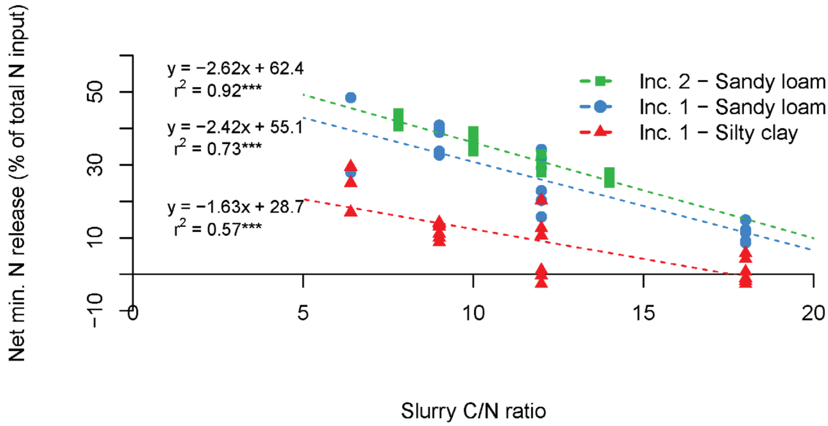

Figure 6. Net mineral N release after 28 days from manure with different C/N ratios at different soil types. Inc. 1 = Incubation 1, Inc. 2 = Incubation 2. Mineral N in a control soil was subtracted to estimate net N release.

Net mineral N release after 28 days incubation was, for all C/N ratios, lower for the clay soil compared with the sandy loam soil (Figure 6). At soils rich in 2:1 type clay minerals such as illite, which is the predominant clay mineral in the area southeast of lake Vänern (Stevens & Bayard, 1994), vermiculite and montmorillonite, positively charged ammonium ions are fixed between the negatively charged surfaces of the clay particle interlayers (Nieder *et al.*, 2011). However, Röing *et al.* (2006) showed that the amount of non-exchangeable ammonium N (NEA) in soils decreased during growth of ryegrass, indicating that part of the NEA is available for plant uptake within the course of a growing season. In the current incubation experiment, the 2M KCl used in the extraction process was likely not strong enough to reflect the amount of plant-available N in the clay soil.

5.1.3 Bedding material properties – effect on slurry N availability (Paper I)

It was hypothesised that straw as bedding material would lead to a greater immobilisation of N, due to a higher content of non-degraded, readily available C as an energy source for microbes, compared with SF. However, the results showed no short-term (28 days) difference between slurries with the two different bedding materials added (Figure 6 and Figure 7). The fibre analysis carried out after the incubation showed similar contents of hemicellulose, cellulose, and lignin in straw and SF, which may explain why they resulted in the same N availability (Paper I). A comparison between chopped and ground straw showed that the particle size of the bedding material did not affect N availability. A study by Ambus and Jensen (1997) showed an increased short-term (15 days) N immobilisation after the grinding of plant materials, while for the long-term (60 days) there were no differences (Ambus & Jensen, 1997).

Sawdust, with a significantly higher lignin content and lower hemicellulose content, had a higher N availability than straw and solid fraction after 28 days incubation (Figure 7). In accordance with what was hypothesised, slurry with an addition of sawdust had the lowest CO_2 production together with the highest N availability, indicating that the immobilisation of N may have been limited by a lack of readily available C (Paper I). For SF and straw which have similar distributions of cellulose, hemicellulose, and lignin, the results suggest that if the same amount of bedding material is added, SF (with a lower bedding material C/N ratio and hence a lower impact on the total slurry C/N ratio) would result in a higher slurry N availability than straw. With sawdust, larger amounts of un-degradable C is added in the form of lignin, and the incubation results indicate reduced N immobilisation and thereby increased N availability, due to limited C availability.

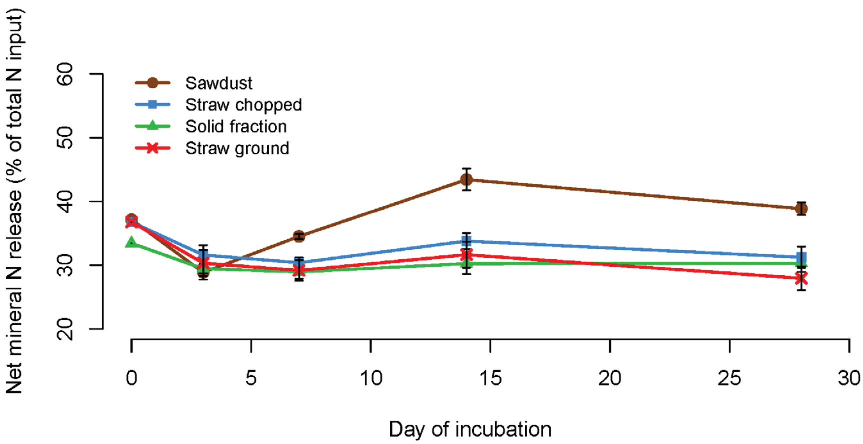

Figure 7. Net mineral N release from slurries with C/N ratio 12 and different bedding materials in a soil incubation.

5.1.4 Ammonia emissions (Paper II)

The NH_3 emission experiments were performed in grass ley on clay soil in June, after the first grass harvest of the year. In the experiment with trailing hose application of untreated and separated cattle slurry, and biogas digeste (Experiment 4), cumulative NH_3 emissions 70 hours after slurry application amounted to 23-32% of the added TAN (Figure 8), corresponding to 16.4-19.4 kg N ha^{-1}. The slurries with reduced DM content had both the highest emissions (biogas digestate) and the lowest (separated slurry), while untreated slurry showed intermediate cumulative emissions. Thus, the hypothesis that a lower slurry DM content would reduce NH_3 emissions, could not be confirmed. In this experiment, the difference between untreated and separated slurry was not statistically significant, while in two similar experiments on oat stubble at Lanna and Götala (Pedersen *et al.*, 2021a), separated slurry had lower NH_3 emissions than untreated slurry at the clay soil, but higher emissions at the sandy loam soil. Biogas digestate, with the lowest DM content but a higher pH, had similar or higher NH_3 emissions than the other slurry types in all three experiments (Paper II and Pedersen *et al.* (2021a)). As shown by Pedersen *et al.* (2021a), the exposed surface area of the slurry, i.e. the extent that the slurry bands are spread out, during the first hours after slurry application is important for the cumulative emissions.

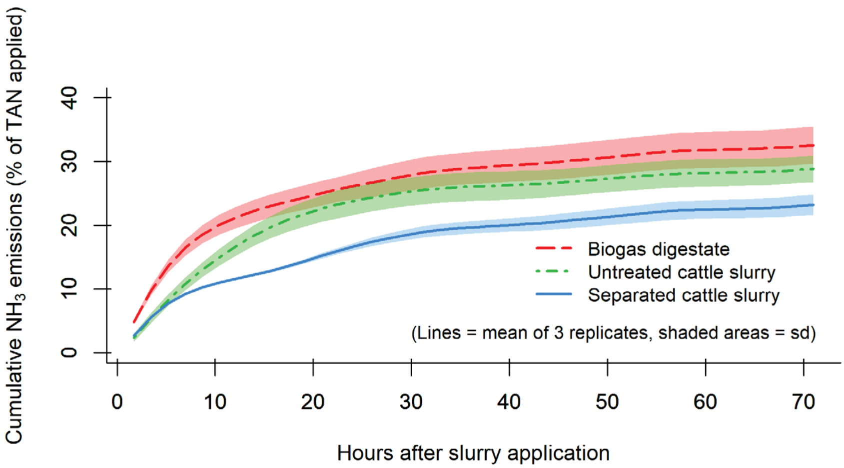

Figure 8. Cumulative ammonia emissions after slurry application to grass ley on clay soil, depending on slurry type.

However, there are several factors influencing the slurry surface area, such as slurry viscosity and particle size distribution, and soil texture and moisture. Therefore NH_3 emissions are difficult to predict.

All slurry types showed the highest emission rate immediately after slurry application, seen as the steepest slope of the emission curves during the first five hours in Figure 8. It was higher for biogas digestate, with a higher pH, than for untreated and separated slurry. Untreated and separated slurry had similar initial emission rates, but as shown in Figure 8, the emissions from untreated slurry continued at a high rate for a longer time, resulting in overall higher cumulative emissions. This can be linked to the higher DM content of untreated slurry, which reduced slurry infiltration rate and thereby prolonged the NH_3 emissions. In the five NH_3 emission experiments included in this study, 50% of the total emissions from biogas digestate occurred within the first 5-7 hours after slurry application, and within 10-14 hours for untreated and separated slurry. This indicates that biogas digestate is particularly vulnerable to unfavourable weather conditions such as a high temperature, solar radiation, and wind speed during slurry application (Paper II).

5.1.5 Nitrogen fertiliser effect in crop production (Paper III)

In winter wheat and spring oats experiments, the apparent N recovery rate (ANR) for mineral fertiliser N application, was generally 50-60% of total added N, while 65-85% in the grass ley experiments (Figure 9). This can be considered typical for crop cultivation in temperate climate (Jensen, 2013). For all slurry treatments, the ANR-values were considerably lower compared with mineral fertiliser application. The results followed the same patterns in both crops, but with larger experimental variation in grass ley (Paper III). When the same crop was grown at both sites in the same year, ANR-values in slurry treatments were consistently lower at the clay soil site (Figure 9). This may indicate a higher N supply from the sandy loam soil, which had regularly been fertilised with animal manure and has a slightly higher soil OM content. However, this was not reflected in the crop N-uptake in unfertilised control treatment, which was similar at the two sites (Paper III). It could also be due to more ammonium adsorption to clay minerals at the silty clay, as indicated by the incubation study (Paper I). A third explanation could be a more rapid slurry infiltration at the sandy loam soil and thereby lower NH_3 emissions. However, no such consistent relationship was observed in the NH_3 emission experiments by Pedersen *et al.* (2021a).

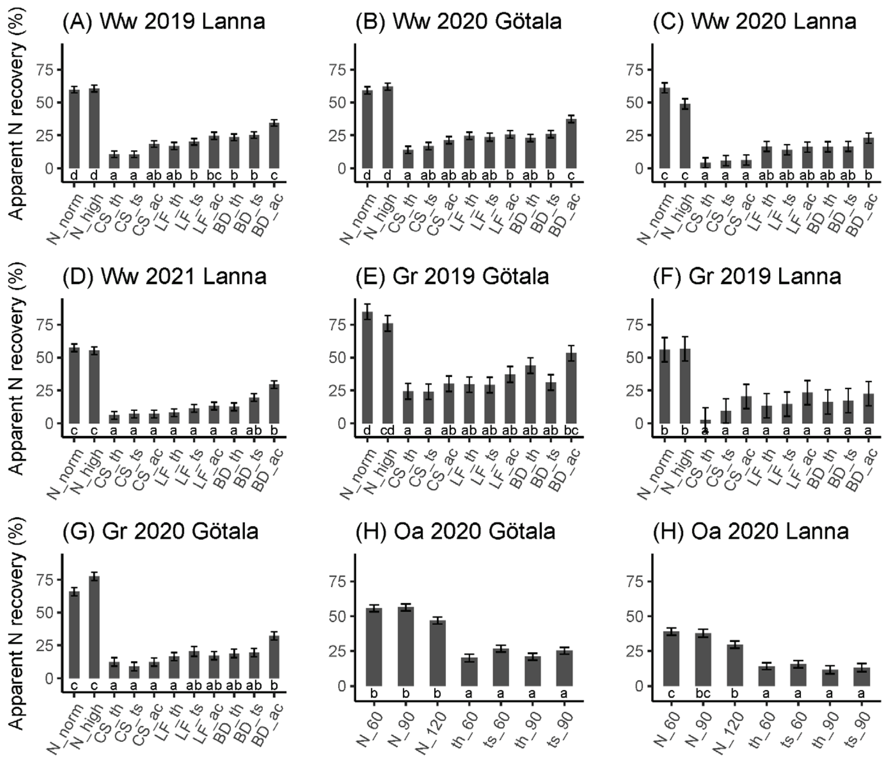

Figure 9. Apparent nitrogen recovery (ANR) in each of the fertilisation experiments. Ww = winter wheat, Gr = grass ley, Oa = spring oats. N_norm, N_high, N_60, N_90, and N_120 = mineral N application rates. CS = untreated cattle slurry, LF = separated cattle slurry, BD = biogas digestate, th = trailing hoses, ts = trailing shoes, ac = acidified.

When comparing the fertiliser effect of slurries in relation to mineral fertiliser N, the Mineral Fertiliser Equivalent (MFE) values are reported for all seven experiments in winter wheat and grass ley together. This is because the ANOVA showed no difference between the two crops, and no interactions between the factor *crop* and any of the other factors in the analysis. Trailing hose and trailing shoe application of untreated cattle slurry showed the lowest MFE values (30% MFE_{TAN}). Both separated cattle slurry and biogas digestate had higher MFE_{TAN} values (50-55%) (Figure 10). Slurry acidification increased MFE for untreated slurry and biogas digestate, but not for separated slurry. Trailing shoe application did not increase MFE for any of the slurry types compared with trailing hoses.

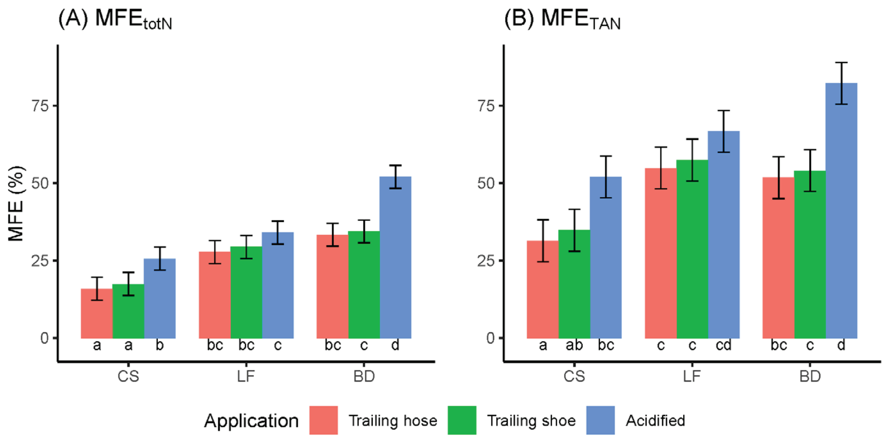

Figure 10. Mineral fertiliser equivalents (MFE) as percent of (A) added total N and (B) added ammonium N (TAN). CS = untreated cattle slurry, LF = separated cattle slurry, BD = biogas digestate.

Based solely on slurry C/N ratio (Table 2) and the negative correlation with N availability shown in Paper I, and with slurry N fertiliser effect seen in previous studies (Pedersen *et al.*, 2020a; Sørensen *et al.*, 2003), one could expect even higher MFE values for biogas digestate than for separated cattle slurry. However, in this study, the higher pH and thereby larger NH_3 emissions from biogas digestate (Paper II), seemed to balance out the positive effects of lower DM content and C/N ratio, resulting in similar MFE values for biogas digestate and separated slurry.

The apparent N uptake efficiency was lower in this study compared with the literature. Average ANR values in winter wheat and grass ley for non-acidified CS and LF, 9-21% of applied total N, were somewhat lower compared with results reported e.g. by ten Huf *et al.* (2023) and Nyameasem *et al.* (2022). For MFE_{totN}, the values of 28% for non-acidified LF and 34% for acidified LF in this study, were considerably lower than the 39% and 63% reported for cattle slurry with similar DM content by Sørensen and Eriksen (2009). Many factors influence the N uptake efficiency, but the high soil clay content in this study, with over 40% clay at Lanna and 14-45% at Götala (Table 1), compared with 7-15% clay in most of the other reported experiments (ten Huf *et al.*, 2023; Nyameasem *et al.*, 2022; Sørensen & Eriksen, 2009), most likely played an important role, with more ammonium adsorption to clay minerals.

5.2 Slurry acidification

5.2.1 Ammonia emissions (Paper II)

In the three NH_3 emission experiments (1, 2, and 3) wherein different application strategies were compared for one slurry type at a time, a pH reduction of approximately 1 pH unit (to 6.0 for untreated and separated slurry, and 6.7 for biogas digestate) reduced the NH_3 emissions by over 75% for all three slurry types, to only 2-6% of added TAN (Figure 11, panel 1-3), or 1.3-3.8 kg ha^{-1}. Slurry pH is a key factor in determining the NH_3 emissions, and emission reductions by >90 % after application of cattle slurry acidified to pH 5.5 has been reported by Keskinen et al. (2022) in grass ley and by Fangueiro et al. (2018) in maize and oats. However, the amount of acid needed implies a higher sulphur supply than the crop needs, and this is also costly. Therefore, under practical agricultural conditions, a lower dose of acid, which still provides an acceptable NH_3 abatement, would be desirable. Under controlled laboratory conditions, a more moderate acidification to pH 6.5 reduced the NH_3 emissions by over 80% (Ellersiek & Olfs, 2024).

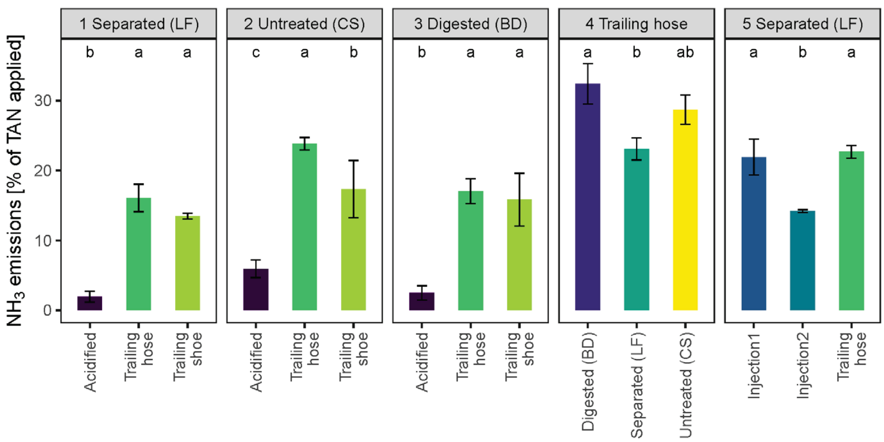

Figure 11. Cumulative ammonia emissions 70 hours after slurry application. Panel 1, 2, 3, and 5 show the results of experiments comparing different low-emission application techniques, with one slurry type in each experiment. Panel 4 shows a comparison between all three slurry types, applied with trailing hoses.

Under field conditions, the emission reduction is more variable, with e.g. Wagner et al. (2021) reporting a reduction by 61-67% from acidification of cattle slurry from pH 7.3-7.4 to pH 6.2-6.5, and Pedersen et al. (2022a) reporting a reduction by 6-16% from acidification from pH 6.9 to pH 6.4-6.5.

5.2.2 Crop response (Paper III)

For both untreated slurry and biogas digestate, acidification increased the MFE values by over 50% (Figure 10). For separated slurry, however, the MFE values were relatively high even without acidification (especially in the two winter wheat experiments in 2020), and the effect of acidification was therefore not significant (Paper III). Acidification to below pH 6 largely eliminates the NH_3 losses after slurry application (Fangueiro *et al.*, 2015), and hence the acidified treatments can be viewed as an indicator of the maximum fertilisation potential of each slurry type, showing the effect that can be obtained if (by acidification or otherwise) NH_3 losses are minimised.

The MFE values from acidification of untreated cattle slurry in the field experiments can be compared with the results from the pot experiment with ryegrass, where the NH_3 emissions were minimised by soil incorporation of the slurries. For acidified cattle slurry in all experiments (both winter wheat and grass ley and on both soil types), the MFE-values were in line with or lower than the lowest MFE-value in the pot experiment. This is not surprising because other factors that could also reduce the yield potential of the crop, such as water stress and N losses through NO_3^- leaching, were minimised in the pot experiment.

5.3 Trailing shoes and slurry injection

5.3.1 Ammonia emissions (Paper II)

Trailing shoes are used to create soil slots with loosened soil, increasing the contact between slurry and soil, enhancing slurry infiltration, and making the applied slurry stay in narrow bands, thereby reducing the NH_3 emissions. Slurry application with trailing shoes reduced NH_3 emissions from untreated cattle slurry by 27% compared with trailing hoses (Figure 11, panel 2), which was in line with the hypothesis. However, no positive effect was observed for the other slurry types (Figure 11, panel 1 and 3). The soil surface was dry and hard at slurry application, resulting in a poor performance of the trailing shoes. For separated slurry, spreading out of the slurry beside the trailing shoe soil slots, which resulted in rather wide slurry bands both with trailing hoses and trailing shoes, is a plausible explanation. For biogas digestate, on the other hand, a lower application rate resulted in narrow slurry bands both with trailing hoses and trailing shoes, and hence similar NH_3 emissions. These experiments were performed on clay soil, and the effect of trailing shoe application may have been larger on a more coarse textured soil. Pedersen *et al.* (2020b) reported differences in trailing shoe performance depending on soil texture, with reduced NH_3 emissions from trailing shoe application on coarse sand, but not on loamy sand or sandy loam soils.

For the simulated slurry injections, surface band application of separated slurry on 50 mm deep slots with loosened soil (Injection 1) reduced NH_3 losses by 37%, compared with trailing hose application (Figure 11, panel 5). Slurry application on 20 mm deep injection slots (Injection 2) did not reduce the NH_3 emissions (Figure 11, panel 5). This confirms that the NH_3 reduction potential of slurry injection is positively correlated to the depth and volume of the injection slots (Hansen *et al.*, 2003), and when the injector is unable to inject the slurry deeper than 20 mm, no reduction in NH_3 emissions compared with band application is observed (Nyord *et al.*, 2008; Rodhe & Etana, 2005). A similar technique, but with a machine creating the soil aeration slots and applying the slurry bands, has been tested by Bittman *et al.* (2005) and Pedersen *et al.* (2021c), showing reduced NH_3 emissions by 35%-48% compared with surface band application without aerating the soil.

5.3.2 Crop response (Paper III)

Despite the reduced NH_3 losses shown for trailing shoe application of untreated cattle slurry (Figure 11), no positive effect on the MFE was observed in the fertilisation experiments in winter wheat and grass ley (Figure 10). As in the NH_3 emission experiments, the soil surface in the fertilisation field experiments was in many cases dry and hard at slurry application, and the force on the trailing shoes was insufficient in creating soil slots that could contain all applied slurry, leading to the spreading out of the slurry. As the exposed slurry surface area is an important factor affecting the NH_3 emissions (Pedersen *et al.*, 2021a; Pedersen *et al.*, 2021c), the spreading out of slurry beside the trailing shoe soil slots could explain why no positive effect was observed.

In the experiment carried out on spring-tilled soil in oats on sandy loam soil (Götala), trailing shoe application of biogas digestate increased the MFE value compared with trailing hose application by 28% (Paper III). The experiment on clay soil (Lanna) was severely affected by an early summer drought, leading to low yield in all treatments and no positive effect of trailing shoe application. The results show that with the type of trailing shoes tested (Zunhammer) it is difficult to obtain any increase in N fertilisation effect compared with trailing hose application, except on newly cultivated, coarse textured soil.

5.4 Economic evaluation

5.4.1 Value of solid-liquid separation and anaerobic digestion

Based on data from the fertilisation field experiments, untreated cattle slurry contained on average 1.78 kg TAN ton^{-1} slurry, but due to N immobilisation and N losses the MFE value (i.e. the corresponding amount of mineral fertiliser N) is only 0.56 kg N ton^{-1} slurry (Table 3). Solid-liquid separation reduces the amount of slurry somewhat since part of the fibre fraction is removed, but despite this, slurry separation increased the MFE value to 0.91 kg N ton^{-1} untreated slurry. With an N price of 14 SEK kg^{-1} N, this means that slurry separation increased the value of cattle slurry by 4.91 SEK ton^{-1}, only considering the value of the liquid fraction. What value that the solid fraction (SF) is assigned is of course also highly important. If the SF is used as bedding material and assigned the same value as the alternative bedding material sawdust, the value of SF per ton slurry that is separated is 116.5 SEK (117 kg × 0.996 SEK kg^{-1} sawdust). If not used as bedding material, SF can be valued based on its P and K content, which provides 7.40 SEK ton^{-1} slurry that is separated.

To evaluate the profitability of slurry separation, the increase in slurry value can be compared with the costs associated with the treatment. Calculations by Poulsen *et al.* (2019) for Danish conditions give a cost for the separation of slurry in the range of 1.9-2.4 DKK (2.95-3.73 SEK). However, this was based on a high (80%) capacity use of the separator.

Table 3. Calculation of the additional economic value of cattle slurry after solid-liquid separation or anaerobic digestion.

	Untreated cattle slurry (CS)	Liquid fraction (LF)	Solid fraction (SF)	Biogas digestate (BD)
Amount of slurry/SF in the calculations (tons)	1.000	0.883	0.117	0.718
DM (%)	8.67	5.28	34.24	5.33
TAN (kg ton^{-1})	1.78	1.87	0.81	3.33
Amount of TAN (kg ton^{-1} untreated slurry)	1.78	1.65	0.09	2.39
MFE (% of applied TAN)	31.4	54.9	-	51.8
MFE (kg ton^{-1} untreated slurry)	0.56	0.91	-	1.24
Increase in MFE (kg ton^{-1}) relative to untreated slurry	-	0.35	-	0.68
Added value (SEK ton^{-1} untreated slurry)[a]	**-**	**4.91**	**7.40-116.53**	**9.52**

A recent Swedish study by Jeppsson *et al.* (2024) showed a lower average bedding material cost for the farms using SF as bedding material compared with wood shavings/sawdust. However, the cost varied widely with the amount of slurry separated and was lowest at the larger farms.

Calculated in the same way as for slurry separation, anaerobic digestion of cattle slurry for biogas production increased the value by 9.52 SEK ton^{-1} untreated slurry (Table 3). This is based on the assumption that the same amount of total N is received back with biogas digestate as is delivered to the biogas plant with untreated slurry. Since other, N-rich substrates are added at the biogas plant, this means that the total N content is higher in the biogas digestate than in the untreated cattle slurry, and the amount of digestate is therefore lower. If instead the assumption is made that the farmers receive the same amount of wet weight digestate as the amount of untreated slurry they deliver, which is the case at the biogas plant where the digestate in this project comes from, the value increase is even greater, in this example 16.32 SEK ton^{-1}.

Whether the increased value of the slurry by separation or anaerobic digestion covers the costs of the slurry treatments at farm level depends on a number of factors and must be evaluated on a case-to-case basis.

5.4.2 Value and cost for slurry acidification

In this section, the increase in slurry N value by acidification of untreated and separated cattle slurry, and biogas digestate is exemplified for a fertilisation rate of 60 kg TAN ha^{-1} (Table 4). Since the amount of TAN differs between the slurry types, the slurry application rates differ. The amounts of acid needed for the desired pH reduction also differ between the slurry types, affecting the cost of acid addition.

Table 4. Additional slurry value obtained by slurry acidification, and associated costs. Example with an N application rate of 60 kg TAN ha^{-1}.

	Untreated cattle slurry (CS)	**Liquid fraction (LF)**	**Biogas digestate (BD)**
TAN (kg ton^{-1} slurry)	1.78	1.87	3.33
Slurry application rate (tons ha^{-1}) for 60 kg TAN	33.8	32.0	18.0
MFE non-acidified (% of TAN added)	31.4	54.9	51.8
MFE acidified (% of TAN added)	52.0	66.8	82.2
MFE non-acidified (kg ha^{-1})	18.8	32.9	31.1
MFE acidified (kg ha^{-1})	31.2	40.1	49.3
Increase in MFE acidified (kg ha^{-1})	12.4	7.1	18.2
Additional slurry N value (SEK ha^{-1})[a]	**173.0**	**100.0**	**255.4**
Amount of acid added (kg ton^{-1} slurry)[b]	6.7 (3.0)	5.0 (3.0)	16.5 (11.0)
Sulphur added with acid (kg ha^{-1})[b]	74.0 (33.2)	52.4 (31.4)	97.3 (64.9)
Value sulphur added (SEK ha^{-1})[c]	100	100	100
Cost for acid addition (SEK ha^{-1})[d]	78.8	74.6	42.0
Left for paying the acid (SEK kg^{-1} acid)[b]	**0.86 (1.93)**	**0.79 (1.31)**	**1.05 (1.58)**

[a]N price = 14 SEK kg^{-1}, [b]Values within parentheses are based on the Danish requirements for acid addition), [c]S price = 5 SEK kg^{-1}, the sulphur value is limited to crop demand i.e. max 20 kg S ha^{-1}, [d]cost based on Danish values i.e. 1.5 DKK ≈ 2.30 SEK m^3 slurry.

The increase in slurry N value from acidification is largest for biogas digestate and smallest for separated slurry (Table 4). The sulphur added with sulphuric acid also has a fertiliser value, however, the amounts added exceed the crops demands, and in this example, I have limited the value to the maximum amount of sulphur needed by the crop. For biogas digestate, the higher slurry N value from acidification is balanced out by a much larger amount of acid required to obtain the same NH_3 abatement. Therefore, for acidification to still be a profitable alternative, the maximum cost that can be paid per kg sulphuric acid added is similar for the three slurry types. In the Danish legislations the amounts of acid required are lower, and if those are used in the calculations (values within parentheses in Table 4), there is room for a somewhat higher cost for the acid. However, those amounts of acid would likely not result in the same increase in slurry N fertiliser value as seen in the field experiments in this study.

Based on these calculations, slurry acidification is difficult to motivate from a purely economic perspective. However, the NH_3 abatement obtained has positive effects on the environment and on human health.

5.5 Limitations of the project

From an environmental perspective, a potential trade off from slurry separation is increased NH_3 emissions from composting of the solid fraction (Fangueiro *et al.*, 2008; Amon *et al.*, 2006), or a risk of nitrous oxide emissions if the solid fraction is stored without air supply (Holly *et al.*, 2017). However, emissions from storage and application of the solid fraction were not included in this study.

Ideally, the study would have included fertilisation experiments in both crops at both sites each year, to enable a better comparison of differences depending on soil type. Experimental sites with more sandy soils could also have been considered, to gain results more comparable with studies from other countries. However, in Sweden these soils comprise only 6% of the agricultural area (Djodjic, 2015), and the sites and soil types chosen can be considered representative for Swedish conditions.

Better equipment for injection of slurries would have been a more interesting alternative to slurry acidification than the trailing shoes used, especially considering that the soils were loam and clay soils, where it is difficult to achieve optimal performance of trailing shoes. There are a number of different slurry injector types, both for open and closed slot injection. However, the ability of the injectors to place the slurry deep enough is a challenge, which has been shown in several studies, e.g. by Rodhe and Etana (2005), Nyord *et al.* (2008), and (Seidel *et al.*, 2017).

An attempt has been made to compare the different slurry treatments in terms of N gained and economic value. A more detailed and thorough system analysis including different alternative uses of the solid fraction, trade-off effects in the form of greenhouse gas emissions etc., would have made the results even more interesting and more applicable, both for the research community and at the farm level.

5.6 Conclusions

From this study, with the combination of fertilisation field experiments, NH_3 emission measurements, and laboratory incubations, the following practically useful conclusions can be drawn:

- There are several ways in which the N fertiliser value of cattle slurry can be increased. For untreated cattle slurry, solid-liquid separation, co-digestion with other substrates for biogas production, and slurry acidification resulted in similar increases of the MFE value. An even larger increase was obtained from the combination of anaerobic digestion and acidification.
- Biogas digestate, with its high TAN content, has great potential as a fertiliser. However, a high pH means that the N is particularly prone to volatilisation and thereby easily lost. Ammonia emissions are also quicker from biogas digestate, with half of the total losses occurring within 5-7 hours after slurry application, compared with 10-14 hours for untreated and separated cattle slurry.
- Slurry acidification reduces NH_3 emissions for all slurry types and increases the MFE values for untreated cattle slurry and biogas digestate. However, with the amount of acid needed to reach the desired emission reduction, the increased slurry value is not enough to pay for the acidification.
- Solid-liquid separation of cattle slurry reduces NH_3 emissions and increases MFE values from the liquid fraction. Acidification of the liquid fraction does not increase MFE further.

- Slurry application with trailing shoes in grass ley and winter wheat in April-June does not increase slurry N fertiliser value compared with trailing hose application. On a spring-cultivated sandy loam soil, trailing shoe application of biogas digestate to spring oats increased MFE by 28 percent.
- The economic calculations show that slurry acidification is difficult to motivate from a purely economic perspective. However, it is a highly efficient measure to reduce ammonia emissions. Use of the solid fraction from slurry separation as bedding material increases the economic value of the separation, but the environmental effects need to be further studied.

5.7 Future perspectives

Increasing the N fertiliser value of cattle slurry is possible, but there is not one simple solution that can solve all problems. A combination of different measures seems to have the best effect, e.g. in this study the combination of anaerobic digestion and acidification resulted in the highest MFE values.

Sweden needs to reduce its NH_3 emissions at a national level, to meet the targets of the National Emission reduction Commitments Directive within the European Union (European Enviroment Agency, 2022). However, mitigating NH_3 emissions with either minimal or no trade-off effects on greenhouse gas emissions is a challenge. A meta-analysis by Emmerling *et al.* (2020), including the whole slurry management chain, concluded that slurry acidification was the only technique that reduced NH_3 emissions without causing "pollution swapping" in terms of increases in at least one of the greenhouse gases CO_2, CH_4 or N_2O.

As demonstrated in this thesis, slurry acidification is effective in reducing NH_3 emissions and increasing the N fertiliser value of high DM slurries and biogas digestate. However, there are certain obstacles such as the cost of acidification, the risks with handling concentrated acid, and issues with excessive foaming when the acid is added, along with the fact that sulphuric acid is not an allowed additive in organic farming. In Denmark, where slurry acidification was widely used as a low-emission application technique until recently, the use has drastically decreased after the implementation of new rules, regulating the amount of acid required per ton of manure, and the use of slurry injection has instead increased (Gerth Pedersen Holm, DME, personal communication). Slurry separation, with a maximum DM content of 3.6% and 3.9% for the liquid fractions from digested slurry and untreated cattle slurry, respectively, has been added as an alternative low-emission technique (Denmark. Ministeriet for Fødevarer Landbrug og Fiskeri, 2024). Simple changes in farmer practices can also have a great impact, e.g. a study with experiments in grass ley (Pedersen *et al.*, 2020b) showed that slurry application by trailing hoses 20 cm above the crop canopy (which is not uncommon) resulted in 40% higher NH_3 emissions compared with slurry application at the crop canopy.

Regarding the efficiency of acidification, a late peak in NH_3 emissions from acidified CS in the current project (Paper II) is interesting. It is hypothesised that for acidified slurries with a high DM content and hence a low infiltration rate, the buffering capacity of the slurry causes an increase in slurry pH before it has completely infiltrated. This would lead to a late peak in the NH_3 emissions days after slurry application, reducing the overall effect of the acidification. This would be of certain significance for the acidification of high DM biogas digestates, since they have both a high initial pH, a high buffering capacity, and slow infiltration. In fact, there is already an ongoing research project with NH_3 emission measurements related to digestate properties such as DM content, viscosity and infiltration rate in Denmark (Pedersen *et al.*, 2023).

For solid-liquid separation of slurry, studies have shown overall positive effects on NH_3 emissions and N fertilisation effect from the liquid fraction (Pedersen *et al.*, 2022). Use of the solid fraction as bedding material has shown no major effects on animal welfare or animal health compared with other bedding materials, however, the high levels of total bacteria in the solid fraction requires particular attention (Jeppsson *et al.*, 2024; Frondelius *et al.*, 2020). For slurry separation, an analysis of NH_3 and greenhouse gas emissions from both the solid and liquid fractions, including the whole slurry management chain with animal housing, storage, and application would be desirable. The analysis would include both comparisons of in-house emissions from solid fraction compared with other bedding materials, and different scenarios for storage of the solid fraction if not used as bedding material.

As seen both in the present study and in literature, it can be a challenge to achieve NH_3 abatement by slurry injection, especially on heavier soils. A prototype with the combination of trailing shoes and a rigid tine loosening the soil has shown promising results (McCollough *et al.*, 2022), however, only on a sandy soil. It is possible that the recommended NH_3 abatement measures need to be different based on the soil type.

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Popular science summary

The aim of this PhD project has been to investigate various ways to increase the nitrogen fertiliser value of cattle slurry. The nitrogen fertiliser effect from cattle slurry is relatively low, which is largely due to its high dry matter (DM) content and low proportion of readily available ammonium nitrogen. The high DM content implies slow slurry infiltration after application, and when the slurry remains on the soil surface it leads to large ammonia losses. This reduces the plant nutrient value of the slurry and leads to environmental problems such as eutrophication and acidification of lakes and streams. A high DM content also renders the slurry nitrogen less available to plants, which is related to the ratio between the amount of carbon and nitrogen, the C/N ratio, applied with the slurry. Readily available carbon (largely contained in the solid fraction of the slurry) acts as an energy source for soil microorganisms, and if large amounts of carbon are applied, e.g. in the form of high-DM slurries, the microorganisms also take up more of the nitrogen present, making it temporarily unavailable to plants. Processes removing some of the carbon-rich solid fraction of the slurry and/or measures to reduce ammonia losses can increase the value of the slurry while reducing its negative environmental impact.

Two processes reducing the DM content of cattle slurry and thus the C/N ratio (increasing the nitrogen availability) have been studied, namely mechanical separation of the manure into a liquid and a solid fraction and co-digestion of slurry together with other substrates for biogas production in a biogas plant. The risk of ammonia losses is reduced if the pH of the slurry is low, and if the surface area of the manure from which ammonia can evaporate is as small as possible. The measures to reduce ammonia losses during slurry application investigated in this study were slurry application with trailing shoes, creating soil slots with loosened soil to enhance slurry infiltration and reduce the slurry area, and acidification of the slurry by addition of sulphuric acid.

Measurement of ammonia losses after slurry application in grass ley resulted in highest losses from biogas digestate followed by untreated slurry, and lowest from separated slurry. Thus, a reduction in the slurry DM content, resulting in faster infiltration, does not always lead to lower ammonia losses. This is explained by the fact that the biogas digestate had a higher pH than the other slurry types, resulting in higher ammonia emissions immediately

after slurry application. Acidification with sulphuric acid proved to be effective and reduced ammonia losses by over 75 percent for all three slurry types. Slurry application with trailing shoes reduced ammonia losses from untreated slurry, but not from separated slurry and digestate.

The results of the fertilisation field experiments conducted in winter wheat and grass ley showed that the measures of separation, biogas digestion, and acidification of cattle slurry gave a similar increase in nitrogen fertiliser effect compared to fertilisation with untreated cattle slurry. For biogas digestate, acidification gave a further increase in nitrogen efficiency, but not for separated slurry. Slurry application with trailing shoes did not increase nitrogen fertiliser value of the slurries in winter wheat and grassland compared to trailing hose application. This was because the soil surface was often too dry and hard for the trailing shoes to perform optimally. On spring tilled soil, a positive effect of trailing shoes occurred in one of two fertilisation experiments in oats.

Overall, the study shows that there is potential to increase the nitrogen fertiliser value of cattle slurry, and that the greatest effect is achieved by a combination of anaerobic digestion and acidification of slurry. However, acidification of digestate is difficult because it has a strong buffering capacity, which causes problems with foaming and means that the amount of acid that must be added is large, making the profitability of the measure questionable. Sweden needs to reduce its ammonia emissions at the national level, and it is of great interest to continue evaluating various ammonia abatement options, without trade-offs in increased greenhouse gas emissions, adapted to Swedish climatic and soil conditions. Evaluations of different scenarios for the entire fertiliser management chain, both from an economic and environmental and climate perspective, would be desirable.

Populärvetenskaplig sammanfattning

Syftet med detta doktorandprojekt har varit att undersöka olika sätt att öka kvävegödselvärdet hos nötflytgödsel. Kvävegödslingseffekten från nötflytgödsel är relativt låg, vilket hänger samman med att den har en hög torrsubstanshalt (TS-halt) och låg andel lättillgängligt ammoniumkväve. Den höga TS-halten gör att gödseln infiltrerar långsamt vid gödselspridning, och när gödseln blir liggande kvar på markytan leder det till stora ammoniakförluster. Detta minskar gödselns växtnäringsvärde och leder också till miljöproblem som övergödning och försurning av sjöar och vattendrag. En hög TS-halt gör också kvävet i gödseln mindre tillgängligt för växterna, vilket hänger samman med förhållandet mellan mängden kol och kväve, den så kallade C/N-kvoten, som tillförs med gödseln. Lättomsättbart kol (som till stor del finns i den fasta delen av gödseln) fungerar som energikälla för mikroorganismer i marken, och tillför man mycket kol, t.ex. i form av gödsel med hög TS-halt, så tar mikroorganismerna även upp mycket av det kväve som finns, vilket gör det otillgängligt för växterna. Genom processer som tar bort en del av den kolrika fasta delen av gödseln och/eller åtgärder för att minska ammoniakförlusterna kan man öka gödselns värde och samtidigt minska dess negativa miljöpåverkan.

Två processer som sänker TS-halten och därmed C/N-kvoten (vilket ökar kvävets tillgänglighet) har studerats, nämligen separering av gödseln i en flytande och en fast fraktion samt rötning av gödsel tillsammans med andra substrat i en biogasanläggning. Risken för ammoniakförluster minskar vid ett lågt pH-värde hos gödseln, samt om gödselytan som ammoniaken kan avdunsta ifrån är så liten som möjligt. De åtgärder för att minska ammoniakförlusterna vid gödselspridning som undersökts i den här studien är spridning med släpskor, som skapar spår i marken med luckrad jord för att öka infiltrationen och minska gödselytan, samt surgörning av gödseln genom tillsats av svavelsyra.

Mätning av ammoniakförlusterna vid gödselspridning i vall visade högst förluster från biogasrötrest följt av obehandlad gödsel, och lägst från separerad gödsel. En minskning av TS-halten hos gödseln, och därmed snabbare infiltration, leder alltså inte alltid till lägre ammoniakförluster. Detta förklaras av att biogasrötresten hade ett högre pH än de andra gödselslagen, vilket ledde till högre ammoniakförluster direkt efter gödselspridning. Surgörning med svavelsyra visade sig vara effektivt och minskade ammoniakförlusterna med över 75 procent för alla tre gödseltyperna. Spridning av gödsel med släpskor minskade ammoniakförlusterna från obehandlad gödsel, men inte från separerad gödsel och rötrest.

Resultaten från de genomförda gödslingsförsöken i höstvete och vall visade att åtgärderna separering, biogasrötning respektive surgörning av nötflytgödsel gav en likartad ökning av kvävegödslingseffekten jämfört med gödsling med obehandlad nötflytgödsel. För biogasrötrest gav surgörning en ytterligare ökning av kvävegödslingseffekten, men inte för separerad gödsel. Spridning med släpskor ökade inte kvävegödslingseffekten från något av gödselslagen i höstvete och vall jämfört med släpslangsspridning, då markytan ofta var för torr och hård för att släpskorna skulle fungera optimalt. På vårbearbetad jord kunde dock en positiv effekt av släpskor ses i ett av två gödslingsförsök i havre.

Sammantaget visar studien att det finns potential att öka kvävegödselvärdet hos nötflytgödsel, och att den största effekten fås genom en kombination av rötning och surgörning av gödsel. Surgörning av rötrest är dock besvärligt eftersom den har en stark buffrande förmåga, vilket orsakar problem med skumbildning samt medför att mängden syra som behöver tillsättas är stor, vilket gör lönsamheten i åtgärden tveksam. Sverige behöver minska sina ammoniakutsläpp på nationell nivå, därför är det av stort intresse att fortsätta utvärdera olika alternativ för att minska dessa. Åtgärderna behöver vara anpassade efter svenska klimat- och jordartförhållanden, och effektivt minska ammoniakutsläppen utan att samtidigt öka utsläppen av växthusgaser. Önskvärt vore att utvärdera hela gödselhanteringskedjan för olika scenarier, både ur ett ekonomiskt och miljö- och klimatmässigt perspektiv.

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Bedding material properties and slurry C/N ratio affect the availability of nitrogen in cattle slurry applied to soil

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Introduction: Cattle slurry used as fertilizer in crop production is a mix of feces, urine, water, and bedding material from the housing system. Previous studies have shown that slurry nitrogen (N) availability to crops is dependent on C/N ratio. As the bedding material can contribute a significant part of total slurry carbon (C), its characteristics may affect the C/N ratio of the slurry. There is increasing interest in using the solid fraction from mechanical slurry separation as bedding material, and therefore this study investigated the potential effect of this fraction on slurry N availability, compared with more commonly used bedding materials such as straw and sawdust.

Methods: In two parallel 28-day laboratory incubations, net mineral N release and C mineralization from slurries applied to sandy loam soil were measured. The slurries comprised a liquid fraction (LF) from mechanical cattle slurry separation with a screw-press and different added bedding materials. Liquid fraction was mixed with two types of bedding material, solid slurry fraction (SF) and chopped straw, in different proportions, resulting in C/N ratios of 10, 12, and 14 in the slurry. In additional treatments, two other bedding materials, ground straw and sawdust, with slurry C/N ratio 12, were used.

Results: For SF and chopped straw, similar negative linear correlations were seen between slurry C/N ratio and net mineral N release after 28 days. Carbon mineralization, expressed as a percentage of total C added, was higher from the mixture containing SF than that containing straw, while no clear relationship with C/N ratio was found. At slurry C/N ratio 12, net release of mineral N was 28–39% of total N and decreased in the order: sawdust>chopped straw=SF=ground straw. Net C mineralization at the same slurry C/N ratio was 33–46% and decreased in the order: SF=ground straw>chopped straw>sawdust.

Discussion: For bedding materials with similar fiber composition (i.e., SF and straw), differences in C availability due to particle size or degree of degradation by microorganisms did not influence slurry N availability measurably. For sawdust, with high lignin content, the results indicate that limited C availability may lead to lower slurry N immobilization.

KEYWORDS

bedding material, solid fraction, straw, sawdust, slurry separation, soil incubation experiment

1 Introduction

Animal manure is an important source of nitrogen (N), a vital plant nutrient in crop production. However, only around 50% of the N in cattle slurry is present as easily available ammonium (NH_4^+), while the rest is in organic form (Webb et al., 2013). The fraction of slurry N available for plant uptake as NH_4^+ or NO_3^- is influenced by several factors, including animal type (Lukehurst et al., 2010), feed (Sørensen et al., 2003; Powell et al., 2006), pre-treatments such as solid-liquid separation (Fangueiro et al., 2015), anaerobic digestion (Möller and Müller, 2012), or acidification (Fangueiro et al., 2013), and slurry storage (Sørensen, 1998). The 1st-year availability of slurry N is negatively correlated with its carbon/nitrogen (C/N) ratio (Sørensen et al., 2003; Delin et al., 2012). Organic bedding materials are rich in C and poor in N, and may hence increase the C/N ratio in slurry. However, organic bedding materials differ in e.g., lignin content, particle size, and degree of degradation of organic matter, which may influence C availability and thereby mineralization/immobilization of N. Commonly used bedding materials include sawdust, sand, woodchips, straw, and peat (Ferraz et al., 2020; Frondelius et al., 2020). With pressurized filtration of slurry using a screw-press separator, a fibrous solid fraction suitable as bedding material can be produced (Hjorth et al., 2010). There is interest in using this solid fraction (SF), also called recycled manure solids (RMS), as bedding material in dairy housing systems (Husfeldt et al., 2012; Lybæk and Kjær, 2019; Leach et al., 2022).

In Sweden, free stall barns with cubicle housing is the most common housing system in dairy production (Växa Sverige, 2024), with concrete passageways cleaned with automatic scrapers, transporting the slurry to storage tanks outside the building. Storage tanks for slurry require coverage, either with a roof or a naturally formed cover crust, to reduce ammonia emissions (Swedish Board of Agriculture, 2021). Swedish standard figures state that the slurry production per dairy cow is 26–29 tons per year (Andersson et al., 2023) and the amount of bedding material needed is 400 kg (Agriwise, 2023), which implies ∼13–15 g bedding kg^{-1} slurry. Depending on the need for and use of the solid fraction, the amount of slurry that is separated may differ between farms with slurry separation. At the farm where the material for this study came from, untreated slurry collected from a pumping pit between the cow shed and the slurry separator had an average DM content of 8.5% which is close to the Swedish standard value of 9% (Andersson et al., 2023), while separated slurry collected from the storage tank had an average DM content of 5.3% ($n = 7$ over 3 years; Andersson and Delin, unpublished[1]). The C/N ratio of untreated and separated slurry was 10.7 and 6.4, respectively.

When SF is used as bedding material, partly degraded material in the form of animal feces is re-circulated in the system. With the most easily degradable C already degraded in the rumen, SF as bedding material could potentially reduce slurry N immobilization, due to a lower C availability than with other bedding materials, but knowledge in this area is limited. For slurry application in spring and summer, less immobilization could enhance crop N uptake and hence reduce the risk of N re-mineralization and nitrate leaching after harvest, since nitrate leaching is strongly dependent on whether there is a crop present to take up the nitrogen (Sørensen et al., 2023). Compared with untreated slurry, solid-liquid separation reduces ammonia emissions from field application of the liquid fraction, while the overall ammonia mitigation effect when including emissions from the solid fraction is more uncertain (Pedersen et al., 2022). Slurry separation can also reduce greenhouse gas emissions (Amon et al., 2006; Holly et al., 2017). Solid-liquid separation thus potentially has important environmental benefits, but a better understanding of the individual and overall effects is needed.

The aim of this study was to investigate if the type and properties of the bedding material added to cattle slurry affects the N and C turnover of the slurry within the 1st month after slurry application, and thus the short term potential of the slurry to supply N to a crop. The results could be used to improve advice to farmers about the 1st year fertilizer value they can expect from their slurry, so that they can adjust their crop fertilization rates accordingly. In all treatments, the liquid fraction from mechanical slurry separation (LF) was used, representing cattle slurry without bedding material. In a two-factor experiment with the factors bedding material and C/N ratio, chopped straw and SF was added in different amounts to obtain certain slurry C/N ratios, and the slurry mixtures were compared to determine whether the relationship between slurry C/N ratio and N availability differed between the two bedding materials. In additional treatments, aiming at the same slurry C/N ratio, addition of different bedding materials (chopped straw, ground straw, solid fraction, and sawdust) to LF was compared, in order to investigate their effect on slurry turnover rate and N availability. The hypotheses were that: (1) carbon dioxide (CO_2) production increases with increasing chemical and physical C availability of the bedding material to microbial utilization, where C availability was assumed to be lowest for sawdust (due to high lignin content), followed by SF (where easily degradable compounds in the consumed feed has been digested in the rumen, increasing the proportion of recalcitrant compounds in the SF), higher for chopped straw with large particle size, and highest for ground straw with small particle size; (2) net mineral N release is related to the C availability of the bedding material, with lower N immobilization, and thereby higher net N mineralization, from slurries with a lower content of easily available C; and (3) in chopped straw and SF bedding materials, there is a negative linear relationship between slurry C/N ratio and net mineral N release and the slope of this relationship is steeper for chopped straw, which is a more easily available C source than solid fraction.

2 Materials and methods

2.1 Soil, manure, and bedding materials

The soil used in laboratory incubation experiments was a sandy loam with 17% clay, 61% sand, 3% organic matter, and pH (1:5 H_2O) 6.7. It was collected from a field at SLU's experimental farm Götala close to Skara in western Sweden (58°22′N, 13°29′E). The crop rotation at the farm includes both 3–4 years grass/clover

[1] Andersson, K., and Delin, S. (unpublished). *Nitrogen Efficiency of Cattle Slurry Depending on Slurry Treatment and Application Method in Different Crops*.

forage leys and annual crops, with frequent addition of slurry and farmyard manure from beef cattle. Top soil was collected from a plowed field after a cereal crop in late autumn, ~2 months before experiment start and stored in open containers at cool temperature (0–5°C) until use. Two separate incubation experiments, one on N turnover and one on C turnover, were performed. These started on different dates (February 8 and 22, 2022, respectively), so soil preparation was performed using two separate batches, one for each experiment. The procedure used was the same for both batches, and started 10 and 13 days before the start of the N and C turnover incubation, respectively. To reduce the water content, soil was air-dried at 16–18°C for 2 days, with gentle mixing twice a day. The soil was sieved through a coarse (7 mm) mesh sieve and water-holding capacity (WHC) was determined using the method described in Jansson (1958). Water content after drying was 56 and 51% of WHC, respectively, for the two batches. Both soil batches were pre-incubated at 15°C in a climate chamber (Termaks, KB 6395, Nordic Labtech AB, Fjärås, Sweden), for 8 and 11 days, respectively.

Liquid (LF) and solid (SF) fractions from screw-press separated cattle slurry were collected from an organic dairy cattle farm (Otterslättens Lantbruk, Hova, Västra Götaland County, Sweden) 2 weeks before the start of the first experiment. Both fractions were collected directly from the slurry separator (CRI-MAN SM 260/75 FA DM, CRI-MAN, Correggio, Italy). The fractions were stored at cool temperature (0–5°C) until the start of the experiment.

Winter wheat straw was collected from a conventional farm. To obtain chopped straw, the straw was cut with scissors into ~0.5–1.0 cm pieces. To obtain ground straw, pieces of straw were milled in a grain mill to maximum particle size 0.8 mm. Sawdust (spruce and pine in unknown proportions) was collected from a commercial sawmill and sieved through a 2 mm mesh sieve.

2.2 Analyses of manure, bedding material, and soil samples

Samples of LF from slurry separation and the bedding materials used in the experimental slurries were analyzed for dry matter content (DM), total C, total N, and ammonium-N at the Soil and Plant Laboratory, Swedish University of Agricultural Sciences (SLU), Uppsala, Sweden. Total C and organic N were analyzed by combustion of dried samples followed by elemental analysis [928 Series Macro Determinator (model CN928), LECO, St. Joseph, MI, USA]. Ammonium-N was extracted with 2M potassium chloride (KCl) and analyzed using a flow injection analyzer [FIAstar Analyzer (5000), FOSS, Hillerød, Denmark]. Total N was calculated as the sum of organic N and ammonium-N.

After the incubations, an additional analysis of fiber fractions [neutral detergent fiber (NDF), acid detergent fiber (ADF), and acid detergent lignin (ADL)] was performed on stored samples of the original bedding materials, at the laboratory of the Department of Applied Animal Science and Welfare, SLU, Uppsala, using the method described by Van Soest et al. (1991). Acid detergent lignin was analyzed by the Klason lignin method, using sulfuric acid (72%). Hemicellulose content was calculated as the difference between NDF and ADF, and cellulose as the difference between ADF and ADL. As shown in Table 1, the concentrations of fiber fractions (hemicellulose, cellulose, and lignin) were similar in solid fraction and straw, while sawdust had a higher content of both cellulose and lignin, and higher total NDF content.

Soil samples from the N incubation were analyzed for ammonium-N and nitrate-N at the Soil and Plant Laboratory, SLU, Uppsala. Milled, frozen soil samples were extracted with 2M KCl and then analyzed by colorimetry on a Seal AA3 auto-analyzer (SEAL Analytical Inc., Mequon, Wisconsin, USA).

2.3 Treatments

In all treatments, the bedding materials were added to the LF from slurry separation, which was assumed to represent "pure" cattle manure without any bedding material added. This allowed differences in slurry properties related to the degree of slurry separation to be analyzed. Using LF (rather than unseparated slurry with higher DM content) as the basis for the treatments also resulted in greater variation in C/N ratio in the mixtures, without reaching unrealistically high C/N ratios in relation to that on real farms. The mixtures of LF and bedding materials are referred to hereafter as "slurry" or "slurries."

The amount of slurry added to the soil was based on supplying 50 kg total N ha^{-1} in all treatments, i.e., 26 tons ha^{-1} with LF, in this experimental setting with a soil depth of 50 mm corresponding to 102 mg kg^{-1} dry weight (DW) of soil. To obtain the different desired C/N ratios, the relative proportions of LF and bedding material were varied between the treatments. Straw and sawdust contributed very little N to the slurries, and in these treatments the proportion of bedding material was low and the amount of LF was more or less constant between C/N ratios. For SF, with higher ammonium N concentration, the amount of LF was reduced by at most 23%. Ammonium N content relative to total N in the slurries varied slightly between treatments and was on average 35% (Table 2). Due to the differences in N concentration, the amount of bedding material that needed to be added varied widely, between 12 and 127 g kg^{-1} LF, and there was also variation in the amount of total C applied to the soil (Table 2).

2.4 Incubation for analysis of mineral N

Small plastic flowerpots (height 50 mm, volume ~98 cm^3) were used as soil containers. On the day before experiment start, a filter paper was placed at the bottom of each pot to prevent soil loss through the drainage holes, and 70 g of pre-incubated soil were added and lightly packed down. The pots were placed in plastic boxes (36 x 24 x 13 cm), each containing pots for three replicates of one treatment and all sampling times, and placed in the same climate chamber where the soil was pre-incubated. The temperature was uniform in all parts of the chamber and was kept at 15 ± 0.3°C throughout the experiment. At the start of the experiment, one box at a time was removed from the climate chamber. The bedding material was added first to each pot and evenly distributed over the soil surface, and then LF was added. The separate weighing and addition of bedding material and LF was done to ensure that accurate C/N ratios were obtained. For ground

TABLE 1 Properties of the different bedding materials used in experimental slurries.

	Liquid fraction (LF)	Solid fraction (SF)	Straw	Sawdust
Number of samples (*n*)	2	1	1	1
Dry matter (%)	3.4	33.3	83.7	90[a]
Total C (g kg^{-1} fresh weight)	15.2	160	396	500[a]
Total N (g kg^{-1} fresh weight)	1.96	4.58	1.86	0.6[a]
C_{tot}/N_{tot} ratio	7.8	35	212	833
Ammonium N (g kg^{-1} fresh weight)	0.71	0.72	0	0
Total NDF (% of dry weight)	-	76.3	77.7	87.2
Hemicellulose (% of dry weight)	-	31.1	32.3	15.3
Cellulose (% of dry weight)	-	37.5	40.3	49.2
Lignin (% of dry weight)	-	7.6	5.1	22.7

Cells with "-" indicate that the parameter was not determined.
[a]From the literature (i.e., not analyzed).

straw and sawdust, the manure layer was stirred gently, taking care not to disturb the soil beneath, in order to create a slurry from the hydrophobic bedding material and the LF. Different amounts of water was added to obtain the same soil water content (70% of WHC) in all treatments and then another 45 g of pre-incubated soil were added to each pot, on top of the slurry, and gently packed down. The pots in their plastic box were then returned to the climate chamber. The reason for adding the experimental slurry as a layer in the middle, rather than thoroughly mixing it with the entire soil volume, was to mimic a real-life situation with animal slurry incorporated by plowing or injection, resulting in a more uneven distribution of slurry within the soil.

The lids of the plastic boxes were kept closed to prevent the soil from drying out. To ensure enough oxygen supply and avoid anaerobic conditions, the boxes were aerated at intervals of 3–4 days by opening the lid for a few minutes. On days 3, 7, 14, and 28, the three replicates of each treatment were destructively sampled for mineral N analysis. The entire soil volume from each replicate was placed in a plastic bag and frozen within 30 min for later analysis. To ensure that the water content remained constant during incubation, the boxes were weighed on each sampling occasion and, when necessary, deionized water was sprayed over the pots to replace the weight loss.

2.5 Incubation for analysis of CO_2 production

The procedure for preparing the soil and adding LF and bedding materials in incubation for analysis of CO_2 production was similar to that used in incubation for N analysis. However, the water content was kept slightly lower, at 65% of WHC, since the 70% used in the N incubation proved to be slightly too high, causing mold in some pots. Three replicates of each treatment were prepared. Each replicate was placed in a glass jar (outer diameter 120 mm, height 185 mm, volume 1,660 cm^3) with metal screw-lid, together with a CO_2 trap. The CO_2 trap consisted of a 50 mL plastic scintillation vial with 30 mL of 0.3 M KOH for the first measurements and 0.2 M KOH from day 10, absorbing the CO_2 produced in the jar. To ensure that all CO_2 produced was captured, the jar lids were kept closed, except when the traps were changed and CO_2 production measured. The KOH concentration was changed in order to permit the desired sampling frequency throughout the experiment, since the activity was expected to be lower during the later stage of incubation and a sufficiently high saturation level in the traps was needed to allow precise measurements. On days 2, 5, 10, 15, 21, and 28, the CO_2 traps were changed and electrical conductivity (EC) in the KOH solution was measured using a conductivity meter (ProfiLine Cond 3310 Portable Conductivity Meter, WTW, Weilheim, Germany). At each sampling occasion, the jars were aerated for a few minutes, thereafter a new CO_2 trap was placed in the jar, the lid was immediately closed and the jar was returned to the climate chamber. To correct for background CO_2 in the air and for basic soil respiration, the set-up included one blank treatment with only a CO_2 trap and one control treatment without any slurry added to the soil.

2.6 Calibrations for CO_2 determination

For calibration of the conductivity meter, solutions with different ratios of KOH and potassium carbonate (K_2CO_3) were prepared in order to imitate absorption of different amounts of CO_2, following the procedure described by Smirnova et al. (2014). Constant *A*, describing the theoretical maximum amount of CO_2 absorbed, was calculated for the 0.3 M and 0.2 M KOH solutions as:

$$N\,(CO_2) = A\,\frac{\mathbf{C\,(t_0)} - \mathbf{C\,(t_1)}}{\mathbf{C\,(t_0)}} \tag{1}$$

where $C(t_0)$ is conductance before any absorption of CO_2, $C(t_1)$ is conductance at time t_1 and *A* is empirically determined.

Temperature calibrations for the 0.3 M and 0.2 M KOH solutions at temperatures between 15 and 30°C revealed a negative linear correlation between conductivity and temperature. The resulting equations, obtained by fitting a straight line to the data points in Excel, were used for temperature correction in the conductivity calculations.

TABLE 2 Properties of the liquid fraction (LF) from separated slurry and of mixtures of LF and different bedding materials [solid fraction (SF), straw, and sawdust] used in experimental slurries.

Treatment code	C_{tot}/N_{tot} ratio	C_{tot}/N_{org} ratio	Bedding material added	Amount bedding added (g kg LF^{-1})	C from bedding (% of tot C)	N from bedding (% of tot N)	NH_4-N (% of tot N)	Tot C added to soil (mg kg^{-1} DW soil)
LF8	8	12.2	None	-	-	-	36.2	794
LF+SF10	10	15.3	SF	38	29	8	34.5	1,022
LF+SF12	12	17.9	SF	79	45	16	33.0	1,226
LF+SF14	14	20.4	SF	127	57	23	31.5	1,430
LF+straw10	10	15.6	Straw—chopped	12	23	1	35.8	1,022
LF+straw12	12	18.6	Straw—chopped	22	37	2	35.5	1,226
LF+straw14	14	21.6	Straw—chopped	33	46	3	35.1	1,430
LF+straw12-ground	12	18.6	Straw—ground	22	37	2	36.0	1,226
LF+sawdust12	12	18.8	Sawdust	17	36	1	36.7	1,220

2.7 Data treatment

2.7.1 Carbon

As a first step, a temperature correction to 20°C was made for the measured conductivity data, using the temperature calibration described above. The equations used for correction (obtained from Excel) were: $y = -0.261x + 69.181$ for the 0.3 M KOH solution and $y = -0.1774x + 47.142$ for the 0.2 M solution, where x is temperature (°C) and y is conductance (mS cm^{-1}). From the conductivity values obtained, CO_2 production for each sampling period was calculated using Equation (1), with constant *A* being 346.7 for 30 mL 0.3 M KOH and 235.2 for 30 mL 0.2 M KOH. Checks for outliers revealed that in a few cases one of the three replicates had considerably higher CO_2 production than the other two, indicating potential leaks between the glass jar and the lid. When the standard deviation within a treatment was more than 25% of the mean, the deviant value was omitted. This was the case for one of the blank samples on sampling days 2, 5, and 10.

For each sampling occasion, the average value of the blank samples was subtracted from all treatment values, to remove effects of background CO_2. To correct for basic soil respiration, the average value of the control treatment replicates with unfertilized soil was subtracted from the CO_2 values for the slurry treatments. As a last step, cumulative CO_2 production and amount of CO_2 produced per day, as a percentage of total added C, were calculated for each treatment. The amount of CO_2 absorbed by the traps was in the range 0.2–0.6 mg day^{-1} for the blank samples and 1.5–2.4 mg day^{-1} for the control treatment with unfertilized soil, with no clear pattern over time. For the fertilized treatments, CO_2 production (before correction for blank and control) decreased from 20.4–26.0 mg day^{-1} at the first sampling occasion to 3.6–6.2 mg day^{-1} after 28 days.

2.7.2 Nitrogen

The amount of mineral N supplied from slurry addition to soil is hereafter referred to as *Net mineral N release*, expressed as % of added total N. It gives a measure of potentially plant-available N from slurry, and was calculated according to Equation (2):

$$\text{Net mineral N release} = \frac{\text{Mineral N in soil with slurry} - \text{Mineral N in control soil}}{\text{Total N added with slurry}} \tag{2}$$

Net mineralization of slurry N over a given period (day 0-x), which describes the mineralization-immobilization processes, was calculated according to Equation (3):

$$\text{Net N mineralization} = \text{Net mineral N release (day x)} - \text{Net mineral N release (day 0)} \tag{3}$$

where net N mineralization is expressed as a percentage of total N added. It was also recalculated as a percentage of mineral N and organic N added.

2.7.3 Statistical analyses

For statistical analyses, the software Minitab (Version 21.3.1, Minitab18, Ltd., Coventry, UK) was used. To test the main effects and potential interactions of the factors *bedding material* (levels chopped straw and SF) and *C/N ratio* (levels 10, 12, and 14) on net mineral N release and CO_2 production, two-way ANOVA analyses were performed using the General Linear Model function. Response variables were mineral N content at day 3, 7, 14, and 28 and cumulative CO_2 production at day 2, 5, 10, 15, 21, and 28. One-way ANOVA (using the General Linear Model) was used to test for effects of different bedding materials at C/N ratio 12. To test whether a transformation of data was needed, results from the Box-cox transformation options "No transformation" and "Optimal λ" in Minitab were compared. Since data transformation in none of the cases significantly affected the results, the ANOVA and regression analyses on non-transformed data are presented. Pairwise differences were tested with a Tukey test at significance level 5%. To test whether the slope of the regression between net mineral N release and slurry C/N ratio at days 14 and 28 differed between SF and chopped straw, regression models were fitted, with *C/N ratio* as a continuous variable and *bedding material* as a categorical variable. An interaction for *C/N ratio*×*bedding material* implies significant differences in the regression coefficient (describing the slope of the regressions for the two bedding materials).

3 Results

3.1 Effect of C/N ratio and differences between chopped straw and SF

3.1.1 Net mineral N release

The two-way ANOVA revealed increased net mineral N release with decreasing slurry C/N ratio, except at day 3 (Supplementary Table 1, Figure 1A). The analyses showed no interaction between the factors C/N ratio and bedding material, except at day 7, when net mineral N release from slurry with straw was higher at C/N ratio 14 than at C/N ratio 12 (Supplementary Table 2). Initial net N immobilization occurred at all C/N ratios, but from day 3 the net mineral N release patterns diverged between C/N ratios (Figure 1A). For C/N ratio 8 and 10, initial immobilization changed to net N mineralization from day 3, while for C/N ratio 12 the immobilization phase lasted somewhat longer. Slurry with C/N ratio 14 showed net N immobilization throughout the incubation, while the net mineralization declined for slurry C/N ratio 10 and 12 toward the end of the experiment (Figure 1A). Averaged over C/N ratios 10–14, chopped straw as bedding material had higher net mineral N release (i.e., less N immobilization) than SF at day 7 and 14, while there was no difference after 28 days (Supplementary Table 1, Figure 1B).

By the end of the experiment, average net mineral N release from the experimental slurries amounted to 43, 36, 30, and 26% of total N added for C/N ratio 8, 10, 12, and 14, respectively (Supplementary Table 2, Figure 1A). This corresponded to net immobilization of 13 and 21% of added mineral N (i.e., 5 and 7% of added total N) for C/N ratio 12 and 14, respectively. At C/N ratio 10, the amount of soil mineral N was similar at the start and end of the experiment, while LF without any bedding material added and C/N ratio 8 resulted in net N mineralization, corresponding to 10% of added organic N (i.e., 6% of added total N).

There was a negative linear correlation between slurry C/N ratio and net mineral N release from the slurries with straw or SF addition after both 14 and 28 days (Supplementary Table 1, Figure 2). On day 14, the correlation was stronger for SF than for straw (Figure 2A), and there was a tendency for a steeper slope of the regression for SF ($p = 0.109$). By day 28, there were no longer any differences between SF and straw (Supplementary Table 1, Figure 2B).

3.1.2 CO_2 production

The two-way ANOVA for CO_2 production showed interaction between the factors C/N ratio and bedding material at days 2, 5, and 10, and a tendency for interaction at days 15, 21, and 28 (Supplementary Table 3). The rate of CO_2 production from both the SF and chopped straw slurries was highest during the 1st days, followed by considerably lower and slowly declining CO_2 production during the rest of the incubation (Figure 3). The decline in CO_2 production rate during the 2 last weeks of the experiment was most pronounced for C/N ratio 8 (Figure 3). Comparing C/N ratios, cumulative CO_2 production at the end of the experiment was lowest for C/N ratio 14, while no difference was seen between the other C/N ratios ($p = 0.002$; Supplementary Table 4). Comparing the two bedding materials, cumulative net CO_2 production by day 28 was higher for slurry with SF than with straw ($p < 0.001$), corresponding to 45 and 39% of total added C, respectively (Supplementary Table 4, Figure 3).

3.2 Effect of bedding material particle size and C availability

3.2.1 Net mineral N release

All four slurries at C/N ratio 12, with different bedding materials, showed initial net immobilization, followed by different patterns for the net mineral N release (Figure 4). Sawdust as bedding material resulted in higher net mineral N release than from the other bedding materials from day 7 (Supplementary Table 5, Figure 4). On day 28, net mineral N release was 39% of total N added for sawdust, compared with 28–30% for the other bedding materials (Supplementary Table 5). Net N immobilization corresponded to 14% of added mineral N for SF, 16% for chopped straw, and 23% for ground straw, while slight net N mineralization occurred for sawdust (corresponding to 3% of added organic N).

3.2.2 CO_2 production

In the comparison between treatments with four different bedding materials, all with C/N ratio 12, CO_2 production (% of C input) at day 2 was highest for SF, followed by ground straw and sawdust, and lowest for chopped straw ($p < 0.001$; Supplementary Table 6, Figure 5A). From day 5, SF and ground straw had, or tended to have, higher CO_2 production per day compared with chopped straw and sawdust (Figure 5A). Accordingly, cumulative CO_2 production at day 28 was highest for SF and ground straw (46% of added total C), followed by

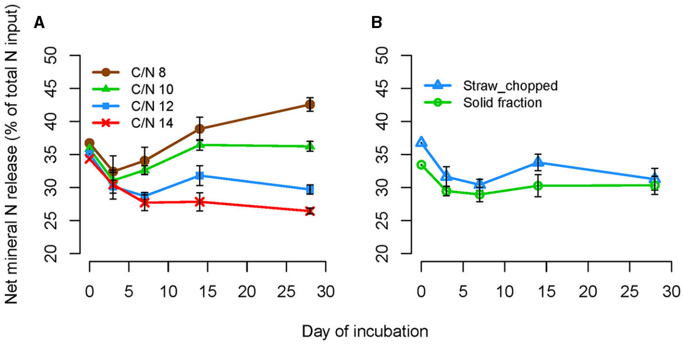

FIGURE 1
Effect of **(A)** C/N ratio and **(B)** bedding material type on net mineral N release from experimental slurries after 3, 7, 14, and 28 days of soil incubation. Values at day 0 are the calculated amounts added for each treatment. In **(A)**, the values for C/N ratio 10, 12, and 14 are mean values for the bedding materials solid fraction and chopped straw ($n = 6$). In **(B)**, the values for the bedding materials are mean values for C/N ratios 10, 12, and 14 ($n = 9$). C/N ratio 8 was not included in the two-way ANOVA, but is shown in the figure for comparison ($n = 3$). Error bars represent standard error of the mean (SEM). The two-way ANOVA showed a significant effect of C/N ratio ($p < 0.05$) for day 7, 14, and 28. For the factor bedding material, $p < 0.05$ at day 7 and 14. An interaction between the factors was seen at day 7 ($p < 0.05$).

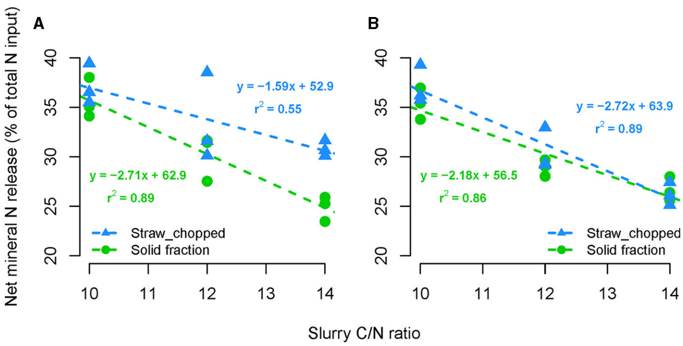

FIGURE 2
Relationship between C/N ratio and net mineral N release from experimental slurries with the bedding materials chopped straw and solid fraction after **(A)** 14 and **(B)** 28 days of incubation. Regression analysis showed a p-value < 0.001 for the factor C/N ratio at both day 14 and 28. For bedding material, the effect was non-significant ($p > 0.05$) both at day 14 and 28. For the interaction between C/N ratio and bedding material, the p-value was 0.109 at day 14 and 0.292 at day 28.

chopped straw (38%), and lowest for sawdust (33%; $p < 0.001$; Supplementary Table 7, Figure 5B).

4 Discussion

4.1 C and N turnover during the 1st days of incubation

The high initial CO_2 production rate in all treatments was most likely an effect of breakdown of short-chain volatile fatty acids (VFA) from LF, as they are an easily available C source that is quickly consumed during the 1st days of incubation (Kirchmann and Lundvall, 1993). The higher initial CO_2 production from slurries with lower C/N ratios (Supplementary Table 5, Figure 3, day 2) can be explained by the smaller amount of "extra" C added with bedding material in the slurry mixes at low C/N ratios (Table 2). Hence, those slurries contained a higher proportion of easily degradable C components, including VFAs.

Slurry with solid fraction had the highest initial CO_2 production of the different bedding materials at C/N ratio 12 (Supplementary Table 6, Figure 5A), probably as a result of an already established micro fauna, due to its previous contact with animal feces. Chopped straw had the lowest initial CO_2 production,

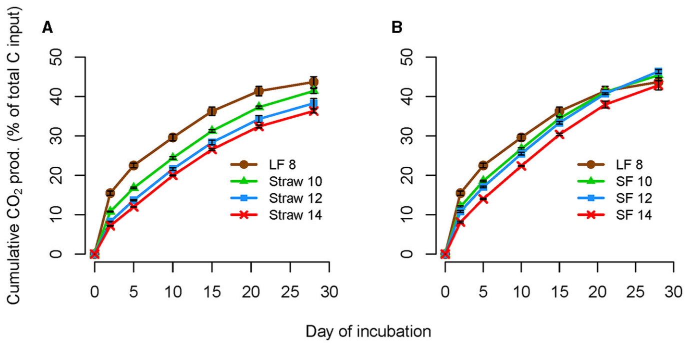

FIGURE 3
Cumulative net CO_2 production from experimental slurries at different C/N ratios and with the bedding materials **(A)** straw and **(B)** solid fraction during soil incubation ($n = 3$). C/N ratio 8 (without bedding material) was not included in the two-way ANOVA analysis, but is shown in the figure for comparison ($n = 3$). Error bars represent standard error of the mean (SEM). The two-way ANOVA showed significant main effects ($p < 0.05$) for both C/N ratio and bedding material at all sampling occasions. The interaction between C/N ratio and bedding material had p-values < 0.05 for day 2, 5, and 10.

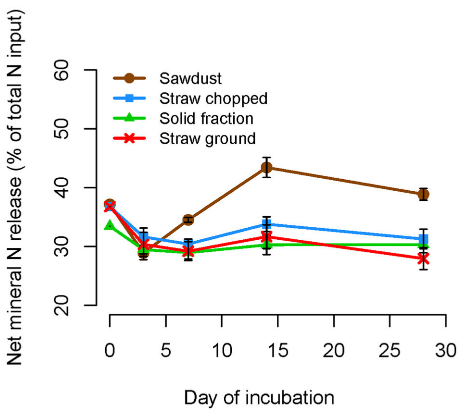

FIGURE 4
Net mineral N release during soil incubation from experimental slurries with different bedding materials, all with C/N ratio 12. Values at day 0 are the calculated amounts added for each treatment. Error bars represent standard error of the mean (SEM). The one-way ANOVA showed p-values < 0.05 for day 7, 14 and 28.

probably owing to its larger particle size and thereby smaller specific surface area compared with sawdust and ground straw. Sawdust had higher initial CO_2 production rate than chopped straw despite lower chemical C availability (more lignin), likely as a result of the smaller particle size making the most easily degradable C fractions accessible from the start (Sørensen et al., 1996).

Rapid initial N immobilization, similar to that in this study (Figures 1, 5), has been observed in a previous study which found a strong correlation between N immobilization and initial VFA content of slurries (Kirchmann and Lundvall, 1993). All treatments in the present study contained similar amounts of LF (the main source of VFAs) and showed similar initial N immobilization. This indicates that mineralization/immobilization processes during the 1st days after soil application were mainly driven by the properties of LF, while the added bedding materials had little influence.

4.2 Later C and N mineralization patterns

From day 5, the CO_2 production rate followed a slowly declining pattern that continued during the rest of the incubation. During this phase, the most easily available C sources had been depleted and the processes controlling C turnover seemed to be dominated by the properties of the bedding materials added. This was reflected e.g., in sawdust giving the lowest CO_2 production through having the highest lignin content, and in higher cumulative CO_2 production from ground straw than chopped straw. Higher CO_2 production from ground wheat straw compared with chopped straw during the 1st weeks after soil application was also observed by Angers and Recous (1997), and was hypothesized to be a result of higher N accessibility with finer particles. In studies with low C/N ratio amendments such as rye residue (Angers and Recous, 1997) and pea residue (Jensen, 1994), where N content in the residues was less limiting for decomposition, the opposite relationship was seen. This was attributed to more intimate association with soil mineral particles for the ground residues, leading to physical protection of C in microbial biomass and by-products. In the present study, CO_2 production rate (averaged over C/N ratio 10–14) was consistently higher from SF compared with chopped straw, despite similar fiber composition, which was likely a combined effect of differences in particle size and degradation by microorganisms, as discussed in the section above for initial CO_2 production. For nitrogen, the decline in net mineral N release at C/N ratios 8, 10, and 12 from

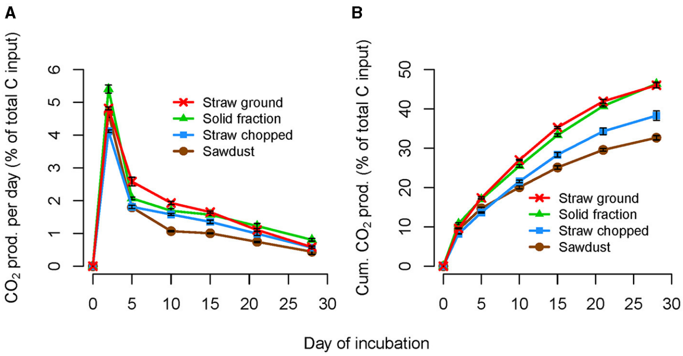

FIGURE 5
(A) Carbon dioxide production per day and **(B)** cumulative CO_2 production from soil incubation of experimental slurries with different bedding materials, all with C/N ratio 12. Error bars represent standard error of the mean (SEM). In both one-way ANOVA analyses, the *p*-values were <0.05 for all sampling occasions. Note different scales on Y-axis.

day 14 to day 28 (Figure 1A) could potentially be explained by re-immobilization of mineralized N and/or gaseous N losses through denitrification (not measured).

4.3 Nitrogen turnover and slurry C/N ratio

The negative linear relationship between C/N ratio and net mineral N release (Figure 2) is in line with hypothesis 3. The slope of the relationship was anticipated to be steeper with chopped straw than SF, based on the assumption that the straw contained more un-degraded and easily available C, resulting in greater N immobilization. This was not the case, and analysis of the fiber fraction revealed very similar content of hemicellulose, cellulose, and lignin in SF and straw (Table 1). The tendency for a steeper slope of the relationship for SF at day 14 (Figure 2A) is likely because microorganisms were already integrated with that material, whereas the straw had to be colonized, slowing down the straw degradation process and hence the N demand and immobilization. In addition, chopped straw had larger particle size, making the C less easy to access, which likely also delayed the degradation process (Sørensen et al., 1996).

A negative relationship between C/N ratio and N availability has been reported previously, for different organic fertilizers by Delin et al. (2012) and for cattle slurry by Sørensen et al. (2003). Sørensen et al. (2003) found that N availability was positively correlated with ammonium N content of the slurry, but in the present study the ammonium N content was within a narrow range (Table 2) and the relationship could not be evaluated. Norris et al. (2019) found that prediction of N supply from slurries could be improved by considering the ratio of total C to water-extractable N (rather than total N), but that considering more labile C fractions rather than total C weakened the predictions. This suggests that the amount of bioavailable N in organic material has a stronger influence on N mineralization/immobilization processes than amount of bioavailable C.

4.4 Bedding material particle size

The hypothesis that higher C availability due to smaller particle size would increase N immobilization was neither supported nor refuted. Ground straw showed higher cumulative CO_2 production than chopped straw (Figure 5) and a tendency for greater net N immobilization (Figure 4), but the difference in net mineral N release was not significant. However, on comparing ground and chopped barley residues added to soil, Ambus and Jensen (1997) found greater N immobilization with finer particle size after 2 weeks, and the difference remained throughout their 60-day incubation. Microbial growth was similar regardless of residue particle size in that study, leading to the conclusion that despite grinding, some C in the residues was still physically protected by lignin. In the present study, there were two different interacting sources of added C (LF and bedding material), which makes the results more difficult to interpret.

4.5 Relationship between C and N turnover

After 28 days, net mineral N release was similar for SF and chopped straw, despite differences in CO_2 production. In a study by Jensen et al. (2005), net N mineralization from a wide variety of mostly low-lignified plant materials was most closely correlated with concentrations of total plant N and neutral-detergent soluble N. In the same study, C mineralization was most closely correlated with holocellulose (hemicellulose + cellulose) content, which is in line with our observation that sawdust had both the lowest cumulative CO_2 production and the lowest holocellulose content. According to Chen et al. (2018), cellulose is rapidly degraded after soil incorporation of straw, while lignin is not broken down, but accumulates in the soil.

With regard to our three starting hypotheses, the results showed that: (1) C availability (as indicated by cumulative CO_2 production) depended on a combination of bedding material particle size, fiber composition, and probably previous degradation by microorganisms; (2) for bedding materials with a relatively low degree of lignification (i.e., straw and solid fraction) there was no relationship between C and N availability, while for sawdust, with higher lignin content, high net mineral N release was associated with low CO_2 production; and (3) for both SF and chopped straw, there was a negative linear relationship between slurry C/N ratio and net mineral N release, with similar slope for both bedding materials. The contrasting results for sawdust, with substantially higher net mineral N release than from the other bedding materials, show that in addition to slurry C/N ratio, bedding material fiber composition is an important factor affecting slurry N availability.

4.6 Practical and environmental implications

The results indicate that bedding materials with similar fiber composition (i.e., SF and straw) give slurries with similar short term fertilizer value, when applied in amounts resulting in the same slurry C/N ratio. Applied at farm scale, this means that since SF has a lower C/N ratio than straw (Table 1), addition of similar amounts of SF as with straw would result in a lower slurry C/N ratio and hence potentially in higher N availability. The results also indicate that sawdust as bedding material could result in higher N availability, provided that the amount used gives the same slurry C/N ratio as with other bedding materials. This study confirms the relationship between C/N ratio and N availability, suggesting that slurry C/N ratio can be used as a predictor for N use efficiency in crop production (Delin et al., 2012). The results indicate that a higher N use efficiency can be expected from slurry containing sawdust, compared with SF and straw.

The environmental effects of slurry separation are more complex and difficult to predict. For the liquid fraction, the lower DM content increases the infiltration rate and hence reduces ammonia emissions after slurry application (Pedersen et al., 2022). When applied in spring or summer, the lower C/N ratio and thereby increased N availability can also improve the synchronization between N supply and crop N uptake. This potentially reduces the risk of mineral N surplus in autumn and subsequent losses during winter, when crop N uptake is low or no crop is present to take up the nitrogen (Delin and Stenberg, 2021; Sørensen et al., 2023). Simultaneous application of easily available carbon and nitrogen, as with all animal slurries, may stimulate denitrification and increase the risk of nitrous oxide emissions (Senbayram et al., 2012). For soil application of the liquid fraction from separated slurries and digestates, studies have shown both similar (Holly et al., 2017) and higher (Fangueiro et al., 2008b; Meng et al., 2022) nitrous oxide emissions, compared with raw slurries.

With covered storage of the liquid fraction, reducing the ammonia emissions, the overall environmental effect of slurry separation is largely dependent on how the solid fraction is treated during storage. With composting of the solid fraction, the total ammonia emissions are increased, while greenhouse gas emissions are reduced (Amon et al., 2006; Fangueiro et al., 2008a). Storage of the solid fraction without adding oxygen by turning, could either increase (Holly et al., 2017) or reduce (Hansen et al., 2006) the emissions of nitrous oxide. A system where the solid fraction is used as bedding material is probably most comparable with composting, with good oxygen supply and hence increased ammonia emissions. However, an evaluation of that system would need to include a comparison of the in-house ammonia emissions with other types of bedding materials.

5 Conclusions

There was a negative linear relationship between slurry C/N ratio and net mineral N release after 28 days, with similar slope of the regression line for the bedding materials straw and solid fraction. Differences in C availability between these two bedding materials, as indicated by cumulative CO_2 production, did not affect N availability. The particle size of straw used as bedding material also had no effect. However, the relationship for sawdust was different, with higher net mineral N release than from SF and straw, in combination with lower CO_2 production, indicating limited C availability due to high lignin content. Therefore, if farmers use C/N ratio as an indicator of short term N availability in slurry, the same relationship can be used for calculation regardless of whether straw or SF is used as bedding material and regardless of particle size of the straw. If the same amount of bedding material is added, SF would result in a lower slurry C/N ratio, and hence potentially in higher N availability. If sawdust is used, N availability may be 24–39% higher at similar C/N ratio.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

KA: Conceptualization, Formal analysis, Investigation, Resources, Visualization, Writing – original draft, Writing – review & editing. AD: Formal analysis, Resources, Writing – review & editing. PS: Conceptualization, Writing – review & editing. SD: Conceptualization, Investigation, Resources, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2024.1393674/full#supplementary-material

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Bedding material properties and slurry C/N ratio affect the availability of nitrogen in cattle slurry applied to soil

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Supplementary material

Table S1. Results from the statistical analyses on net mineral N release. The response variable in all analyses is net mineral N release (% of total N input).

Two-way ANOVA: C/N ratio (10, 12, 14), bedding material (SF, straw), C/N ratio×bedding material			
Day	Main effects and interactions (p-value)		
	cn	bedding	cn×bedding
3	0.915	0.259	0.495
7	<0.001	0.019	0.001
14	<0.001	0.010	0.313
28	<0.001	0.220	0.427
Regression: C/N ratio (continous predictor), bedding material (categorical predictor)			
Day	Main effects and interactions (p-value)		
	cn	bedding	cn×bedding
14	<0.001	0.231	0.109
28	<0.001	0.292	0.292
One-way ANOVA: bedding materials at C/N ratio 12			
Day	p-value		
3	0.493		
7	0.003		
14	0.006		
28	0.001		

Table S2. Net mineral N release (% of total N input). Values within the same column and the same comparison (C/N ratio, Bedding material and Bedding material×C/N ratio) sharing the same lowercase letter is not significantly different at p=0.05 level (Tukey's test).

	Net mineral N release (% of total N input)			
C/N ratio	**Day 3**	**Day 7**	**Day 14**	**Day 28**
10	31.1 a	32.4 a	36.4 a	36.2 a
12	30.1 a	28.6 b	31.5 b	29.6 b
14	30.4 a	27.1 b	27.5 c	26.4 c
Bedding material				
Solid fraction	29.5 a	28.1 b	29.5 b	30.0 a
Straw	31.6 a	30.0 a	33.4 a	30.7 a
C/N ratio×bedding material				
LF+SF10	31.5 a	31.8 ab	35.7 ab	35.3 a
LF+SF12	28.7 a	29.7 bc	30.1 bc	28.9 bc
LF+SF14	28.2 a	25.1 d	24.8 c	26.6 bc
LF+straw10	30.6 a	33.2 a	37.1 a	37.0 a
LF+straw12	31.6 a	27.7 cd	33.0 ab	30.4 b
LF+straw14	32.7 a	30.3 abc	30.8 abc	26.2 c
LF8[a]	32.4	33.4	38.7	42.5

[a]Since LF8 was not included in the two-way ANOVA, the treatment is not included in the comparisons and hence has no grouping letters.

Table S3. Result from the statistical analyses on CO_2 production. Response variable in in the two-way ANOVA is cumulative CO_2 production (% of total C input).

Two-way ANOVA: C/N ratio (10, 12, 14), bedding material (SF, straw), C/N ratio×bedding material			
Day	Main effects and interactions (p-value)		
	cn	bedding	cn×bedding
2	<0.001	<0.001	0.001
5	<0.001	<0.001	0.002
10	<0.001	<0.001	0.022
15	<0.001	<0.001	0.087
21	<0.001	<0.001	0.118
28	0.002	<0.001	0.082
One-way ANOVA: bedding materials at C/N ratio 12 CO_2 production per day (% of total C imput)			
Day	p-value		
2	<0.001		
5	<0.001		
10	<0.001		
15	<0.001		
21	0.001		
28	0.009		

One-way ANOVA: bedding materials at C/N ratio 12 Cumulative CO_2 production (% of total C input)			
Day	p-value		
2	<0.001		
5	<0.001		
10	<0.001		
15	<0.001		
21	<0.001		
28	<0.001		

Table S4. Cumulative CO_2 production (% of total C input). Values within the same column and the same comparison (C/N ratio, Bedding material and Bedding material×C/N ratio) sharing the same lowercase letter is not significantly different at p=0.05 level (Tukey's test).

	Cumulative CO_2 production (% of total C input)					
C/N ratio	**Day 2**	**Day 5**	**Day 10**	**Day 15**	**Day 21**	**Day 28**
10	11.4 a	17.6 a	25.5 a	32.9 a	39.3 a	43.4 a
12	9.4 b	15.2 b	23.3 b	30.8 b	37.6 b	42.3 a
14	7.6 c	12.9 c	21.1 c	28.4 c	35.3 c	39.6 b
Bedding material						
SF	10.2 a	16.3 a	24.64 a	32.7 a	40.0 a	44.9 a
straw	8.6 b	13.9 b	21.7 b	28.7 b	34.7 b	38.7 b
Bedding material×C/N ratio						
SF 10	12.0 a	18.6 a	26.7 a	34.4 a	41.1 a	45.4 a
SF 12	10.8 b	17.0 b	25.5 ab	33.3 a	40.7 a	46.4 a
SF 14	8.1 c	14.0 c	22.4 c	30.4 b	38.0 b	42.8 ab
Straw 10	10.8 b	16.8 b	24.4 b	12.5 b	37.3 b	41.4 bc
Straw 12	8.2 c	13.7 c	21.6 c	28.3 c	34.3 c	38.3 cd
Straw 14	7.2 d	12.0 d	19.9 d	26.6 d	32.4 c	36.3 d
LF 8[a]	15.4	22.5	29.5	36.2	41.4	43.7

[a]Since LF8 was not included in the two-way ANOVA, the treatment is not included in the comparisons and hence has no grouping letters.

Table S5. Net mineral N release from different bedding materials, all with C/N ratio 12. Within each column, values sharing the same lowercase letter are not significantly different at p=0.05 level (Tukey's test).

	Net mineral N release (% of total N input)			
Bedding material	**Day 3**	**Day 7**	**Day 14**	**Day 28**
Sawdust	28.7 a	34.5 a	43.4 a	38.9 a
SF	28.7 a	29.7 b	30.2 b	28.9 b
Straw chopped	31.2 a	27.7 b	33.3 b	30.5 b
Straw ground	29.8 a	29.2 b	31.6 b	27.9 b

Table S6. Carbon dioxide production per day (% of total C input) from different bedding materials, all with C/N ratio 12. Within each column, values sharing the same lowercase letter are not significantly different at p=0.05 level (Tukey's test).

Treatment	**Day 2**	**Day 5**	**Day 10**	**Day 15**	**Day 21**	**Day 28**
Sawdust	4.67 b	1.80 b	1.07 c	1.01 c	0.74 c	0.44 b
Solid fraction	5.41 a	2.07 b	1.69 b	1.57 a	1.23 a	0.81 a
Straw chopped	4.12 c	1.81 b	1.58 b	1.35 b	0.99 b	0.57 ab
Straw ground	4.81 b	2.58 a	1.93 a	1.65 a	1.11 ab	0.58 ab

Table S7. Cumulative CO_2 production (% of total C input) from different bedding materials. all with C/N ratio 12. Within each column, values sharing the same lowercase letter are not significantly different at p=0.05 level (Tukey's test).

Treatment	**Day 2**	**Day 5**	**Day 10**	**Day 15**	**Day 21**	**Day 28**
Solid fraction	10.8 a	17.0 a	25.5 b	33.3 b	40.7 a	46.4 a
Straw ground	9.6 b	17.4 a	27.0 a	35.3 a	41.9 a	46.0 a
Straw chopped	8.2 c	13.7 b	21.6 c	28.3 c	34.3 b	38.3 b
Sawdust	9.3 b	14.7 b	20.1 d	25.1 d	29.6 c	32.7 c

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Research Paper

Ammonia emissions from untreated, separated and digested cattle slurry – Effects of slurry type and application strategy on a Swedish clay soil

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Animal slurry contains plant nutrients such as nitrogen (N) that are essential for crop production. Inorganic slurry N is easily volatilised as ammonia after field application, reducing slurry fertiliser value and causing environmental problems. Ammonia emissions can be reduced by lowering slurry pH, rapid infiltration or incorporation of slurry into soil. This study investigated the effect of different combinations of slurry types and application strategies on ammonia emissions. The slurry types tested were untreated cattle slurry (CS), the liquid fraction from mechanical solid–liquid separation of cattle slurry (LF) and biogas digestate based mainly on cattle slurry (BD). The application strategies tested were trailing hoses, trailing shoes, trailing hose application of acidified slurry and slurry injection. Ammonia emissions after slurry application were measured using wind tunnels, with continuous measurements of ammonia concentrations in outgoing air. Comparisons were also made between measured ammonia emissions and emissions predicted by the ALFAM2 model. Cumulative ammonia emissions after 70 h from LF, CS and BD represented 23%, 29% and 32% of total ammoniacal nitrogen (TAN) applied. Trailing shoes and 50 mm deep injection slots reduced ammonia emissions by on average 17% and 37%, respectively, compared with trailing hoses. Slurry acidification resulted in an average reduction in ammonia emissions of 83%. The ALFAM2 model was reasonably accurate in predicting cumulative emissions (70 h). Accuracy in predicting emission dynamics was low in some cases, likely due to differences between wind tunnel measurements and open-air emissions and to model error.

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Nomenclature

A	soil area covered by the wind tunnel (m^2)
ANOVA	analysis of variance
BD	biogas digestate
C	ammonia concentration ($g\ N\ L^{-1}$)
c_i	inlet (background) concentration or mixing ratio (ppb_v)
c_o	outlet concentration or mixing ratio (ppb_v)
CRDS	cavity ring-down spectrometer
CS	untreated cattle slurry
DM	dry matter
F_{NH3}	ammonia flux ($g\ N\ m^{-2}$)
IHF	integrated horizontal flux
LF	liquid fraction from solid–liquid separation of cattle slurry
M	molar mass of nitrogen (14.01 $g\ mol^{-1}$)
N	nitrogen
NH_3	ammonia (gas)
NH_3 (aq)	free ammonia in a liquid
NH_4^+	ammonium
P	total pressure (1 atm)
q	volumetric airflow rate ($L\ min^{-1}$)
R	gas constant ($L\ atm\ K^{-1}\ mol^{-1}$)
SDG	sustainable development goals
T	temperature (K)
TAN	total ammoniacal nitrogen

1. Introduction

1.1. Background

Animal manure from livestock production is an important source of nitrogen (N) and other plant nutrients for crop production worldwide (Warren Raffa, Turner, Tubiello, & Serrano, 2018). However, besides being a valuable fertiliser, manure is also an important source of emissions of ammonia (NH_3) and greenhouse gases (Oenema, Bannink, Sommer, Van Groenigen, & Velthof, 2008). Ammonia losses result in lower and more uncertain fertiliser value of the manure when applied to crops. The ammonia emissions from manure are also a health risk for humans and animals, and a cause of environmental problems such as eutrophication and acidification of soils and water (Anderson, Strader, & Davidson, 2003; Webb et al., 2013). In the European Union (EU) and UK, more than 4.8 million tonnes of N are applied to agricultural soils annually, making up about 20% of N input (Misselbrook & Bittman, 2022). In Sweden, one-third of the land area used for crop production receives animal manure, with 84% of the total manure volume applied in liquid form, as slurry (Statistics Sweden, 2020). The most common application technique is trailing hoses, applying 75% of the total slurry volume, followed by broadcast application and slurry injection at 20% and 5%, respectively. With over 56,000 ha in Sweden fertilised with animal slurry yearly (Statistics Sweden, 2020), increased use of ammonia abatement techniques would have positive effects on the environment and on nitrogen use efficiency in crop production.

Ammonia emissions from the agricultural sector and potential subsequent processes, including conversion to nitrate and nitrous oxide, are relevant to several of the United Nations Sustainable Development Goals (SDG), including those regarding human health (SDG 3), clean water (SDG 6), climate change, and life in water and on land (SDG 11–15) (Kanter, Zhang, & Howard, 2016). Ammonia is also one of five pollutants regulated under the EU National Emission Reduction Commitments Directive, with the target of reducing the threat to ecosystem biodiversity and premature deaths caused by air pollutants (European Enviroment Agency, 2022). Sixteen of the 27 EU member states met their national targets for ammonia reductions in 2020, while the others, including Sweden, need to reduce their emissions further to meet the targets for the period 2020–2029. For these reasons, various approaches are being used to reduce NH_3 losses from field-applied slurry (Misselbrook & Bittman, 2022).

1.2. Ammonia emissions and emission abatement

Ammonia emissions after field application of slurry depend on a number of factors, including soil and slurry properties, application technique and weather conditions such as temperature, solar radiation, wind speed and precipitation (Sommer et al., 2003; Sommer & Hutchings, 2001). The emission rate is highest immediately after slurry application, with 50% of total ammonia losses usually occurring within the first 12 h (Hafner et al., 2018).

Ammonium (NH_4^+) and free ammonia (NH_3 (aq)) are an acid–base-pair, with the chemical equilibrium between NH_4^+ and volatile NH_3 (aq) depending on manure pH. Therefore ammonia emissions are strongly pH-dependent, and slurry acidification can be used for ammonia abatement in animal housing systems, during slurry storage and after field application. The most common additives used to lower the pH are strong acids, e.g. sulphuric acid (Fangueiro, Hjorth, & Gioelli, 2015). In earlier studies testing different acids and with different target pH values, reported ammonia abatement after land application of acidified cattle slurry has been within the range 15–80% (Fangueiro, Hjorth, et al., 2015). In more recent studies, acidification of cattle slurry with sulphuric acid to pH 6.0 or lower has been found to reduce ammonia emissions by on average around 80% (Fangueiro et al., 2017; Pedersen, Feilberg, & Nyord, 2022; Seidel et al., 2017).

Ammonia abatement techniques for slurry application have been investigated and reviewed in a number of publications over the years (Häni, Sintermann, Kupper, Jocher, & Neftel, 2016; Misselbrook, Smith, Johnson, & Pain, 2002; Nicholson, Bhogal, Rollett, Taylor, & Williams, 2018; Seidel et al., 2017; Webb, Pain, Bittman, & Morgan, 2010), with the most common reference technique being broadcast application. The abatement effect achieved by trailing shoes and open-slot injection is generally larger than that from trailing hoses, but the measured effect varies widely for each technique (Webb et al., 2010). In a recent study with trailing hoses as the reference technique, the ammonia abatement from trailing shoes was on average 20%, with the greatest effects seen on more coarse-textured soil (Pedersen, Feilberg, Kamp, Hafner, & Nyord, 2020). Slurry separation, decreasing the slurry dry matter (DM) content of the liquid fraction, has the potential to significantly reduce ammonia losses after land application of the liquid fraction (Amon, Kryvoruchko, Amon, & Zechmeister-Boltenstern, 2006; Nyord, Hansen, & Birkmose, 2012), presumably as a result of increased slurry infiltration. Studies investigating the effect of slurry injection have reported variable results, with the ammonia abatement effects strongly related to the performance of injectors and the volume of injection slots (Hansen, Sommer, & Madsen, 2003; Rodhe & Etana, 2005).

1.3. *Measuring ammonia emissions*

The techniques available for measuring ammonia emissions in field experiments can be divided into two main categories: micrometeorological methods, requiring large experimental areas, and enclosure methods using smaller experimental plots (Misselbrook, Nicholson, Chambers, & Johnson, 2005). Micrometeorological methods give reliable measurements in terms of absolute ammonia emissions under ambient conditions. Wind tunnels generally use constant airflow, and mass transfer properties differ from open-air conditions and depend on wind tunnel design (Saha, Wu, Zhang, & Bjerg, 2011; Scotto di Perta et al., 2019), and therefore wind tunnel measurements may under- or overestimate emissions (Sommer & Misselbrook, 2016). A new technique combining wind tunnels with continuous online measurements of ammonia concentrations has recently been developed (Pedersen et al., 2020). It provides high-time resolution data, making it possible to monitor emission patterns closely. In addition, the low measurement variability makes it a suitable technique for comparing different low-emission application strategies.

1.4. *Modelling ammonia emissions*

Models for prediction of ammonia emissions can be used for different purposes, e.g. evaluation of mitigation strategies, as a basis for regulations related to field application of animal manure, or for national emissions inventories (Hafner et al., 2019). Several models have been developed during recent decades, some mainly process-based and others of a more empirical nature, with different degrees of complexity and different input requirements (e.g. Congreves et al. (2016); Genermont and Cellier (1997); Nicholson et al. (2013)). Empirical models have the advantage that they are usually less complex, need relatively few input variables and are easy to use. Process-based models, on the other hand, may be more accurate under new conditions and may be better at predicting complex responses (Hafner et al., 2019).

1.5. *Aim and hypotheses*

The aim of this study was to quantify interactive effects between treatments reducing slurry DM content and low-emission slurry application strategies. Ammonia emissions were measured in field experiments with slurry application to grass ley, on a heavy clay soil with 43% clay content. The slurry types used were untreated cattle slurry (CS), the liquid fraction from screw-press separated cattle slurry (LF) and biogas digestate based mainly on cattle slurry (BD). The application strategies used were trailing hoses, trailing shoes, acidified slurry applied by trailing hose and slurry injection (see Fig. S1 for more information on application methods). The system used, combining wind tunnels with online ammonia measurements, yielded values with high time resolution and low variability, thereby providing accurate detailed data on the ammonia flux patterns throughout the experiments. Flux data from the field experiments were compared against emissions predicted by the ALFAM2 model (Hafner & Haeni, 2022), in order to explore the applicability of the model to wind tunnel measurements and to evaluate its performance and possibly identify areas of interest for further model development. This is especially relevant for acidified slurry and digested slurry, for which the underlying data in the ALFAM2 database are limited.

The hypotheses tested were that ammonia emissions after land application of cattle slurry can be reduced by 1) lowering the slurry DM content by solid–liquid separation or biogas digestion, thereby increasing infiltration, and 2) lowering the slurry pH by slurry acidification. Two further hypotheses tested were that ammonia emissions can be reduced by 3) reducing the slurry area after application and aiding slurry infiltration by trailing shoe application, and 4) further increasing the slurry-soil contact and lowering the exposed surface area by slurry injection. Differences in the effect depending on how these measures were combined were also investigated. The hypotheses in that context were that 5) slurry acidification has the largest effect on digestate, due to higher initial pH and 6) there is no difference in effect between different slurry types when using trailing shoes.

2. Materials and methods

2.1. *Ammonia emissions experiments*

In total, five ammonia emissions experiments were conducted at Lanna field research station in south-west Sweden (58°20′N, 13°7′E) during June and early July 2019. The experiments lasted from 70 h to 120 h, with the shortest time (i.e. 70 h) being selected for comparison of cumulative ammonia emissions. A randomised block design was used in all experiments. In experiments 1–3, application with trailing hoses was compared with application with trailing shoes (Zunhammer GmbH, Traunreut, Germany), together with slurry acidification (with trailing hose application) for each of the three slurry types. Experiment 4 involved comparison of the three slurry types, all applied with trailing hoses. In experiment 5, separated slurry (LF) was used and trailing hose application was compared with simulated slurry injection at two depths, 20 mm and 50 mm (Injection1 and Injection2, chosen to reflect poor and optimal performance of slurry injection, respectively). For both Injection1 and Injection2, a pickaxe (Fig. S1) was used to break the soil surface crust and create the slots, resulting in a 40–50 mm wide band with loosened soil in the uppermost soil layer.

2.1.1. *Soil, crop and weather conditions*

All experiments were conducted in the same field, on a silty clay soil with 43% clay, 14% sand, 40% silt, 2.6% organic matter and pH 7.0 (1:5 soil:H_2O). Clay content was analysed using a sedimentation method modified from Gee and Bauder (1986), the sand fraction was determined by sieving and the residual sample mass was assumed to be the silt fraction. Preceding crop was winter wheat in both 2018 and 2017. In experiments 1–4, the crop was a forage grass ley consisting of 40% timothy (*Phleum pratense* L.), 30% meadow fescue (*Festuca pratensis* L.) and 30% perennial ryegrass (*Lolium perenne* L.), sown in September 2018 and with a first cut on 27 May 2019, before experiment start. Experiment 5 was conducted in an adjacent part of the same field, in spring barley stubble. The barley (flowering completed, crop still green) was cut with a grass plot harvester three days before experiment start and removed from the field. It was expected that spring cultivation would result in a less compact uppermost soil layer and thereby more optimal conditions for ammonia emissions abatement from slurry injection. Soil water content and soil dry bulk density in the top 50 mm were determined by collection of soil cores (n = 4) at 0–50 mm depth in grass ley before the start of experiment 1 and in barley stubble after the end of experiment 5. The soil was first dried at 55 °C for >48 h, followed by 6 h at 105 °C. Soil dry bulk density was 1.05 g mm^{-3} (sd = 0.06) in grass ley and 1.07 g mm^{-3} (sd = 0.02) in barley stubble. Soil water content was 0.21 g g^{-1} (sd = 0.015) at the start of the experimental period and 0.15 g g^{-1} (sd = 0.016) at the end. Weather data were obtained from a weather station at the experimental site. Start times and temperature data for the different experiments are summarised in Table 1.

2.1.2. *Slurry types and application*

The untreated (CS) and separated (LF) slurry were obtained from the same commercial organic dairy farm. The untreated slurry was collected from a pit outside the cowshed, from where slurry is pumped to a screw press separator (CRI-MAN SM 260/75 FA DM, CRI-MAN, Correggio, Italy), removing approximately 50% of the DM, creating a solid fraction used as bedding material for the cows and a liquid fraction that is stored in a large storage tank until field application. The separated slurry for the experiments was collected from the storage tank. The biogas digestate (BD) came from a biogas plant processing cattle slurry from about 20 different farms, together with other substrates such as pig slurry and waste products from slaughterhouses and the food industry. The amount of cattle slurry in the substrate mix was about 65% (fresh mass basis). All slurry types were collected at the beginning of April and stored in 1000-L plastic tanks in a barn until experiment start.

Shortly after slurry collection, one representative sample of each slurry type was sent for laboratory analysis, the results of which are shown in Table S1. Slurries were analysed for total ammoniacal nitrogen (TAN), total carbon (C), nitrogen (N), calcium (Ca), potassium (K), magnesium (Mg), sodium (Na), phosphorus (P), sulphur (S) and pH. TAN was extracted with 2 M KCl according to modified Swedish standard ISO 11732 (International Organization for Standardization [ISO], 1995) and analysed using a flow injection analyser (FIAstar Analyzer (5000), FOSS, Hilleroed, Denmark). Total C and N were analysed according to modified standards ISO 10694 (ISO, 2005) and ISO 13878 (ISO, 1998), respectively, with combustion of dried slurry samples followed by elemental analysis (928 Series Macro Determinator (model CN928), LECO, St. Joseph, MI, USA). Total Ca, K, Mg, Na, P and S were determined according to modified Swedish standard SS 028311 (ISO, 2017) by extraction with 7 M HNO_3, followed by analysis of the elements using an ICP spectrometer (SPECTROBLUE ICP-OES, SPECTRO Analytical Instruments GmbH, Kleve, Germany). At the laboratory, slurry pH was measured at room temperature with a glass pH electrode (Jenway™ 924005 pH Temperature Electrode, Cole-Parmer, Stone, Staffordshire, UK). Slurry pH at acidification and field application was measured at ambient temperature with a pH-meter (MW102, Milwaukee Instruments Kft., Szeged, Hungary).

The TAN content was used to calculate slurry application rates corresponding to 60 kg ha^{-1} TAN for the different treatments. This N rate was chosen as it is within the range for normal summer N application to grass ley, and the resulting amounts of slurry applied in the experiments were 35.0 metric tonnes ha^{-1} for CS and LF, and 17.5 metric tonnes ha^{-1} for BD. At the start of each experiment, a new slurry sample was taken (in experiment 4, one sample for each slurry type) and sent for laboratory analysis of DM content and TAN (Table 2).

For each of the experiments with slurry acidification, 5 L of slurry were acidified by manual addition of concentrated (96%) sulphuric acid while mixing until the target pH was reached. For practical reasons, and to give time for the foam that formed to wane, acidification was performed 2–4 h before slurry application. The target pH was set to 6.0 for CS and LF, as a balance between maximising ammonia abatement and minimising the amount of acid added. For BD the target pH was 6.7, to match the pH value in a concurrent fertilisation field experiment using the same slurry types, where the acidification of BD to pH 6.0 failed due to high buffering capacity and extensive foaming.

Table 1 – Start time, duration and temperature data for ammonia emissions experiments 1–5. Values within brackets represent minimum and maximum temperature.

Experiment	Experiment start (yyyy-mm-dd and time)	Experiment duration (h)	Air temp. during slurry application (°C)	Air temp., mean, 70 h (°C)
1	2019-06-05 10:30	73	25.6	19.3 (7.6, 28.9)
2	2019-06-13, 12:30	94	21.8	18.2 (8.7, 25.3)
3	2019-06-18, 13:00	122	24.6	18.3 (10.9, 25.8)
4	2019-06-25, 11:00	71	22.1	19.2 (9.3, 27.5)
5	2019-07-02, 13:45	93	19.7	14.9 (6.0, 26.1)

Table 2 – Overview of slurry types, slurry characteristics and application rates used in experiments 1–5.

Experiment	Slurry type	DM (%)	TAN ($kg\ ton^{-1}$)	Application rate ($kg\ TAN\ ha^{-1}$)	pH	H_2SO_4 (96%) added ($kg\ ton^{-1}$)	pH acidified
1	Separated (LF)	4.8	1.8	64	7.1[a]	7.4	6.0
2	Untreated (CS)	9.5	1.9	65	6.8[a]	8.3	6.0
3	Digested (BD)	3.9	3.2	56	7.6[a]	11.0	6.7
4	Untreated (CS)	9.0	1.9	66	6.8	0	–
4	Separated (LF)	4.6	2.0	71	7.1	0	–
4	Digested (BD)	4.4	3.4	60	7.8	0	–
5	Separated (LF)	4.5	2.0	72	7.1[a]	0	–

[a] pH measured on different samples from the same slurry types on other dates (June 25 for CS and LF, June 14 for BD).

Before the start of each experiment, the grass height within the plot area in experiments 1–4 was manually adjusted with scissors to 50–70 mm, to make it similar in all experiments. For slurry application, a 10-L watering can equipped with a watering hose was used to mimic trailing hose application. The slurry was applied in three bands at 0.25 m spacing within a metal frame (293 mm × 674 mm inner dimensions), inserted to 40 mm depth into the soil. Immediately after slurry application, the wind tunnel was mounted upon the frame and sealed to be airtight. In treatments with slurry injection or trailing shoe application, the soil slots were created before mounting the frame. Trailing shoe soil slots were created manually one by one with a trailing shoe disassembled from a Zunhammer slurry spreader, with the aim of mimicking those made by the slurry spreader in fertilisation experiments on the same experimental site. Injection slots were created manually with a two-sided metal pickaxe. One of the 90° corners of the broad (axe) end was used to create the 20 mm deep injection soil slots (Injection1), and the pointed (pick) side was used for the 50 mm deep (Injection2) slots (Fig. S1).

2.1.3. *Wind tunnels and ammonia emissions measurements*

The technique of using wind tunnels for ammonia emissions experiments has been described previously (e.g. by Lockyer (1984) and Misselbrook et al. (2005)). Full details of the experimental equipment and procedures for online ammonia measurements used in the experiments can be found in Pedersen et al. (2020). The setup comprised nine wind tunnels, which allowed for three treatments with three replicates per experiment. Each wind tunnel consisted of a rectangular, open-bottomed stainless steel emissions chamber connected to a fan via a steel duct. A small (335 mm × 13 mm) air inlet was positioned in the end of the wind tunnel, and the fan created a constant airflow through the tunnel (air exchange rate 25 min^{-1}, resulting in a calculated mean air velocity of 0.33 $m\ s^{-1}$). From the wind tunnels, air was led through insulated Teflon tubes heated to approximately 40 °C, via a valve block controlling which tube to measure from, to a cavity ring-down spectrometer (CRDS) (G2103 NH3 Concentration Analyzer, Picarro, CA, USA). Measurement interval of the instrument was 1 s and the measurement units were ppb, with a lower instrument detection limit of 0.03 ppb. Background ammonia emissions were measured at the air inlet of three of the tunnels, and one overall background measurement was made about 20 m away from the experimental site. The concentration of ammonia in the air from one measuring point was measured for 8 min, which was sufficient for reaching stable values under summer conditions. With 13 measuring points in total (9 wind tunnels, 4 background measurements), a complete round of measurements took 104 min. Data from the CRDS included measurements of ammonia concentration every second, as well as 30-s averages. The last 30-s average from each 8-min measuring period was used for ammonia flux calculations.

2.1.4. *Data treatment and statistical analysis*

For data treatment and statistical analysis, R (version 4.1.2) was used (R Core Team, 2022). The ammonia concentration data from the CRDS were converted from ppb to units of $g\ N\ L^{-1}$ and corrected for background concentrations based on the ideal gas law:

$$C = M \cdot (c_o - c_i) \cdot P / (R \cdot T) \tag{1}$$

where C is ammonia concentration ($g\ N\ L^{-1}$), P is total pressure (1 atm), R is the gas constant (0.08206 $L\ atm\ K^{-1}\ mol^{-1}$), T is temperature (K), c_o is outlet concentration or mixing ratio (ppb_v), c_i is inlet (background) concentration (ppb_v) and M is molar mass of nitrogen (14.01 $g\ mol^{-1}$) (equation (1) modified from Pedersen et al. (2020)). When outlet concentration was not higher than inlet concentration, C was set to zero. From the ammonia concentration calculated by equation (1), volatilisation flux of ammonia, in $g\ N\ m^{-2}$, was calculated as:

$$F_{NH_3} = C \cdot q / A \tag{2}$$

where F_{NH_3} is ammonia flux ($g\ N\ m^{-2}$), C is ammonia concentration ($g\ L^{-1}$), q is volumetric airflow rate (2016 $L\ min^{-1}$) and A is area under the wind tunnel (0.2 m^2) (Pedersen et al., 2020).

Cumulative ammonia emissions were calculated using the trapezoidal rule (Burden & Faires, 2001). Data gaps occurred in experiments 1, 2 and 3 (in experiment 1 at 45.1–50.7 h from start, in experiment 2 at 17.3–22.5 h from start and in experiment 3 at 29.5–33.7 h from start). Ammonia flux within the data gaps was estimated using linear interpolation between the two data points on each side of the data gap. In experiment 2, vacuum in the tubing system between the wind tunnels and the CRDS resulted in very low and unreliable data during the first measurement cycle. Correction for the low initial measurements was made by replacing the ammonia concentration values from the first measuring cycle by the values from the following measurement. This assumption was based on the emission patterns in the other experiments, where the values in the first measurement cycle were 76–136% of those at the next measurement. The difference in total emissions after 70 h before and after this correction was less than 4% for all treatments.

Statistical analysis of the results from the wind tunnel experiments was performed for each experiment separately. Analysis of variance (ANOVA) was carried out using a linear mixed-effects model, lmer()function, in the R package *lme4* (Bates, Maechler, Bolker, & Walker, 2015). For the ANOVA analyses, type III sums of squares and F-tests were used, and degrees of freedom were calculated with the Kenward–Roger method. Normality of the data was checked by normal probability plots, where sample quantiles are plotted against theoretical normal quantiles (Q–Q-plot). Homogeneity of variance within the data was checked by plotting model residuals against fitted values. In addition, the powerTransform()function in the R-package *car* (Fox & Weisberg, 2019) was used to determine whether transformation of data with the Box–Cox method (Box & Cox, 1964) would significantly increase model likelihood, and to find the optimal power transformation. For experiment 5, this resulted in transformation of the data to a power of −3 prior to the ANOVA analysis. Thereafter, the estimated means and the 95% confidence interval limits were back-transformed to the original scale.

The response variable in all analyses was cumulative ammonia emissions at 70 h, as fraction of TAN applied. Predictor variables in all single-experiment analyses were treatment and block, with treatment always set as a fixed factor and block always set as random factor. The overall analysis of experiments 1–3 used the same predictor variables, with the addition of experiment as a fixed variable. The R package *emmeans* (Lenth et al., 2022) was used for estimation of least-squares means, contrasts and related confidence intervals and *p*-values. Pairwise differences were analysed with the Tukey method and a significance level of 0.05.

2.2. Modelling ammonia emissions with the ALFAM2 model

The R version of the ALFAM2 model (v2.0) (Hafner & Haeni, 2022) was used in the modelling work in this study. The ALFAM2 model is a semi-empirical dynamic model for prediction of ammonia emissions, described in detail by Hafner et al. (2019). The latest model parameter values (Set 2) are based on measured ammonia emissions from field experiments in six countries and in total over 600 experimental field plots from the ALFAM2 database (Hafner et al., 2018). In the ALFAM2 model, values for most parameters (including those for wind speed) were based on micrometeorological measurements only, and therefore application to wind tunnel measurements is not straightforward (see also Sections 1.4 and 4.2). Parameter Set 2 is described in more detail in Hafner, Nyord, Sommer, and Adamsen (2021).

In the model, ammonia emissions are calculated based on the predictor variables slurry DM, application method, application rate, incorporation (shallow or deep), air temperature, wind speed and rainfall rate. Slurry TAN is divided into a “fast” pool and a “slow” pool from which emissions occur. This division of slurry TAN between the pools and rates of emission from the pools are quantified by a set of so-called primary parameters. The value of each primary parameter is based on a number of predictor variables, each combined with a coefficient (secondary parameter), the default values of which are determined by fitting the model with emissions data from the ALFAM2 database (Hafner et al., 2021).

For the experiments with wind tunnel measurements, precipitation was set to zero and wind speed was set to the average air velocity inside the wind tunnels, calculated from volumetric airflow rate and cross-sectional area. This substitution of average wind tunnel velocity for average open-air wind speed is undoubtedly imperfect, but it provides a simple way to evaluate the model using wind tunnel measurements (see Section 4.2 for more discussion on this topic). Slurry characteristics and pH values in the different experiments (Table 2) were used in the modelling. By comparing the emissions predicted by the model with those measured in the wind tunnels, it was possible to evaluate model performance, both in terms of flux patterns over time and as cumulative ammonia emissions after 70 h.

3. Results

3.1. Ammonia emissions

Cumulative ammonia emissions after 70 h varied between 2 and 32% of TAN, depending on the combination of application method and slurry type (Fig. 1). In the experiments with ammonia measurements lasting longer than 70 h (90–120 h), in most cases more than 90% of the total emissions occurred within the first 70 h after application (Table S2). The exceptions were acidified CS in experiment 2, with 79% of total emissions occurring within the first 70 h, and injection of LF to 50 mm in experiment 5, with 89% within the first 70 h, the total measuring time being around 90 h in both experiments (Table S2).

In most cases, 50% of the total ammonia emissions occurred within the first 14 h after slurry application (Table 3). For CS and LF, the time to reach 50% of total emissions was on average 11 h for all experiments with trailing hose application, while for BD with higher initial emissions rate it was only 6 h. In experiment 5, slurry injection slowed down the emissions, with the time taken to reach 50% of total emissions almost doubling for injection to 50 mm compared with trailing hose application (Table 3). The differences between trailing hoses and trailing shoes were small. Slurry acidification had varying effects, with acidified CS and LF taking the longest time of all treatments to reach 50% of total emissions and acidified BD taking the shortest time.

3.1.1. Slurry types

Cumulative ammonia emissions 70 h after trailing hose application of LF, CS and BD in experiment 4 represented 23%, 29% and 32% of applied TAN, respectively (Table 3). For LF and BD, the differences relative to CS were non-significant at 0.05 level ($p = 0.078$ and $p = 0.224$, respectively.) The highest initial emission rate was seen in BD, while CS showed the slowest decline in emissions (Fig. 2).

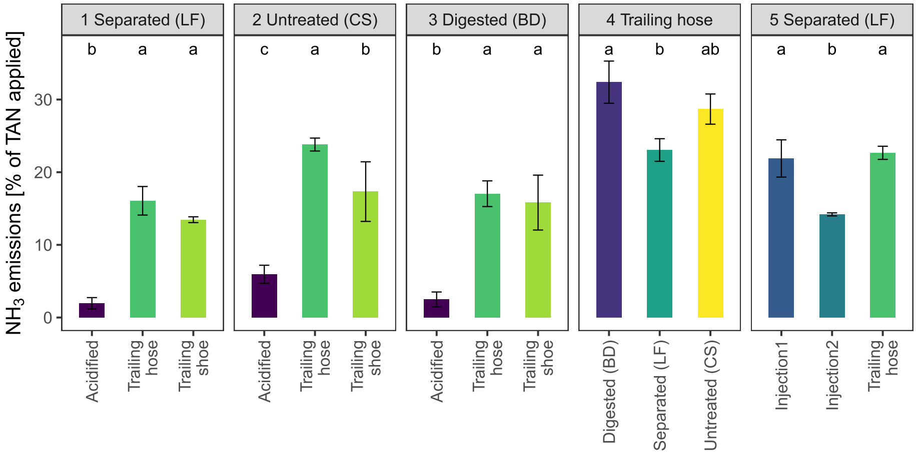

Fig. 1 – Cumulative ammonia emissions 70 h after slurry application (n = 3). Error bars represent 1 sd. Within each experiment (1–5), treatments with different lowercase letters are significantly different from each other.

3.1.2. *Application strategies*

The reduction in ammonia emissions for slurry application by trailing shoe compared with trailing hose was largest for CS (27%, $p = 0.036$), while it was lower and non-significant for LF (16%, $p = 0.127$) and BD (7%, $p = 0.826$) (Table 3).

For slurry injection in 50 mm deep open slots, cumulative ammonia emissions after 70 h were reduced by 37% ($p < 0.001$) compared with trailing hoses, while no difference was seen between trailing hoses and 20 mm injection slots (Table 3). The deep (50 mm) injection slots reduced ammonia flux compared with trailing hoses for more than 24 h, while the shallow (20 mm) soil slots did not reduce the emissions (Fig. 2).

3.1.3. *Slurry acidification*

Slurry acidification reduced cumulative ammonia emissions after 70 h for BD by 85% ($p = 0.004$), for LF by 88% ($p < 0.00103$) and for CS by 75% ($p = 0.001$) (Table 3). The emission patterns after acidification differed between the slurry types (Fig. 2). Acidified BD had the highest initial flux, with over 50% of the total ammonia emissions occurring within 5 h from slurry

Table 3 – Cumulative ammonia emissions after 70 h from experiment start. Within each experiment, treatments with different lowercase letter are significantly different from each other (letters and *p*-values based on Tukey's HSD test).

Experiment and slurry type	Treatment	Cumulative NH_3 emissions 70 h			Time for 50% of cumulative NH_3 loss
		Percent of TAN applied	95% confidence limits	*p*-value for the difference relative to reference treatment[b]	Hours from experiment start
1	Trailing hose	16.1 a	14.3, 17.8		10
Separated slurry (LF)	Trailing shoe	13.5 a	11.7, 15.2	0.1272	10
	Acidified[a]	2.0 b	0.2, 3.7	0.0003	39
2	Trailing hose	23.8 a	20.0, 27.6		13
Untreated slurry (CS)	Trailing shoe	17.3 b	13.5, 21.2	0.0362	12
	Acidified[a]	5.9 c	2.1, 9.8	0.0009	49
3	Trailing hose	17.0 a	13.5, 20.5		5
Digested slurry (BD)	Trailing shoe	15.8 a	12.3, 19.3	0.8263	8
	Acidified[a]	2.5 b	−1.0, 6.0	0.0044	4
4	Untreated[a]	28.7 ab	25.5, 31.9		10
All slurry types	Separated[a]	23.1 b	19.9, 26.2	0.0784	12
	Digested[a]	32.4 a	29.2, 35.6	0.2244	7
5	Trailing hose	22.6 a	20.5, 25.9		14
Separated slurry (LF)	Injection1	21.5 a	19.8, 24.1	0.7118	18
	Injection2	14.2 b	13.8, 14.6	0.0002	27

[a] Slurry applied by trailing hose.
[b] Reference treatment for comparison of slurry types in experiment 4 is Untreated; reference treatment for comparison of application strategies in experiments 1–3 and 5 is Trailing hose.

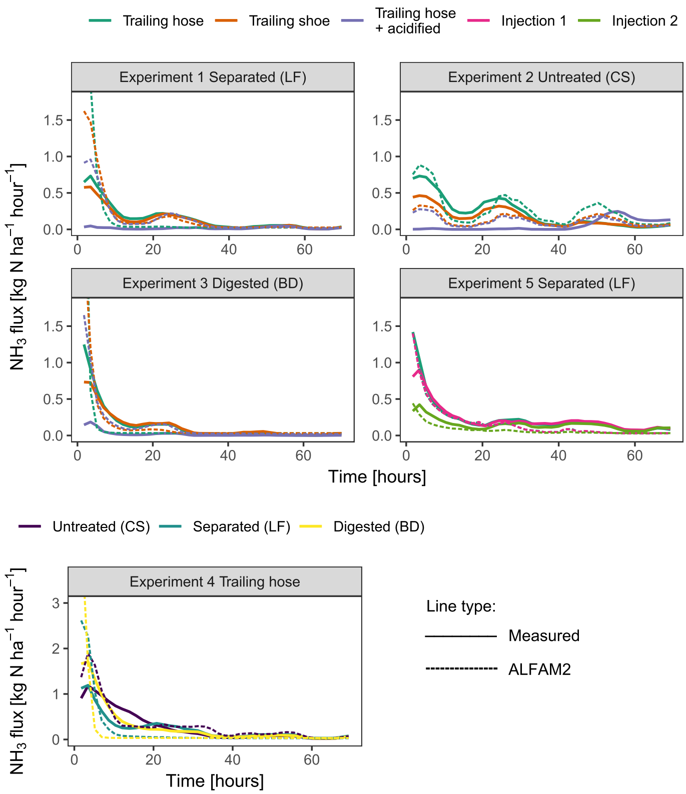

Fig. 2 – Ammonia flux over time in experiments 1–5. For experiments 1, 2 and 4, extreme ALFAM2 values (max 4.7 kg N ha^{-1} h^{-1}) immediately after slurry application have been excluded from the diagrams.

application (Fig. 2, Table 3), and negligible emissions from 30 h until experiment end (Fig. 2). For CS and LF, the initial flux was low and 50% of total ammonia emissions was reached after 49 and 39 h, respectively (Table 3). For acidified CS, the low initial emissions were followed by an increase after around 48 h, with higher emissions continuing until around 70 h after slurry application (Fig. 2).

3.2. *Ammonia emissions modelled with ALFAM2*

The ALFAM2 model with parameter set 2 predicted cumulative ammonia loss at 70 h with reasonable accuracy across all experiments (Fig. 3). The error ranged from −13 to +15% of applied TAN. Median model absolute error was 6% of applied TAN or 30% of measured emissions.

Differences between trailing hose, trailing shoe, trailing hose with acidification and open slot injection to 50 mm were approximately captured. However, the model substantially underestimated the effect of acidification for LF and BD. Although absolute cumulative emissions were relatively low for measurements (<5%) and for the model (10–20%), predicted ammonia emissions from acidified slurries were 7-fold and 8-fold higher, respectively, than those measured in wind tunnels. Both model and measurements showed a reduction in emissions from slurry separation in experiment 4, but this agreement did not hold for comparisons across experiments, where other variables presumably had an effect. Anaerobic digestion did not consistently affect measured or model-predicted emissions.

Comparison of measured ammonia flux against model predictions showed varying model performance. In three experiments (1, 3 and 4), the ALFAM2 model predicted much higher flux immediately after application than found in measurements (Fig. 2). In some cases, these high fluxes were followed by under-prediction of later fluxes, e.g., trailing hose application of LF and BD (Fig. 2). Diurnal patterns in measured flux (generally an increase at midday, but with diminishing magnitude over days) were approximately captured by the model in some cases (e.g. trailing hose in experiment 2). For acidified slurry in experiment 2, the overall increase in emission rate after 2 days was not reflected in model predictions.

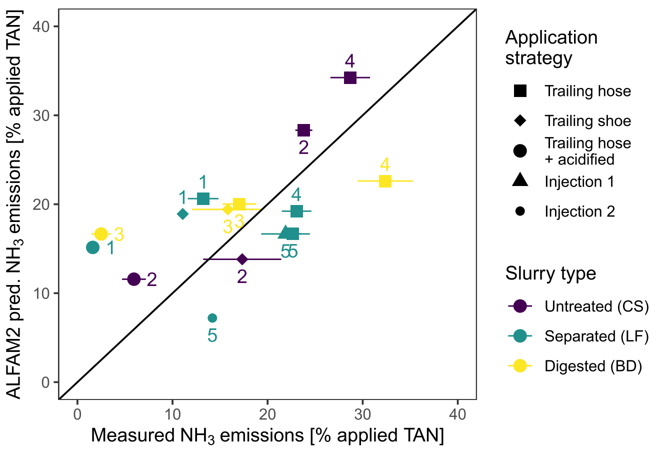

Fig. 3 – Measured cumulative ammonia emissions at 70 h from slurry application compared with values predicted by the ALFAM2 model, error bars representing 1 standard deviation. Experiment number is indicated beside each symbol. For observations above the solid black line, ALFAM2 over-predicted emissions compared with measurements, while values below the solid black line are under-predictions.

4. Discussion

4.1. *Ammonia emissions*

4.1.1. *Effects related to weather parameters*

No correlation was found between total ammonia emissions and mean temperature during the experiments or temperature at experiment start (Table 1). This is in line with results from previous modelling work by Pedersen, Nyord, Feilberg, and Labouriau (2021) based on data from wind tunnel experiments, which showed that if air temperature exceeds ~14 °C at slurry application, ammonia emissions do not increase with increasing temperature. Balsari, Dinuccio, Santoro, and Gioelli (2008) observed greater ammonia emissions from slurry application under summer conditions compared with autumn application, probably related to a considerably larger temperature span (mean experiment temperature 27.8 °C and 12.5 °C for summer and autumn, respectively) than in the present study.

Rainfall is known to cause crust formation at the surface of clay soils and thereby reduce the infiltration capacity (Lado, Ben-Hur, & Shainberg, 2007; Morin, Benyamini, & Michaeli, 1981). In the present study, a thunderstorm with hail and intense rainfall occurred six days before the start of experiment 4, followed by a period with no precipitation and increasing temperatures. This could possibly have contributed to the greater ammonia emissions during experiment 4 compared with experiments 1–3 (Fig. 1). For slurry types with low DM content in particular (such as LF and BD), the reduced infiltration capacity could have caused increased spreading out of slurry on the soil surface, and thereby greater ammonia emissions. In experiment 5, the greater overall emissions were most likely caused by factors other than crust formation, since a dense barley crop protected the soil surface during the thunderstorm.

4.1.2. *Effects of DM content and slurry pH*

Low DM content, as for BD and LF in the present study, may increase the infiltration rate of slurry into the soil and thereby lower ammonia emissions (Bhandral et al., 2009; Pedersen, Nyord, Feilberg, Labouriau, et al., 2021; Sommer, Jensen, Clausen, & Søgaard, 2006). In the present study, this was evident in the faster decline in emission rate for BD and LF compared with CS in experiment 4 (Fig. 2). Pedersen, Andersson, et al. (2021), who used the same slurry types as in our experiments, reported varying results depending on soil texture, with lower ammonia emissions from LF compared with CS in an experiment on clay soil, but higher ammonia emissions from a sandy loam, after application by trailing hoses. Only a few studies have included trailing hose application of the liquid fraction from separated slurry, but e.g. Amon et al. (2006) and Fangueiro, Pereira, et al. (2015), have reported lower total ammonia emissions from the liquid fraction compared with untreated cattle slurry. Studies on broadcast spreading report decreasing (Balsari et al., 2008; Owusu-Twum et al., 2017) or increasing (Dinuccio, Berg et al., 2011) ammonia emissions after application of the liquid fraction compared with the untreated slurry, as well as varying effects (Bhandral et al., 2009; Vandre, Clemens, Goldbach, & Kaupenjohann, 1997). Regarding important parameters such as infiltration, the same behaviour as seen for broadcast spreading can be expected for band application, as there is no soil manipulation with either of those application methods.

The low DM content in biogas digestate is counteracted by high pH, posing a greater risk of ammonia losses. In the present study, the higher initial emission rate from BD compared with LF was most likely a pH-related effect. Other studies, e.g.

Nicholson et al. (2018) and Amon et al. (2006), report greater emissions from biogas digestate than from untreated cattle slurry, while Pedersen, Andersson, et al. (2021) observed higher emissions from biogas digestate on a sandy loam soil, but no difference on clay soil.

The application rate in tonnes per ha was 50% lower for BD. Results from earlier studies on whether that would result in greater ammonia emissions expressed as % of TAN applied are inconsistent. Thompson, Pain, and Rees (1990) found decreasing relative ammonia emissions with increasing application rate, while Balsari et al. (2008) did not observe any difference. Klarenbeek and Bruins (1991) found a non-linear relationship, with reduced relative emissions when application rate increased from 30 to 90 m^3 ha^{-1}, but no difference at lower application rates between 10 and 30 m^3 ha^{-1}. In parallel experiments to the present study with the same slurry types applied with trailing hoses on clay soil (Pedersen, Andersson, et al., 2021), but with the same volumetric application rate of all slurry types, the differences in relative emissions between slurries, expressed as % of TAN, were very similar to those in the present study.

4.1.3. Effect of application method

4.1.3.1. Trailing shoes – experiments 1, 2 and 3. The average ammonia abatement of 17% is in line with findings by Pedersen et al. (2020), but lower than the >50% reduction reported by Malgeryd (1998), although there were large differences between the slurry types. The performance of the trailing shoes was affected by the hard, dry surface of the clay soil, resulting in rather shallow (approx. 10 mm deep) soil slots with very little loose soil, and hence little or no reduction in exposed slurry area compared with application with trailing hoses. On softer soil, the trailing shoe would ideally make approximately 30 mm deep slots, filled up to 10 mm with loose soil, to enable quick slurry infiltration. The study by Pedersen et al. (2020) indicated that the abatement effect of trailing shoes is dependent on soil properties, with the largest effects on more coarse-textured soils.

Larger slurry area has been shown to increase ammonia emissions (Sommer & Hutchings, 2001; Webb et al., 2010). The greater abatement effect for CS (27%) compared with LF (16%) could be attributable to the higher DM content of CS, reducing the spreading out of slurry beside the trailing shoe soil slots. For BD, the lower application rate, leading to narrower slurry bands and quick infiltration for both trailing hose and trailing shoe application, could explain the small ammonia abatement from trailing shoes (7%).

In earlier studies in which trailing hoses and trailing shoes were compared with broadcast application, the abatement effects were in some cases similar, e.g. Häni et al. (2016) found values of 51% and 53%, respectively. In other cases, trailing shoes have been found to be more effective than trailing hoses, e.g. Misselbrook et al. (2002) found a 57% and 26% emissions reduction, respectively. The DM content and pH of the slurry types in those two studies make them most comparable with LF in the present study, where the differences between trailing hoses and trailing shoes were non-significant. Misselbrook et al. (2002) found the weakest abatement effect from trailing shoe application to grass ley in experiments with very short grass and newly established (less dense) crop, conditions similar to those in experiments 1–4 in the present study.

4.1.3.2. Slurry injection – experiment 5. The 50 mm injection slots (Injection2) are most comparable with disc injection, but with slurry applied on the surface rather than injected belowground, while the shallow soil slots (Injection1) are comparable with trailing shoe application. The 37% reduction in ammonia emissions that was achieved with slurry injection in the present study is within the range observed in earlier studies in Scandinavia (Hansen et al., 2003; Rodhe & Etana, 2005), but would probably have been larger if the slurry had actually been injected below the soil surface, rather than being applied manually after slots were created. Earlier studies evaluating different slurry injectors have found large variations in the degree of reduction in ammonia emissions, depending on soil conditions and function of the injectors (Hansen et al., 2003; Misselbrook et al., 2002; Nicholson et al., 2018; Rodhe & Etana, 2005). Rodhe and Etana (2005) reported a reduction in ammonia emissions of 52% relative to band application for the most effective type of slurry injector (making a 40–50 mm deep open slot), while the least effective injector showed no reduction. In a Danish study evaluating different slurry injectors (Hansen et al., 2003), ammonia emissions were found to be reduced by 20–75% compared with band application. Hansen et al. (2003) found that ammonia emissions reduction potential was linearly correlated to the volume of the slots created by the injectors, and concluded that high reduction potential is dependent on creation of slots with sufficient volume to contain all the slurry applied.

A potential trade-off with slurry injection that cannot be neglected is the possible increase in nitrous oxide emissions reported e.g. by Emmerling, Krein, and Junk (2020) and Duncan, Dell, Kleinman, and Beegle (2017). Other studies have reported no increase in nitrous oxide emissions (Fangueiro et al., 2017) or differing results (increase and no increase) between years (Seidel et al., 2017) and between crops (Fangueiro et al., 2018). Although emissions of nitrous oxide need to be minimised from a climate perspective, from an agronomic perspective the reported nitrous oxide emissions from slurry injection are small (less than 2% of total applied nitrogen) (Fangueiro et al., 2018; Duncan et al., 2017; Seidel et al., 2017) compared with the nitrogen saved in the form of reduced ammonia emissions.

4.1.4. Effect of slurry acidification

The ammonia emissions abatement from slurry acidification was similar for all slurry types (75–88%), as could be expected since the reduction in pH was similar (Table 2). In other studies (Fangueiro et al., 2017, 2018; Seidel et al., 2017), acidification to pH 6.0 or below effectively reduced ammonia emissions. This was also the case in the present study, both for CS and LF acidified to pH 6.0 and for BD with a higher pH of 6.7. In a study by Wagner, Nyord, Vestergaard, Hafner, and Pacholski (2021), with pH reductions after field acidification comparable to those in the present study, the ammonia emissions reduction was smaller for anaerobic digestate, but similar for cattle slurry.

The higher initial emission rate from acidified BD (Fig. 2) compared with LF and CS can be explained by its higher pH value. The very low emissions during the remainder of the experiment were probably caused by a combination of factors reducing the remaining amount of TAN available for emission, i.e. high initial emission rate depleting the slurry TAN pool and quick infiltration due to low slurry DM content.

For acidified CS, the pronounced increase in emission rate after around 48 h (Fig. 2) is most likely explained by an increase in slurry pH, since no corresponding increase was seen in the non-acidified treatments and there was no extreme temperature increase at that time (Fig. S2). Several factors possibly contributed to a late increase in slurry pH, including microbial oxidation of slurry volatile fatty acids (VFAs) to carbon dioxide and the buffering capacity of slurry and soil (Sommer et al., 2003). Pronounced increases in ammonia emissions from acidified slurry several days after application have only been reported previously by Pedersen et al. (2022), who found that emissions were low during the first 120 h after application of cattle slurry, and thereafter increased.

4.2. *Modelling ammonia emissions with ALFAM2*

Although a wind tunnel environment cannot be expected to mimic open-air mass transfer perfectly, it has been found to provide similar ammonia emission measurements to open-air studies in some cases. The analysis in the original ALFAM work did not find a clear difference between wind tunnel and micrometeorological measurements by different research groups (Søgaard et al., 2002, Table 2). Sommer and Misselbrook (2016) reviewed cases involving active adjustment of wind tunnel air flow rate to match open-air wind speed close to the ground and found that this approach yielded similar emissions values to the Integrated Horizontal Flux (IHF) mass balance method. In the present study, the difference in cumulative emissions between wind tunnel measurements and ALFAM2 model predictions was small on average (see Section 3.2). High temperatures at the time of slurry application (Table 1) could have contributed to this better match, since recent experiments comparing wind tunnels with micrometeorological measurements have shown that under high-emission ambient conditions (in that case high temperature and wind speed), the two methods yield similar cumulative emissions (Hafner, Kamp, & Pedersen, 2023).

Differences between measured ammonia emissions and those predicted with the ALFAM2 model reflect several factors, including differences between wind tunnel measurements and the micrometeorological measurements used to estimate most model parameters, limitations in model structure, inaccurate parameter values, measurement error and effects not included in the model. With parameter values based on hundreds of plots from several countries, model evaluation with measurements from a single field study is not a sufficient basis for making changes to the model. However, the results in the present study indicate some processes that could benefit from additional attention.

Despite the importance of interactions between slurry and soil (Sommer et al., 2003, Sommer, Jensen, Clausen, & Søgaard, 2006), soil properties are not included as ALFAM2 predictor variables. This absence undoubtedly contributes to model error in general (Hafner et al., 2019) and possibly also in the present study. This limitation could be addressed by inclusion of soil effects in a new parameter set, but the large field studies necessary for isolating soil effects have not been carried out to date.

The effect of anaerobic digestion of slurry on ammonia loss is an important topic because of a recent increase in the practice in some regions (Adamsen & Hafner, 2021; Statistics Sweden, 2020). According to model parameters (Hafner et al., 2021) and interpretation of emissions measurements (Chantigny et al., 2009; Evans et al., 2018; Neerackal et al., 2015), the reduction in DM and increase in pH due to digestion have opposing effects on ammonia emissions. In experiment 4, BD had much lower DM and much higher pH compared with CS but measurements showed little difference in cumulative emissions, implying that the two changes practically cancelled each other out. In model predictions the DM reduction effect dominated, leading to over-prediction of the difference between BD and CS. Improving the ability of the model to capture these important effects should be a goal of future work.

An increase in ammonia emissions flux over time in the absence of an increase in temperature or wind speed, as seen for acidified CS, is incompatible with the ALFAM2 model structure. If this phenomenon (see Section 4.1.4) is found to be widespread following acidification, model structure would need to be changed to enable more accurate predictions. Although slurry surface pH is known to change following application, incorporation of these dynamics into a simple model seems implausible (Hafner, Montes, & Rotz, 2013; Pedersen et al., 2022).

In experiments 1, 3 and 4, the ALFAM2 model predicted much higher initial flux than shown by measurements. The wind tunnel system can underestimate ammonia flux in the first measurement cycle in some cases (Pedersen et al., 2020), and therefore it is not possible to determine the magnitude of model error during these periods. The pattern with high initial flux was associated with later underestimation due to depletion of TAN from the “fast” pool in the model. Within the model, the pattern was caused by very high values for the emission rate constant from the “fast” pool (r_1), caused in turn by somewhat higher air temperature and initial pH values (see Sections 2.1.1 and 2.1.2). Although parameter values were originally determined from measurements (Hafner et al., 2021), the comparison shown here suggests that the nature of the response ($\log_{10}(r_1)$ is directly proportional to wind speed, temperature and pH) should be examined.

Some of the apparent error in model predictions is undoubtedly due to the substitution of wind tunnel average air velocity for open-air wind speed, or stated differently, problems in taking wind tunnel results as representative of open-air emission (Section 1.4). Despite some evidence that wind tunnels which match ambient air speed can replicate micrometeorological results (Sommer & Misselbrook, 2016), average velocity may differ from speed, especially with highly turbulent flow. A comparison of ethanol evaporation with the same wind tunnels as used in the present study showed that the average wind velocity within the tunnels corresponds to a much higher numerical value of air speed in open air in terms of mass transfer (Pedersen et al., 2020). This suggests that the model may underestimate wind tunnel emission rate, but the opposite was actually observed. Considering all this, careful evaluation of the magnitude and nature of wind speed or wind tunnel airflow on emission rate is warranted.

The ALFAM2 model consistently underestimated the effect of acidification in reducing ammonia emissions in the wind tunnel experiments described here (Section 3.2). Unfortunately, emission reduction is not precisely related to pH or the change in pH (Nyord, Hafner, Adamsen, & Sommer, 2021) and resulting model parameters show high variability (Hafner et al., 2021).

5. Conclusions

As hypothesised, this study revealed an effect of slurry DM content on ammonia emissions, with a strong tendency for lower ammonia emissions (20% lower, $p = 0.078$) from LF than from CS. For BD, the low DM content was counteracted by high pH, and thus cumulative ammonia emissions were not different from CS. The performance of the trailing shoes was affected by a hard, compact soil surface and the ammonia abatement compared with trailing hoses was non-significant for LF (16%) and BD (7%). For CS the ammonia abatement was greater (27%) and statistically significant, most likely related to the higher slurry DM content reducing the spreading out of slurry beside the trailing shoe soil slots. As hypothesised, slurry injection into 50 mm deep open slots reduced ammonia emissions compared with trailing hoses (by 37%) and slurry acidification effectively reduced ammonia emissions for all slurry types, by 75–88%. An unexpected increase in emission rate from acidified CS was seen from 48 to 70 h after slurry application, indicating an increase in slurry pH. The understanding of differences in ammonia flux patterns and cumulative emissions would benefit from further investigation of changes in slurry surface pH after field application, especially for acidified slurries. The effect of the structure of the uppermost soil layer (e.g. crust formation on dry clay soils) on slurry infiltration and ammonia emissions also needs further investigation.

Comparison of emissions measured in wind tunnels and values predicted by the ALFAM2 model showed reasonable agreement for cumulative emissions, but poorer model performance in predicting emission dynamics in some cases. Although this assessment of the model must be tempered by problems in relating wind tunnel measurements to open-air emissions, future work on the ALFAM2 model might benefit from an evaluation of air temperature, airflow and slurry pH effects.

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Author contribution

Karin Andersson: Planning and preparation, Field experiments, ALFAM2 modelling, Data analysis, Writing – original draft (except Sections 3.2 and 4.2), Writing – review and editing, Visualisation. Sofia Delin: Funding, Planning and preparation, Field experiments, Writing – review and editing. Johanna Pedersen: Planning and preparation, Data analysis (support), Writing – review and editing. Sasha Hafner: ALFAM2 modelling, Writing – original draft (Sections 3.2 and 4.2), Writing – review and editing, Visualisation. Tavs Nyord: Planning and preparation, Writing – review and editing.

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.biosystemseng.2023.01.012.

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Ammonia emissions from untreated, separated and digested cattle slurry – effects of slurry type and application strategy on a Swedish clay soil

Supplementary material

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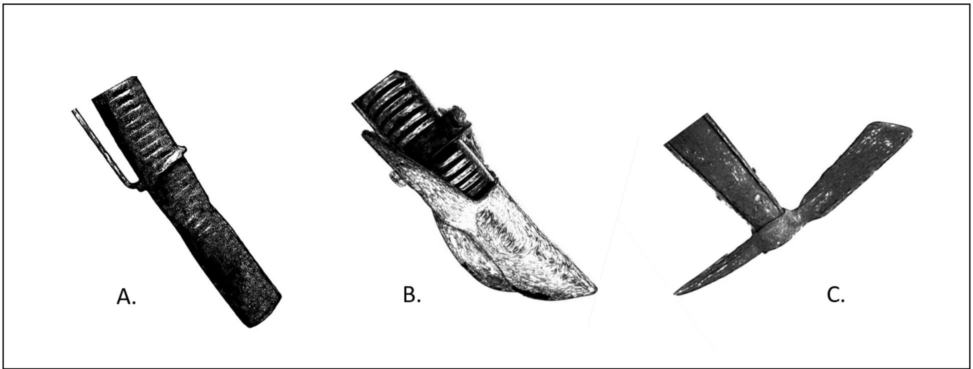

Figure S1. Schematic figure of A. trailing hose, B. trailing shoe and C. the metal hoe used to create slurry injection slots.

Table S1. Analysis of slurry samples collected in April 2019.

Slurry	**DM** (%)	**N_{tot}** (kg ton^{-1})	**N_{org}** (kg ton^{-1})	**TAN** (kg ton^{-1})	**C_{tot}** (kg ton^{-1})	**P_{tot}** (kg ton^{-1})	**K_{tot}** (kg ton^{-1})	**S_{tot}** (kg ton^{-1})	**pH**
BD	5.2	5.1	1.6	3.4	21.7	0.58	3.02	0.36	8.0
CS	9.8	3.7	2.0	1.7	45.6	0.54	3.72	0.41	7.7
LF	4.8	3.2	1.5	1.7	21.4	0.37	3.80	0.33	7.2

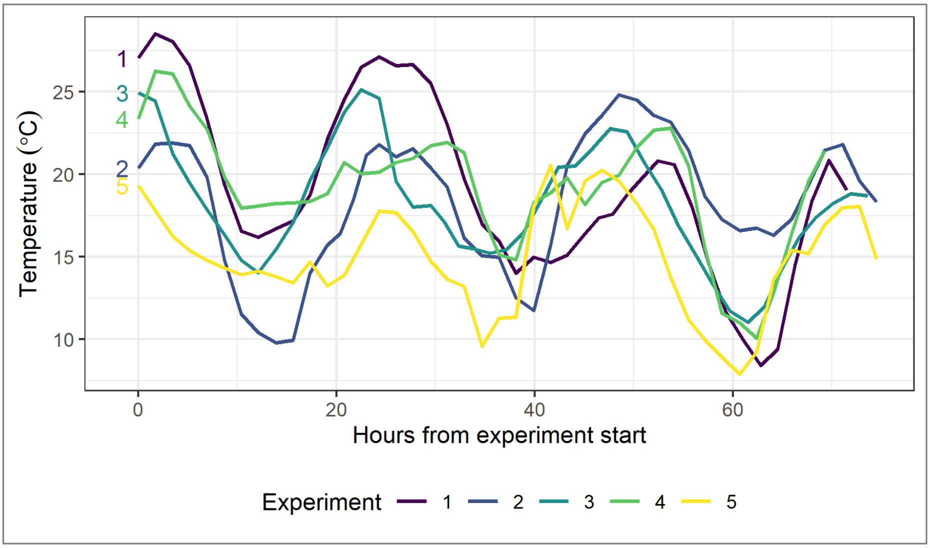

Figure S2. Air temperature for each of the five ammonia emissions experiments.

Table S2. Percentage of total cumulative ammonia losses occurring during the first 70 hours from experiment start.

Experi-ment	Slurry	Application method	Cumulative NH_3 emissions 70 hours (percent of TAN applied)	Total experiment duration (hours)	Cumulative NH_3 emissions total experiment time (percent of TAN applied)	Percentage of total NH_3 emissions during the first 70 hours
1	Separated	Trailing hose	16.1	73.2	16.3	98.6
1	Separated	Trailing shoe	13.5	73.2	13.7	98.6
1	Separated	Acidified	2.0	73.2	2.1	93.3
2	Untreated	Trailing hose	23.8	93.6	24.1	98.8
2	Untreated	Trailing shoe	17.3	93.6	18.3	94.7
2	Untreated	Acidified	5.9	93.6	8.0	74.5
3	Digested	Trailing hose	17.0	121.7	17.9	95.4
3	Digested	Trailing shoe	15.8	121.7	17.0	93.2
3	Digested	Acidified	2.5	121.7	2.6	94.5
4	Untreated	Trailing hose	28.7	71.0	28.9	99.3
4	Separated	Trailing hose	23.1	71.0	23.2	99.2
4	Digested	Trailing hose	32.4	71.0	32.6	99.4
5	Separated	Trailing hose	22.7	91.8	24.4	93.0
5	Separated	Injection1	21.9	91.8	24.0	91.1
5	Separated	Injection2	14.2	91.8	16.3	87.3

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This thesis investigated different strategies to increase the nitrogen fertiliser value of cattle slurry. Combinations of treatments reducing slurry dry matter content and measures to mitigate ammonia emissions after slurry application were tested. The results showed that both solid-liquid separation and anaerobic digestion increased the N fertiliser value of cattle slurry, and slurry acidification with sulphuric acid effectively reduced the ammonia emissions. The largest increase in slurry fertiliser value was obtained by the combination of anaerobic digestion and slurry acidification.

Karin Andersson received her doctoral education at the Department of Soil and Environment, SLU, Skara. She holds a Master of Science in Agriculture from SLU, Uppsala.

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