Exploring the potential of manure management for increasing nutrient circularity of intensive dairy farming systems

Qingbo Qu

Propositions

1. Mechanical solid-liquid separation is the most sustainable manure management technology in intensive dairy farms.
   (this thesis)

2. (Re)integration of crop farming into livestock production is the only way towards sustainable livestock intensification.
   (this thesis)

3. Including edible animal organs into human's diets promotes circular food systems.

4. Artificial intelligence relies fully on human intelligence.

5. Empathy, rather than awareness, fosters intercultural communication.

6. Social media companies have a social responsibility to address the consequences of social manipulation by their users.

Propositions belonging to the thesis, entitled

Exploring the potential of manure management for increasing nutrient circularity of intensive dairy farming systems

Qingbo Qu
Wageningen, 3rd July 2023

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Exploring the potential of manure management for increasing nutrient circularity of intensive dairy farming systems

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Thesis

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Abstract

Dairy farming systems have been rapidly intensified over the past several decades in many world regions. One of the primary challenges in these intensive dairy farming systems is managing manure in a way that benefits agricultural production while minimizing environmental impacts. To increase the understanding of losses of manure constituents during manure management, we first zoomed in on gaseous emissions (mainly NH_3, CH_4 and N_2O) from different manure management facilities. By conducting systematic literature reviews, we found large variation in reported nutrient losses across publications, especially for NH_3 and CH_4 emissions. Manure characteristics and temperature were identified as the main factors influencing these gaseous emissions. Based on the database compiled from systematic literature reviews, we proposed a modular approach and developed a flexible modular manure management (FarmM3) model. With contrasting manure management scenarios, the FarmM3 model allowed to quantify the degradation and losses of different manure constituents (e.g., OM, C, N, P and K) from manure management chains with different complexity, and to identify the most important parameters determining these losses. For highly intensive confinement dairy farms, improving manure management alone may not be enough to reduce nutrient losses due to high nutrient surpluses within farms. Thus, we zoomed out from nutrient losses from manure management chains and further investigated the impacts of various manure management chains and integration of crop and dairy production on nutrient use efficiency and circularity at whole farm level, including different farm components, such as dairy, manure, soil and crop. To simulate this, a whole farm model (FarmDESIGN) extended with a manure management module (FarmM3) was used to an intensive mixed crop-dairy farm in China. We found that manure management chains could be designed effectively to reduce nitrogen volatilization and soil N losses while improving soil OM balance. However, individual manure management technologies were insufficient to reduce N losses due to compensatory losses. Instead, combinations of slurry solid-liquid separation, covered storage of solid and liquid fractions during storage, and improved manure application could remarkably reduce N losses at manure management. Overall, we concluded that, to move towards sustainable intensification of dairy production, increasing nutrient circularity by improving manure management with multiple mitigation measures and integrating crop and dairy production within farm or between farms are essential. Policy support and improved communication of manure treatment technologies can facilitate adoption of improved manure management practices. Additionally, recoupling crop and dairy production beyond the farm scale is necessary for intensive dairy farms with limited land availability, and participatory approaches can help design effective scenarios for crop-livestock integration at the local or regional level.

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Chapter 1

General introduction

1.1 Intensification of dairy production

Dairy farming systems have been rapidly intensified over the past several decades in many world regions (Clay et al., 2020). With higher production efficiency, these intensive systems facilitated the increase in world milk production, from 378 million tons in 1999 to 544 million tons in 2022 (Food and Agriculture Organization of the United Nations (FAO), 2022). This intensification occurred with specialization, meaning fewer dairy farms and larger herd size per farm. For instance, the average dairy farm size in the United States increased from 80 cows in 1997 to 234 cows in 2017 (MacDonald et al., 2020), with more than half of cows kept in herds of more than 900 (Clay et al., 2020). Similarly, the number of dairy farms in the United Kingdom fell from around 35000 to 13000 between 1995 and 2017, while average herd size tripled (Dairy UK, 2017). The Netherlands is a highly intensive dairy farming country in the EU-27. Despite a considerable decrease in the number of dairy farms, the number of cows per farm has doubled from 51 cows in 2000 to 103 cows in 2021 (Wageningen Economic Research, 2022). Over the last two decades, China also has experienced a rapid intensification of dairy farming, with a significant decrease in the number of small-scale farms and a notable increase in the number of larger, industrial-scale operations (Fig. 1.1). According to data from the China Animal Husbandry and Veterinary Yearbook (2021), the number of dairy farms with less than 50 cows has decreased by 80% since 2007, while the number of farms with more than 1000 dairy cows has tripled from 339 in 2007 to 1338 in 2020. As of 2021, approximately 70% of cows in China were housed in herds with more than 100 cows.

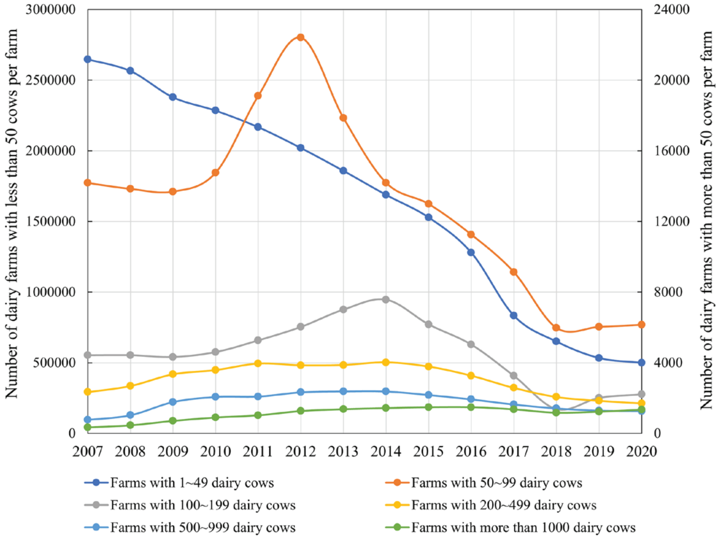

Figure 1.1. Changes in number of dairy farms with different sizes in China. Data source: China Animal Husbandry and Veterinary Yearbook, 2008-2021.

As a result of intensification and specialization, confinement dairy farming systems have become more common, characterized by high stocking densities and a heavy reliance on external feeds. In these systems, stocking rates can reach as high as 5-6 cows per hectare of land, or even higher in some cases (FAO, 2018). This high stocking density contributes to a substantial manure load per area. As shown in Fig. 1.2, in China, the amount of manure nitrogen excretion of dairy cows has increased 150% since 2000, while the total arable areas have remained stable (FAO, 2023). With little chance to recycle manure to croplands, these intensive systems have incurred site-specific environmental concerns (Chadwick et al., 2015; Clay et al., 2020). Environmental problems, such as climate change, nutrient surplus, eutrophication of ecosystems and biodiversity decline, have become main sustainability challenges of dairy intensification and gained worldwide attention (Rotz et al., 2006; Oenema et al., 2007; Garrett et al., 2020). The poor on-farm manure management and the spatial decoupling of crop and dairy farms were perceived as the major causes of undesirable environmental impacts of intensive confinement dairy farms (Bai et al., 2013; Chadwick et al., 2020). With a large amount of excreted manure but a limited area of cropland to utilize the produced manure, manure has become a burden for intensive confinement dairy farms. The management of manure in a way that is beneficial for agricultural production with minimal environmental and public health impacts has become one of the key challenges of intensive confinement dairy farming systems.

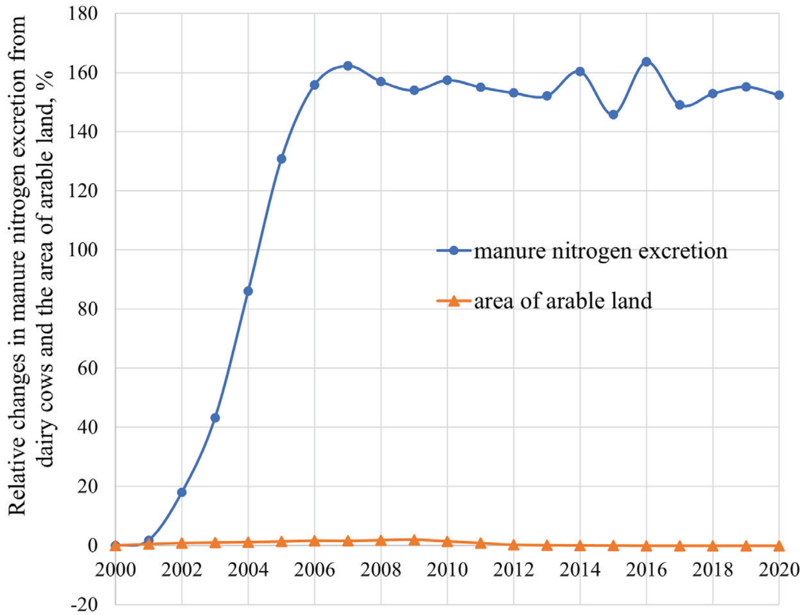

Figure 1.2. Relative changes in manure nitrogen excretion from dairy cows and the area of arable land in China since the year 2000. Data source: https://www.fao.org/faostat/en/#data

1.2 Losses of manure constituents from manure management

1.2.1 Pathways and magnitude of losses

Manure management is a continuum from livestock excretion to storage and treatment and finally to land application (Chadwick et al., 2011). Manure constituents, including organic matter (OM), carbon (C), nitrogen (N), phosphorus (P) and potassium (K) are easily to be degraded or lost from manure management systems. Manure carbon (C) could be lost via emissions of carbon dioxide (CO_2) and methane (CH_4) due to the degradation of organic C under aerobic and anaerobic conditions. Nitrogen, especially in inorganic form, can be easily lost by emissions of ammonia (NH_3), nitrous oxide (N_2O), nitric oxide (NO), nitrogen (N_2), leaching and runoff. Phosphorus and potassium are mainly lost by leaching and runoff. The pathways and the magnitude of losses of manure constituents depend on environmental factors and manure management practices (Oenema et al., 2007). Studies have presented the important effect of temperature on NH_3 and CH_4 emissions from manure management, with higher temperature increasing the release of gaseous losses from manure (Sommer et al., 2004; 2006). In addition to temperature, manure management practices also play a vital role in determining nutrient losses by gaseous emissions, leaching and runoff from manure management. For instance, significant reductions in NH_3 emissions have been observed by covering of manure storages, dilution with water, slurry acidification or injection of liquid manure (Chadwick et al., 2005; Berg et al., 2006; Shah et al., 2012; Petersen, 2018). There is an increasing interest in understanding effects of manure management practices on nutrient losses, which could contribute to improving the accuracy of loss estimations from manure management (Chadwick et al., 2011; Baral et al., 2018).

1.2.2 Manure management practices

Manure management practices, i.e. how manure is collected, stored, treated, and used, are diverse in farms with different sizes (Aguirre-Villegas and Larson, 2017; Niles et al., 2022). In traditional dairy farms, cows are fed with on-farm produced feed and manure is collected in solid form from deep litter, which is then stored for a certain period before being applied to fields. However, since the 1960s, there has been a shift towards the use of free-stall cubical stables, which encourages the collection of both feces and urine together in slurry (Bewley et al., 2017). In addition, with intensification of dairy production, on-farm manure management has evolved beyond traditional collection, storage, land application approaches. A series of new manure management facilities and technologies (e.g., covering, acidification, anaerobic digestion (AD), solid–liquid separation (SLS) and composting) have been developed and included in manure management systems to increase efficiency of use of nutrients, to reduce GHG emissions, to facilitate manure transportation and to produce energy. The use of these technologies leads to particularities and differences of the manure management chains in intensive confinement dairy farming systems compared to land-based livestock systems elsewhere in which manure can be collected and recycled to farmland after storage and only a few manure treatment technologies are applied. This difference clearly highlights the need for analysis of nutrient losses from different manure treatment technologies and for evaluation of impacts of these treatment technologies on flows and losses of nutrient from the whole manure management chains.

1.2.3 Pollution swapping of different manure constituents

Application of these emerging manure treatment technologies may induce changes in physical, chemical and/or biological properties of manure and in biophysical processes during handling, storage and application, and hence influence the fate of manure constituents within manure management chains (MMCs) (Hou, 2014; Khalil et al., 2016; Aguirre-Villegas et al., 2019). Previous studies have proven that the single mitigation measure on a single loss pathway might lead to the pollution swapping by increased losses of other compounds (De Vries et al., 2015). For example, the reduced NH_3 emissions by covering slurry storage and by injection of liquid manure can result in increased N_2O emissions (Sommer and Hutchings, 2001; Berg et al., 2006). Moreover, the reduced gas emission at previous stage might lead to an increased losses at later manure management stages (Shah et al., 2013). Given the possible interactive effects of manure management practices on emissions, the importance of integrated modelling approaches in estimating gaseous emissions and nutrient flows from a whole chain perspective has been pointed out (Hou et al.,2014; Sajeev et al., 2017; Wei et al., 2021).

1.2.4 Modelling approaches for quantifying losses from MMCs

Modelling approaches to quantify flows and losses of manure constituents associated with livestock manure management systems have been developed using either mechanistic process-oriented approaches, empirical equations, or emission factors, varying in the complexity of application and accuracy of estimation. Process-based mechanistic models, such as Manure-DNDC (Li et al., 2012) and Integrated Farm System Model (IFSM) (Rotz et al., 2012) consider biochemical and biophysical processes that govern the transport and transformation of nutrients in the manure life cycle. These mechanistic approaches offer robustness and flexibility of use in different manure management systems. However, simulations based on these models require large sets of data for parameterization and are sometimes too complex for assessments at the farm scale.

Modelling approaches based on mass-flow analysis and emission factors need fewer parameters and are much simpler in principle for estimating nutrients losses from MMCs when compared with mechanistic and empirical models and tools (Petersen et al., 2013). However, most of these approaches mainly focus on traditional manure management facilities, i.e., a linear process of manure excretion, manure storage and application (Webb and Misselbrook, 2004; Olesen et al., 2006; Dämmgen and Hutchings, 2008). Few of them allow to integrate the emerging on-farm manure management facilities (e.g., SLS, AD, composting, etc.) and to evaluate the impacts of these new manure management facilities on nutrient losses along the whole MMC. In addition, most of these approaches focus on only one or a few manure constituents or gaseous emissions. Conversions and losses of manure OM, P and K from MMCs are sparsely considered. It was reported that over 50% of the excreted manure P and K could be lost from MMCs (Bai et al., 2016). Knowledge of degradation of OM and losses of P and K along various MMCs could contribute to a more comprehensive assessment of the performance of MMCs, further promoting adoption of sustainable manure management technologies.

1.3 Effects of manure management on farm nutrient management

1.3.1 The importance of manure management

Manure management is a highly important part of farm carbon and nutrient cycles (Rufino et al., 2007). Proper manure management could reduce nutrient losses throughout manure management chains and conserve nutrients in manure products which can serve as valuable organic fertilizers for croplands and can result in reduced use of synthetic fertilizers. The effects of improved manure management on farm nutrient use efficiencies have been reported to vary among farms. For instance, in a study by Rotz et al. (2006) for a farm with 100 cows on 100 ha cropland and a farm with 1000 cows on 600 ha cropland, implementing nutrient conservation technologies, including a barn floor for feces and urine separation, covered six-month manure storage, and manure injection, reduced total farm N losses by 24% to 29% and improve whole farm N use efficiency by 5% to 7%. The largest reductions in N losses were obtained on the 1000-cow farm where initial losses were high due to a greater number of animals per unit of cropland (Rotz et al., 2006). Tan et al. (2022) found that the increases in farm N use efficiencies ranged from 0 to 53%, with large differences among dairy farms and among emission mitigation measures. For highly intensive confinement dairy farms, it might be insufficient to increase nutrient use efficiencies by only improving manure management due to high nutrient surplus within farms. The integration of crop and dairy production offers an effective approach to addressing the nutrient surplus challenge by exchanging manure as a source of crop fertilizer and crop products as feeds. Research has highlighted the importance of reintegration dairy and crop production to close nutrient loops and to increase nutrient use efficiencies (Garrett et al., 2020; Schut et al., 2021).

1.3.2 Integration of crop and dairy production

Integrated crop and dairy production systems increase the possibilities of better recycling of nutrients within systems, limiting recourse to the purchase of increasingly expensive inputs and safeguarding the biodiversity of agricultural ecosystems (Ryschawy et al.,2012; Peyraud et al., 2014; Lemaire et al., 2014). The reintegration of crop and dairy production can be envisaged at farm, regional and national scales (Russelle et al., 2007; Peyraud et al., 2014; Ryschawy et al., 2017). For intensive and specialized dairy farms, the availability of land that can be used to produce feeds for cows and to utilize manure and of labor to manage croplands are the main constraints to recouple crop and dairy production (Regan et al., 2017). An alternative way for these farms is to exchange animal manure and crop products of surrounding crop farmers. This form of integration needs strategic planning among crop and livestock farmers to match supply and demand of exchangeable materials (Martin et al., 2016).

1.3.3 Whole farm models

A variety of quantitative and qualitative farming system assessment approaches have been developed to support the analysis of current farming systems and the design and evaluation of alternatives (Martin et al., 2013). Among these approaches, optimization approaches can generate alternative farming systems by exploring the solution space using problem-solving algorithm (Martin et al., 2013; Ryschawy et al., 2014). Whole-farm models that can represent the interactions between farm components (i.e., livestock, manure, soils and crops) and optimize farm configurations based on mathematical techniques could be powerful tools to design integrated crop-dairy farms that achieve a better balance between supply and demand of feedstuff and manure.

The whole FarmDESIGN model, developed by Groot et al. (2012), (i) considers various aspects of mixed crop-dairy farm management, such as herd size and composition, manure management, crop production and feed management; (ii) can identify complicated interactions among farm components and quantify material flows among these components; (iii) allows to evaluate farm performance with environmental, economic and productive indicators; (iv) supports to explore alternative farm configurations using Pareto-based multi-objective optimization algorithm, and to provide redesign plans under given constraints and conditions. These characteristics make this whole farm model a useful tool to provide redesign plans for integrated farms that combine livestock with crops and/or grassland. These redesigns aim to improve farm nutrient use efficiency, to increase self-sufficiency of feed and to guarantee food production.

1.4 Research objectives

The main objective of this thesis was to increase the understanding of conversion and losses of different manure constituents along complex manure management chains, and to explore alternative options to increase nutrient use efficiency and circularity of intensive dairy farming systems with diverse manure management chains.

Specific research objectives were:

- To provide comprehensive overviews on gaseous emissions from different facilities of dairy manure management chains, and to identify environmental and biophysical factors affecting these emissions (Chapters 2, 3 and 4).
- To develop and test a flexible and extendable manure management model that allows to quantify conversions and losses of manure constituents along manure management chains with different complexity (Chapters 5 and 6).
- To investigate the effects of improved manure management chains on nutrient losses from the whole farm and to explore scenarios to increase nutrient use efficiency and circularity of intensive dairy farming systems (Chapter 7).

1.5 Research methods

I approached the above research objectives by using the following research methods:

1.5.1 Systematic literature reviews

Systematic literature reviews were carried out to increase the understanding of gaseous emissions (e.g., NH_3, CH_4, N_2O) from dairy barns, slurry storage and solid manure composting. Search queries with keywords were first formulated and systematic searches were conducted in different bibliographical databases, such as Web of science, Scopus and CAB abstracts via Ovid, to fully cover the published studies. A series of selection criteria was further set for a systematic selection of articles to ensure reliability of searched publications. A database including reported gaseous emissions from dairy manure management facilities and secondary variables were compiled and analyzed using linear mixed effects models and principal component analysis. Results about gaseous emissions from dairy barns, slurry storage and solid manure composting, and their potential environmental and biophysical influencing factors are presented in detail in Chapters 2, 3 and 4.

1.5.2 Model development

For objective 2, we combined a mass balance approach with a modular concept to increase the flexibility of the developed manure management module (FarmM3) in quantifying flows and losses of manure nutrients (Chapters 5 and 6). The model framework was represented in a modular way, by which the methods of manure management can be selected by the user of the model. Within this framework, the quantity and quality of the input manure and other materials can be calculated or can be defined by the user. The effluent of one step is the influent to the next step, whereby within each step, a mass balance is calculated, keeping track of all changes in different manure constituents (OM, C, N, P and K) due to conversions, losses and emissions. This modular approach makes the developed model highly adaptable and allows model users to build simulations tailored to their manure management systems.

1.5.3 Whole farm analysis

In Chapter 7, the whole-farm model FarmDESIGN and the newly developed FarmM3 model were used to investigate the impacts of improved manure management on nutrient losses from an intensive mixed crop and dairy farm and to explore alternative solutions to increase nutrient use efficiency and circularity of this farm. The FarmDESIGN model is a bio-economic whole farm model that supports evaluation of mixed crop-livestock farm performance comprehensively with various agronomic, environmental and economic indicators (Groot et al., 2012). This model mainly consists of five types of components, including household, animals, manures, soils and crops (Ditzler et al., 2019). It can be used to simulate flows of organic matter, carbon, nitrogen, phosphorus and potassium to, through and from these components on an annual basis. It is also able to capture the complicated interactions among different components. Taking FarmM3 as an external manure module of FarmDESIGN could improve the flexibility of FarmDESIGN model in estimating nutrient losses from various manure management chains and allow to evaluating the effects of various manure management chains on increasing nutrient use efficiency and nutrient circularity at whole farm level.

1.6 Thesis outline

This thesis contains a general introduction (Chapter 1), six thesis chapters (Chapters 2-7) and a general discussion (Chapter 8). The outline of thesis chapters is represented in Fig. 1.3. Chapters 2, 3 and 4 zooms in on gaseous emissions (mainly NH_3, CH_4 and N_2O) from different manure management facilities. Based on the database complied from systematic literature reviews, we propose a modular approach that allowed to quantify N flows and losses from the whole manure management chains with different complexity in Chapter 5.

Chapter 6 extends this modular approach by including other manure constituents (i.e., OM, C, P and K) and develops a modular manure management model (FarmM3) model. Chapter 7 zooms out from nutrient losses from manure management chains and investigates the impacts of various manure management chains on nutrient use efficiency and circularity at whole farm level including different farm components (e.g., dairy, manure, soil and crop).

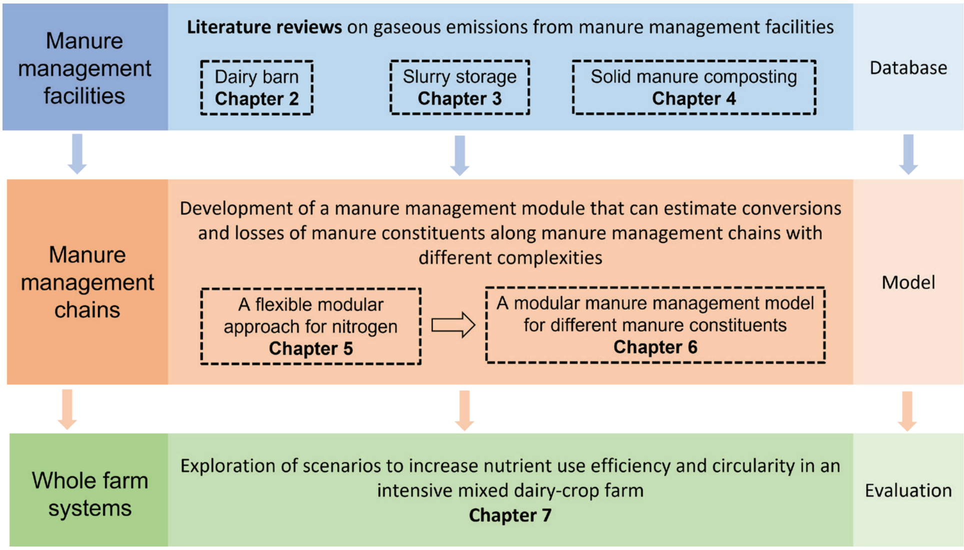

Figure 1.3. Thesis outline.

Chapters 2, 3 and 4 give a comprehensive overview of the magnitude of nutrient losses via gaseous emissions at different stages of manure management chains. We further analyze to what extent the environmental and biophysical factors could affect these losses based on the compiled measurement data.

Chapter 5 introduces a modular approach to estimate TAN and organic N flows, and to quantify different N species loss (e.g., NH_3, N_2O, NO and N_2 emissions, N leaching and runoff) from MMCs with different complexity in dairy farms. The feasibility of this modular approach is validated by comparing estimated results with other approaches.

Chapter 6 presents a newly developed modular manure management model (FarmM3) that allows to quantify degradation and losses of manure OM, C, N, P and K throughout MMCs with diverse manure management facilities. Winding stairs sensitivity analysis is performed to identify the most important parameters for determining losses of manure constituents from complex manure management chains.

Chapter 7 focuses on the impacts of manure management on improving nutrient use efficiency and nutrient circularity at whole farm level. Using FarmDESIGN and FarmM3 models, environmental performance of an intensive dairy-crop farm is simulated under various manure management chains. Alternative farm configurations to (re)integrate crop and dairy production are explored with multi-objective exploration to further improve nutrient use efficiency and increase circularity.

Chapter 8 synthesizes and discusses the outcomes of the research chapters. This final chapter puts the findings in this thesis in a broader context, reflects on methodologies used and identifies remaining future research needs.


UNIFARM

Chapter 2

Effects of housing system, measurement methods and environmental factors on estimating ammonia and methane emission rates in dairy barns: a meta-analysis

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Abstract

This study presented a meta-analysis of measured ammonia (NH_3) and methane (CH_4) emissions from dairy barns. A total of 27 peer-reviewed articles were selected to explore relationships between gas emission rates and housing system, measurement methods and environmental factors using linear mixed effect models. A large variation in measured gas emission rates from dairy buildings were observed, with 3.6 to 109.4 g/AU/d (AU refers to Animal Unit equaling 500 kg live weight) for NH_3 emission and 102.1 to 462.2 g/AU/d for CH_4 emission. Ammonia emissions were mainly influenced by temperature and relative humidity, with higher temperature leading to higher NH_3 emission but conversely for relative humidity. There were no significant differences in NH_3 emission rates among different measurement techniques for ventilation rate and gas concentration. The emission of CH_4 from dairy barns increased with the increase of temperature but was not significantly affected by relative humidity and wind speed. Measurement techniques for ventilation rate could significantly affect CH_4 emission estimates, with higher emission measured by CO_2 balance methods and inverse dispersion and lower emission measured by anemometers. Both NH_3 and CH_4 emissions presented no significant difference between solid floor and slatted floor, or between flushed and scraped systems. Our results indicate that environmental factors have more pronounced effects on NH_3 and CH_4 emissions than housing factors. It is necessary to establish gaseous emission factors for particular climate zones. Standardized measurement methods for gas emission rates from dairy barns are needed to reduce large variability and uncertainty.

Keywords: dairy housing, gas emission, floor type, manure handling method, measurement technique, temperature

2.1 Introduction

Emissions of ammonia (NH_3) and methane (CH_4) from livestock production are a major environmental concern worldwide as these gases contribute to global warming and play an important role in eutrophication of ecosystems, and airborne particulate matter pollution (United States Environmental Protection Agency, 2004; 2006; Stokstad, 2014). With the intensification of dairy production, dairy barns have been identified as an important source of NH_3 and CH_4 (Ngwabie et al., 2009; Owen and Silver, 2015; Drewry et al., 2018). About 20% of ammoniacal nitrogen or even more is lost via NH_3 emission from dairy housing (European Environment Agency, 2016; Sommer et al., 2019). The total CH_4 emission from dairy barns ranges from 132 - 390 g/head/d, most of which arising from enteric fermentation and less from manure storage in barns (Monteny et al., 2001; Joo et al., 2015; Cortus et al., 2015; Schmithausen et al., 2018). Quantification of these gas emission rates in dairy cow barns could contribute to developing accurate emission inventories and effective mitigation strategies.

Many previous studies have been performed to measure these gaseous emissions in dairy cow barns (Zhang et al., 2005; Leytem et al., 2012; Chiumenti et al., 2018). The relationships between gaseous emissions and potential influencing factors (e.g., floor type, manure handling methods and ambient environmental factors) have also been investigated (Sommer et al., 1991; Moreira and Satter, 2006; Cortus et al., 2015; Edouard et al., 2019). However, in commercial dairy cow barns, due to the compounded effects among potential influencing factors, it is unclear whether reported variations in gas emission rates can be attributed to effects of manure management or ambient environment. In addition, the gas emission estimates in dairy barns depend highly on the measurement of ventilation rate and gas concentration (Ogink et al., 2013). Different measurement techniques might lead to great variation in measured gas emission rates (Samer et al., 2011; Bai et al., 2017; Janke et al., 2020). Investigating the influence of measurement techniques on estimating gas emission rates will contribute to a better understanding of the variability of gas emission rates from commercial dairy barns.

The goals of this study were to collect and analyze data on measured NH_3 and CH_4 emission rates in commercial dairy housing systems and to investigate potential factors influencing these emission rates. Published measurements of NH_3 and CH_4 emissions from commercial and research-oriented dairy housing systems were reviewed. The relationships between these gas emission rates and floor type, manure handling methods, measurement techniques, and ambient factors were investigated. The results will allow us to refine our understanding of the role of different potential factors in influencing gas emission rates and how gaseous emissions may change as factors change in practical dairy barns.

2.2 Materials and methods

2.2.1 Data sources and selection

To fully cover the published research on reported NH_3 and CH_4 emissions from dairy housing, a systematic literature search was performed using CAB abstracts via Ovid, Scopus and Web of Science bibliographical databases. Specific search categories of dairy cows, manure, housing and gas emissions were combined in the literature retrieval for each database (See Box A.1 in Supporting Data). The body of literature was limited to journal articles published in English before July of 2018. The searched papers were first selected based on the title and abstract. Then the full paper was inspected. Selected articles were required to have the following characteristics: (i) published after 1990, (ii) focused on dairy cows that were kept in barns all day, (iii) reported at least one gaseous emissions (NH_3 or CH_4) from housing, (iv) emissions were measured for at least 24 hours, (v) conducted in commercial barns; that means studies based on laboratory study or model simulation were not included. The various reported units of gas emissions from articles were converted into a uniform unit of g/AU/d (1 AU equals 500 kg live weight) to make emission rates comparable for cows at different growth stages. The unit conversion of emission data is shown in Table A.1.

2.2.2 Data preparation

A total of 27 articles were included for data extraction. Detailed information of the 27 selected studies is shown in Table 2.1. To investigate the relationship between potential influencing factors and gas emission rates from dairy housing, we extracted NH_3 or CH_4 emission rates, housing factors (floor type and manure handling methods), features of measurement methods (ventilation rate and gas concentration) and environmental factors (air temperature, relative humidity and wind speed) from the selected articles to compile the database. Key factors were further categorized as following: (i) floor type: solid floor vs. slatted floor; (ii) manure handling methods: scraped vs. flushed; (iii) measurement methods of ventilation rate: anemometers vs. inverse dispersion vs. passive flux sampling vs. CO_2 balance method vs. tracer decay method; (iv) measurement methods of gas concentration: photoacoustic spectroscopy vs. spectrophotometry vs. other gas analyzers (e.g., based on electrochemical detection or gas chromatography or other). The principal of each category of measurement method of ventilation rate and gas concentration are shown in Table A.2 and Table A.3, respectively. If extracted factors were presented only in graphs without reporting the corresponding numeric values, we quantified the values using the software WebPlotDigitizer that allows extraction of numeric values from images or graphs (Burda, O'Connor, Webber, Redmond, Perdue, 2017). Among 27 publications, data from 7 articles were obtained using the WebPlotDigitizer. A total of 100 and 55 observations relating to NH_3 and CH_4 emission rates from dairy barns were extracted, respectively.

2.2.3 Statistical analysis

There were 81 NH_3 emission measurements and 37 CH_4 emission measurements simultaneously reporting floor type, manure handling methods, measurement methods of ventilation rate and gas concentration. The effects of these potential factors on gas emission rates were analyzed together with linear mixed effect models using the *lme4* package (Bates, Maechler, Bolker, Walker, 2014). We selected floor type, manure handling methods and measurement methods of ventilation rate and of gas concentration as main fixed effects and references were considered as random effects. Due to the limited sample size, we only considered main effects of predictors and did not investigate their interactions. In addition, the effects of temperature, relative humidity and wind speed on gas emission rates were investigated separately by linear mixed effect models. Environmental factors were selected as main effects and references were selected as random effects. The composition of fitted models and corresponding number of measurements are shown in Table 2.2.

Table 2.2. The composition of fitted models to investigate the effects of floor type, manure handling methods, measurement methods of ventilation rate and of gas concentration and environmental factors (temperature, relative humidity, wind speed) on gas emission rates.

Models	Number of measurements
Model 1: lmer (log (NH_3) ~ floor type + manure handling methods + measurement techniques of ventilation rate + measurement techniques of gas concentration + (1|Reference))	81
Model 2: lmer (log (NH_3) ~ Temperature + (1|Reference))	89
Model 3: lmer (log (NH_3) ~ Relative humidity + (1|Reference))	24
Model 4: lmer (log (NH_3) ~ Wind speed + (1|Reference))	46
Model 5: lmer (CH_4 ~ floor type + manure handling methods + measurement techniques of ventilation rate + (1|Reference))	37
Model 6: lmer (CH_4 ~ Temperature + (1|Reference))	47
Model 7: lmer (CH_4 ~ Relative humidity + (1|Reference))	29
Model 8: lmer (CH_4 ~ Wind speed + (1|Reference))	25

The normality of gas emission rates was tested using Shapiro-Wilk test (Royston, 1995). When gas emission rates did not fit a normal distribution, we used the *powerTransform()* function in the *car* package to determine the optimal normality transformation power (Box and Cox, 1964; Fox and Weisberg, 2019). We applied log-transformation for NH_3 emission and no transformation for CH_4 emission. Significance test for fixed effects and random effects were performed using Type III analysis of variance and likelihood ratio test, respectively. Pairwise difference of levels in each fixed effect type were tested using package *multcomp* (Hothorn et al., 2008). Goodness-of-fit of linear mixed effect models were computed using *r2()* function in the *sjstats* package (Nakagawa et al., 2017). All statistical analyses were performed using R statistical software (R Core Development Team, 2017).

Table 2.1. Detailed information of 27 selected studies in this study.

Reference	Region	Floor type	Manure handling methods	Measurement Methods	
				Ventilation rate	Gas concentration
Demmers et al., 1998	England	solid	scraped	Tracer decay method	gas analyzer
Phillips et al., 1998	England	solid	scraped	Passive flux sampling	spectrophotometry
Groot Koerkamp et al., 1998	England, Netherlands, Denmark, Germany	NA [a]	NA	CO_2 balance method	gas analyzer
Amon et al., 2001	Austria	slatted	NA	anemometers	photoacoustic spectroscopy
Jungbluth et al., 2001	Germany	slatted	NA	anemometers	photoacoustic spectroscopy
Snell et al., 2003	Germany	slatted, solid	scraped	Tracer decay method	Innova
Dore et al., 2004	England	solid	scraped	Passive flux sampling	spectrophotometry
Zhang et al., 2005	Denmark	solid	scraped	CO_2 balance method	Innova [b]
Flesch et al., 2009	America	solid	scraped	inverse dispersion	photoacoustic spectroscopy
Ngwabie et al., 2009	Sweden	slatted	scraped	CO_2 balance method	Innova
Pereira et al., 2010	Portugal	solid	scraped	Passive flux sampling	spectrophotometry
Ngwabie et al., 2011	Sweden	solid	scraped	CO_2 balance method	Innova
Samer et al., 2011	Germany	solid	scraped	CO_2 balance method	Innova
Samer et al., 2011	Germany	solid	scraped	CO_2 balance method	Innova
Zhu et al., 2012	China	solid	NA	CO_2 balance method	Innova
Leytem et al., 2012	America	solid	flushed	inverse dispersion	Innova
Schrade et al., 2012	Switzerland	solid	scraped	Tracer decay method	photoacoustic spectroscopy
Schiefler., 2013	Germany	slatted	flushed	Tracer decay method	Innova

	Gas emissions, g/AU/d		N [c]	Temperature, °C	Relative humidity, %	Wind speed, m/s
	NH_3	CH_4				
	31.6		1	NA	NA	NA
	3.6 - 8.9		4	4.5 -13.5	NA	1.4 - 5.1
	20.2 - 42.5		4	8.4 -10.10	NA	NA
	5.7	194.4	1	7.1	77	NA
	6.7 - 19.7	213 - 233	6	NA	NA	NA
	38.9 - 40.3	267.1 - 389.5	2	NA	NA	NA
	50 - 106		6	2.4 - 8.6	NA	0.5 - 3.2
	11.9 - 85	254.3 - 428.8	5	6.1- 22.4	NA	NA
	6.0 - 37.0		6	(-6.4) - 21.2	NA	NA
	23.8 - 27.1	271.2 - 312	4	1.0 - 7.0	78 - 93	5.0 - 7.0
	65.8		1	15.7	NA	NA
	19.44	259.2	1	5.9	75	NA
	87.67	432.2	1	NA	NA	NA
	69.6	348	1	NA	NA	NA
	53.6 - 79.1	204.0 - 395.0	4	(-0.6) - 30.3	49.4 -64.5	NA
	8.7 - 109.4	144.2 - 462.2	11	(-8.3) - 23.8	NA	2 - 5.4
	6.7 - 58.1		12	1 - 19	NA	0.8 - 2.0
	29.8 - 38.4	324.9 - 381.7	2	17	NA	1.5

Reference	Region	Floor type	Manure handling methods	Measurement Methods		
				Ventilation rate	Gas concentration	
Bleizgys et al., 2014	Lithuania	solid, slatted	scraped, NA	Dynamic flux chamber	gas analyzer	
Saha et al., 2014	Germany	solid	scraped	CO_2 balance method	Innova	
Ngwabie et al., 2014	Canada	solid	scraped	CO_2 balance method	Innova	
Joo et al., 2015a	America	solid	flushed	anemometers	Innova	
Joo et al., 2015b	America	solid	flushed	anemometers	Innova	
Cortus et al., 2015	America	solid	flushed, scraped	anemometers	gas analyzer	
Wang et al., 2016	America	solid	flushed	anemometers	Innova + spectrophotometry	
Rzeźnik et al., 2016	Poland	solid, slatted	scraped, NA	CO_2 balance method	Innova	
Schmithausen et al., 2018	Germany	slatted	flushed	CO_2 balance method	Innova	

[a] NA means not available.
[b] Innova belongs to photoacoustic spectroscopy category.
[c] N means number of measurements.

2.3 Results and discussion

2.3.1 Gas emission rates from dairy barns

Reported gas emission rates covered all sources within the building which implied emissions released by the animals and the emissions from manure management. As shown in Fig. 2.1, the NH_3 and CH_4 emission rates presented wide ranges with different frequency distributions. Reported NH_3 emission rates in dairy barns showed a non-normal distribution deflecting to lower values distributed between 4 to 30 g/AU/d. Average NH_3 emission rate in dairy housing was 36 g/AU/d with confidence interval between 30.5 g/AU/d and 41.5 g/AU/d. The mean NH_3 emission rate in this study was much lower than reported by Bougouin et al. (2016) with 60.1 g/head/d, mainly because there are much more measurements under cold conditions included in this study, which significantly lowers the mean. For CH_4 emission rates from dairy barns, a normal distribution was observed (Shapiro-Wilk test: W = 0.99, p = 0.77). The emission factors were centered around 200 to 400 g/AU/d with an average value of 279.4 g/AU/d (Fig. 2.1b). These values were in accordance with reported values by Chianese et al. (2009) who found CH_4 emission rates varying from 258 to 332 g/AU/d.

	Gas emissions, g/AU/d		N [c]	Temperature, °C	Relative humidity, %	Wind speed, m/s
	NH_3	CH_4				
	21.9 - 32.1		2	1.2 - 8	75.6 - 79.7	NA
	18.5 - 45.6	242.4 - 312.0	4	(-1.1) - 15.9	73.7 - 94.7	NA
	10.3 - 15.4	292.8 - 333.6	2	1.0 - 7.0	79 - 89	3.2 - 3.6
	14.9 - 36.7		12	3.6 - 26.9	NA	NA
		102.1 - 188.5	6	14.3 - 27.1	29 - 51	1.6 - 3.3
		215 - 331	4	9.2 - 20.3	66 - 71	NA
	20.6 - 21.2		3	29.5	36.8	1.78
	18.0 - 28.8	283.2 - 333.6	3	14.8 - 19.2	66.8 - 78.7	NA
	18.4 - 27.4	225.1 - 307.1	2	4.4 - 15.2	NA	2.2 - 2.0

The gas emission rates from dairy cow barns showed large variation among different studies. This could be further explained by the difference in manure management of dairy barns such as floor type, manure handling methods and ambient conditions of measurement period. In addition, the measurement techniques of ventilation rate and gas concentration used in commercial barns to estimate gas emission rates might lead to over- or underestimation. The high variability in reported gas emission rates from dairy barns highlighted that one estimated gas emission factor could not fit all commercial dairy cow barns due to the complexity of processes and conditions in dairy cow housing systems.

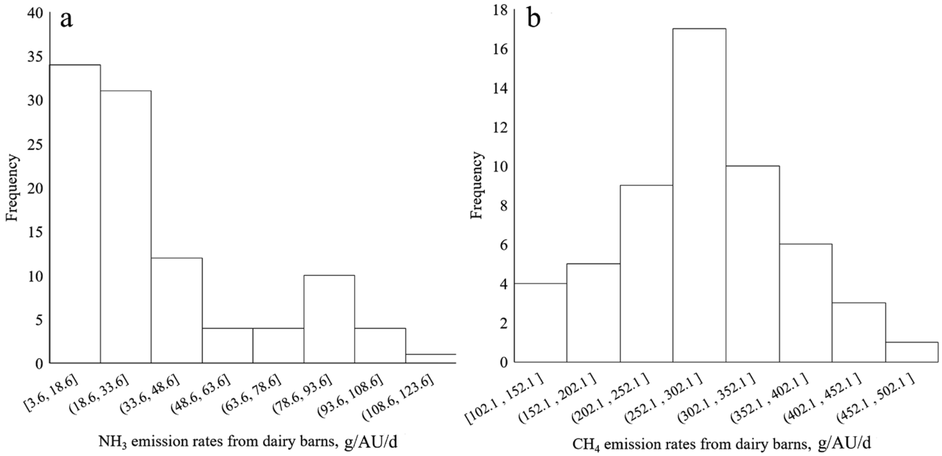

Figure 2.1. Reported (a) NH_3 and (b) CH_4 emission rates from dairy barns. AU refers to an animal unit with 500 kg live weight.

2.3.2 Factors affecting NH_3 emission rate

2.3.2.1 Floor type and manure handling methods

There is no significant difference in NH_3 emission rates between solid floor and slatted floor (Fig. 2.2), which is in line with previous results. Schiefler (2013) measured NH_3 emission rates in dairy barns with solid floor and slatted floor. Results showed no significant difference in mean NH_3 emission rates between solid and slatted floors. Meta-studies performed by Bougouin et al. (2016) and Poteko et al. (2019) also demonstrated that there was little difference in NH_3 emission rates between solid floor and slatted floor.

Differences in floor design (sloped or levelled) and surface (smooth or rough) played a larger role in modulating NH_3 emission rates than floor type (solid or slatted). It was reported that NH_3 emission rate from a non-sloping solid floor was almost equal or higher than slatted floor (Braam et al., 1997; Pereira et al., 2011). However, a solid floor with slope could significantly reduce NH_3 emission compared to a slatted floor (Swierstra et al., 1995; Monteny and Erisman, 1998). As observed by Zhang et al. (2005), a solid floor with a smooth surface, scraper and drain may reduce the NH_3 emission from dairy cattle buildings. The texture and porosity of the floor influences the contact area between urine and urease present in the feces, which affects the percentage of urea actually converted into NH_3 (Braam and Swierstra, 1999). Thus, besides floor type, other floor properties of design and surface will have to be considered to assess the influence of the barn floor on NH_3 emission rates in commercial dairy barns.

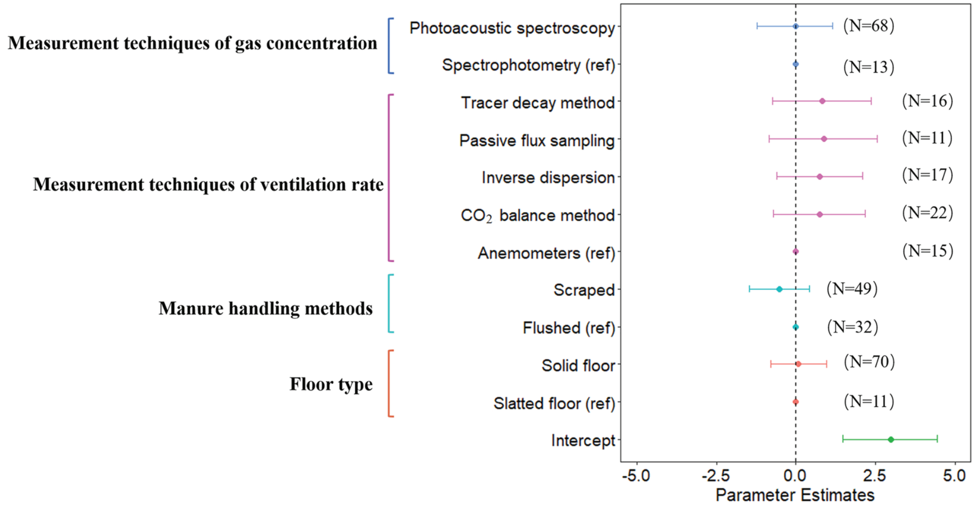

Figure 2.2. Parameter estimates and ±95% confidence intervals from linear mixed effect models for estimating NH_3 emission rates (g/AU/d) (after log-transformation) from dairy barns. Model 1: lmer (log (NH_3) ~ floor type + manure handling methods + measurement techniques of ventilation rate + measurement techniques of gas concentration + (1|Reference)). Explanatory variables are listed at far left with reference groups given (ref = reference). The reference level in each effect type is selected alphabetically and is set at zero. Positive estimates refer to increased emission and negative estimates refer to decreased emission compared to the reference. AU refers to an animal unit with 500 kg live weight.

In consistent with the results of Bougouin et al. (2016), we found that NH_3 emission rate was not significantly affected by manure handling methods (Fig. 2.2) when analyzed together with the other variables. However, previous studies showed that the flushing system could significantly reduce NH_3 emission rates when compared to the use of scrapers. For dairy barns with a solid floor, NH_3 emission rate from a flushing system was up to one order of magnitude lower compared to scrapers (Baldini et al., 2016). Scrapers usually leave a thin layer of slurry increasing the surface area onto which urine is spread and decreasing the thickness of urine pools, thus enhancing N volatilization (Moreira and Satter, 2006). Using flushing systems could dilute ammonia concentration of urine pools remaining on the floor, leading to the reduction of NH_3 emission rate (Ogink and Kroodsma, 1996). Despite that, in commercial dairy barns, NH_3 emission rates might be simultaneously affected by floor design and manure handling methods. Schiefler (2013) observed that the NH_3 emission rates were not significantly different between a barn equipped with solid floor and scraping system and a barn with slatted floor and flushing system. Therefore, more data are needed to investigate the interaction between floor design and manure handling methods to verify the effects of physical factors of dairy barns on NH_3 emission rates.

2.3.2.2 Measurement techniques of ventilation rate and gas concentration

Measurement methods of ventilation rate and gas concentration did not show significant effects on estimated NH_3 emission rates (Fig. 2.2). Although average NH_3 emission rate measured by anemometers was lower than other measurement techniques (Fig. 2.2, parameter estimates of other measurement methods were higher than 0), the statistical analysis revealed no significant difference ($p = 0.88$, Table A.4). A comparison between passive flux samplers and anemometers for mechanically ventilated buildings showed no significant differences in NH_3 emission estimates (Mosquera et al., 2003). Edouard et al. (2016) indicated similar ventilation rates measured by CO_2 balance method and SF_6-based tracer decay method. However, Samer et al. (2011) showed the estimated ventilation rate with the tracer gas technique (^{85}Kr) was about twice as high as with CO_2 balance method. Van Buggenhout et al. (2009) also demonstrated that, in a mechanically ventilated dairy barn, the errors in the ventilation rate using the decay method can rise to 86% of the actual ventilation rate. It is hard to draw any conclusion as to which method is more accurate and superior. Standardization of measurement methods of ventilation rate for dairy barns is highly needed.

There was also little difference between NH_3 emission rates measured by photoacoustic spectroscopes and spectrophotometry, which agreed with the result of Wang et al. (2016) that showed NH_3 emission rates based on photoacoustic spectroscopy and spectrophotometry were similar. Although there is no significant difference in NH_3 emission rates among different measurement techniques of ventilation rate and of gas concentration, it is crucial to define criteria for the choice and application of these different measurement methods, which would provide more reliable data for estimating gas emission rates (Jungbluth et al., 2001).

2.3.2.3 Environmental factors

Temperature had significant influence on NH_3 emission rates, with higher temperature leading to higher emission rates (Fig. 2.3a). The significant and positive relationship between NH_3 emission rate and ambient temperature in this study is consistent with other literatures (Wu et al., 2012; Rong et al., 2014; Poteko et al., 2019, Sanchis et al., 2019). The increase of NH_3 emission rate with temperature is related to the following two factors. First, the conversion from urea in the urine pool to NH_3 is determined by the enzyme urease that mainly exists in feces (Monteny et al., 2002). The urease activity increases exponentially with temperature between 10 and 40 °C (Sommer et al., 2006; Pereira et al., 2011). The enhanced urease activity at relative high temperature will result in more NH_3 release. Second, in the mixed urine and feces pool, NH_3 (unionized) and NH_4^+ (ionized) are in equilibrium (dissociation). When the temperature rises, the coefficient of Henry's equilibrium increases and the dissociation of NH_3 between the liquid and the gas phase will be enhanced.

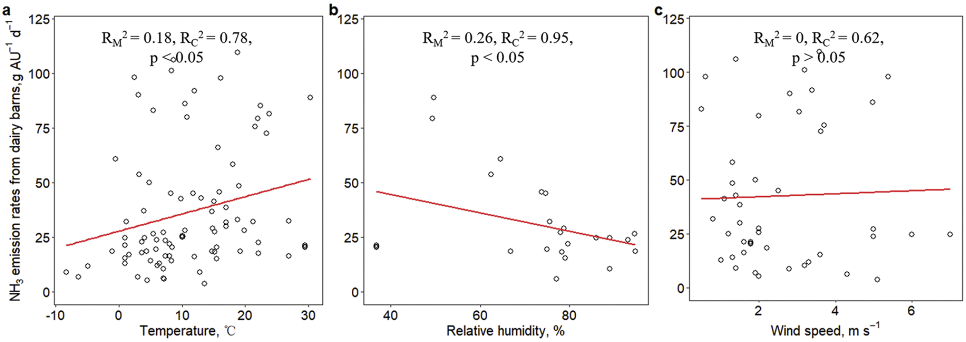

Figure 2.3. Relationships between NH_3 emission rates from dairy barns and ambient temperature (a), relative humidity (b) and wind speed (c). AU refers to an animal unit with 500 kg live weight. The marginal R-squared (R_M^2) means the variance of the fixed effects, while the conditional R-squared (R_C^2) takes both the fixed and random effects into account.

By contrast, a negative relationship was found between NH_3 emission rates and relative humidity (Fig. 2.3b), which agreed with Saha et al. (2014) who observed a negative relationship between NH_3 emission rates and relative humidity. The effect of relative humidity might be related to changes in animal activity in response to heat stress, expressed in the combined effect of temperature and relative humidity (Provolo and Riva, 2008). When both temperature and relative humidity are high, animals will become inactive and excrete less urine and feces, leading to lower NH_3 emission. Additionally, ammonia is water soluble. Higher relative humidity might lead to more NH_3 being dissolved in the moist air and less gaseous NH_3 being measured (Saha et al., 2014).

Wind speed did not significantly influence NH_3 emission rates (Fig. 2.3c). The result agreed with meta-studies conducted by Bougouin et al. (2016) and Sanchis et al. (2019). However, Wu et al. (2012) showed that wind speed had a strong positive effect on NH_3 emission rate. High wind speed could result in a high air exchange rate that plays a key role in determining the emission rates of aerial pollutants from animal buildings (Zhang et al., 2005; Rong et al., 2014). Besides, high air velocities in barns could accelerate the processes of NH_3 volatilization from liquid to gaseous phase in the surface of slurry (Sommer et al., 1991; Cortus et al., 2008). However, the influence of wind speed on NH_3 emission rate might be valid only when wind speed is within a certain range. As shown by Sommer et al. (1991), the ammonia loss rate increased when wind speed increased up to 2.5 m/s, but no consistent increase in ammonia volatilization was found when the wind speed increased from 2.5 to 4 m/s. According to Snoek, Stigter et al. (2014), the effect of wind speed on NH_3 emission depends on the height of wind speed measured. Wind speed measured at manure level (i.e., above the floor) might have positive effects on NH_3 emissions. But the averaged wind speed in the whole barn or outside the barn varies a lot and might have no significant influence on NH_3 emission rates.

2.3.3 Factors affecting CH_4 emission rate

2.3.3.1 Floor type and manure handling methods

There was no significant difference in CH_4 emission rates in dairy barns with solid floor and slatted floor (Fig. 2.4), which is in accordance with the result of meta-study conducted by Poteko et al. (2019). Schiefler (2013) also indicated no significant difference in CH_4 emission rates between solid floor and slatted floor. The effects of floor type on CH_4 emission rates from dairy barns depend on whether the manure is stored under slatted floor pit for a long time. As previous studies showed, CH_4 emission rate from slatted floor with under pit storage was higher than the emission from solid floor, since the reduced removal frequency of the slurry from the pit below the pavement and the long retention period of manure in barns allowed anaerobic fermentation processes to occur, generating CH_4 emissions (Sommer et al., 2007; Baldini et al., 2016). In this study, the retention time of manure under slatted floor varied from less than one day to a few months, which could explain the lack of significant effects on CH_4 emission rates. Therefore, the slurry management (e.g., under-pit storage or slurry homogenization) and floor type can together contribute to the difference of CH_4 emission rates between solid floor and slatted floor.

Manure handling methods did not affect CH_4 emission rates (Fig. 2.4). This is consistent with Cortus et al. (2015) who also reported that changing manure handling method from flushing to scraper did not affect the CH_4 emission rates. In addition, although the accumulated manure in the barn could contribute to greater CH_4 emission rate, the contribution of enteric fermentation represents the main source of methane in dairy barn (Sun et al., 2008). Thus, this might explain why CH_4 emission rates from the barn is not significantly influenced by floor type and manure handling methods.

2.3.3.2 Measurement techniques of ventilation rate

We observed a significant difference in CH_4 emission estimates using anemometers and CO_2 balance method or inverse dispersion (Fig. 2.4). The average CH_4 emission rate measured by anemometers was 129 g/AU/d, 58% lower than CH_4 emission rates measured by CO_2 balance method and inverse dispersion, while these two methods did not significantly differ in CH_4 emission rates (Fig. 2.4). There are few studies focusing on the effects of measurement methods on CH_4 emission rates, which prevents comparisons with published articles. This further highlights the importance of standardized measurement methods on estimating gas emission rates from dairy barns.

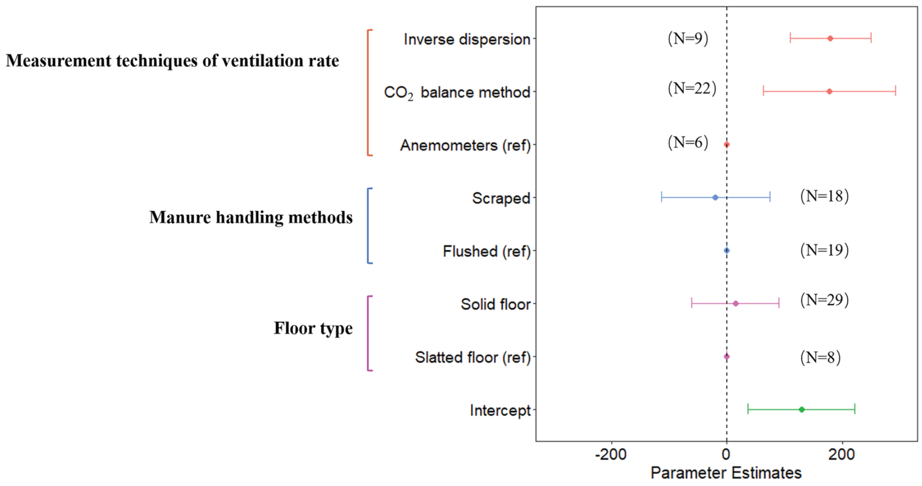

Figure 2.4. Parameter estimates and ±95% confidence intervals from linear mixed effect models for estimating CH_4 emission rates (g/AU/d) from dairy barns. Model: lmer (CH_4 ~ floor type + manure handling methods + measurement techniques of ventilation rate + measurement techniques of gas concentration + (1|Reference)). Explanatory variables are listed at far left with reference groups given (ref=reference). The reference level in each effect type is selected alphabetically and is set at zero. Positive estimates refer to increased emission and negative estimates refer to decreased emission compared to the reference. AU refers to an animal unit with 500 kg live weight.

2.3.3.3 Environmental factors

The emission rates of CH_4 had a tendency to increase with increasing temperature, ranging from -8.3 °C to 30.3 °C (Fig. 2.5a). This result is in line with the result of meta-study of Poketo et al. (2019) in which the range of temperature was -3.2 °C to 27.3 °C. The relationship between temperature and CH_4 emission strongly depends on the considered temperature interval (Hempel et al., 2020). Pereira et al. (2011) found that CH_4 emission increased significantly with temperature from 5 °C to 25 °C. However, Hempel et al. (2016) observed that CH_4 emission rates did not vary significantly at temperature range from 14.4 °C to 27.8 °C. Yadav et al. (2016) observed that the CH_4 emission declined with the increase in exposure temperature from 25 °C to 40 °C. This was because the exposure to high temperature reduced the time that cows dedicate to feeding and ruminating, which leads to the decrease of quantity in CH_4 produced (West, 2003; Ngwabie et al., 2011). In this study, average air temperature from selected studies ranged from -8.3 °C to 30.3 °C, most of which were centered between 0 °C to 25 °C and few data lower than 0 °C or higher than 30 °C (Fig. 2.5a). The large and realistic temperature range confirmed the positive relationship between CH_4 emission rates and temperature.

In comparison with temperature, there were few studies focusing on the effect of relative humidity and wind speed on CH_4 emission rates. There was no significant relationship between CH_4 emission rates and relative humidity (Fig. 2.5b). The effect of wind speed on CH_4 emission rate depends on the compensation between CH_4 concentration and ventilation rate (Fig. 2.5c). Increases in wind speed results in increased ventilation rates in naturally ventilated buildings but a decrease in CH_4 concentration (Joo et al., 2014; 2015).

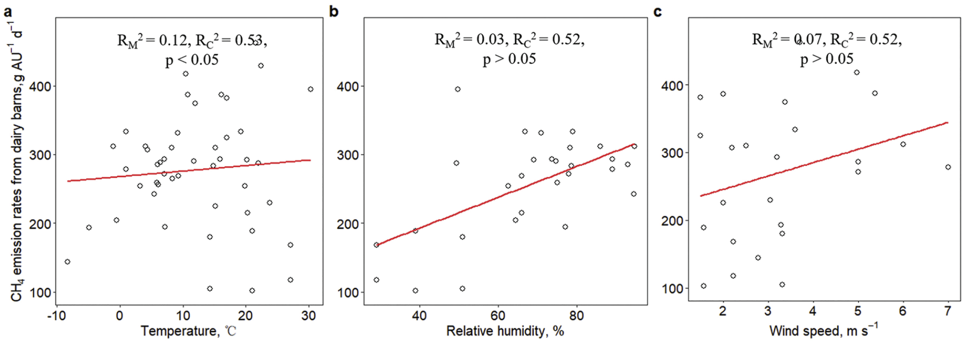

Figure 2.5. Relationships between CH_4 emission rates from dairy cow barns and ambient temperature (a), relative humidity (b) and wind speed (c). AU refers to an animal unit with 500 kg live weight. The marginal R-squared (R_M^2) means the variance of the fixed effects, while the conditional R-squared (R_C^2) takes both the fixed and random effects into account.

2.4 Conclusions

The data synthesis for NH_3 and CH_4 emission rates in dairy cow barns shows large variation across publications. This highlights that a single emission factor for all dairy farms barns is not realistic. In commercial barns, NH_3 emission is much more affected by environmental factors than housing factors and measurement methods, thus NH_3 emission factors should be defined for particular climate zones. A positive relationship between temperature and CH_4 emission rates was proven with a practical temperature range. Both NH_3 and CH_4 emissions presented no significant difference between solid floor and slatted floor, or between flushed and scraped systems. Observed emission rates differed more between measurement methods for CH_4 than for NH_3. Studies using CO_2 balance method or inverse dispersion tended to estimate higher CH_4 emission rates, while anemometers tended to estimate lower values. Standardization of measurement methods and reported results of continuous measurement are needed to reduce the large variability and uncertainty of estimating gaseous emissions from dairy buildings. These studies will be useful for enhancing our understanding of potential influencing factors of gas emissions from commercial dairy housing, improving the accuracy of gaseous emission inventories as well as developing and evaluating emission mitigation measures.

Supplementary materials

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biosystemseng.2021.02.012.

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Chapter 3

Effects of pH, total solids, temperature and storage duration on gas emissions from slurry storage: a systematic review

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Abstract

Gaseous emissions are the main loss pathways of nutrients during dairy slurry storage. In this study, we compiled published data on cumulative ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) emissions from dairy slurry storage and evaluated the integrated effects of slurry pH, total solids (TS), ambient temperature (T) and length of storage (LOS) on emissions using linear mixed effects models. Results showed that the average nitrogen (N) loss by NH_3 volatilization from slurry storage was 12.5% of total nitrogen (TN), while the loss by N_2O emissions only accounted for 0.05%−0.39% of slurry TN. The NH_3−N losses were highly related to slurry pH, lowering slurry pH leading to significant decrease of emissions. Temperature also affected NH_3−N losses, with higher losses from slurry storage under warm conditions than cold conditions. No significant relationship was observed between NH_3−N losses and slurry TS contents within a range from 21−169 g/kg. The losses of N_2O−N from dairy slurry storage were not significantly affected by slurry pH, TS contents and temperature. The carbon (C) loss as CH_4 emissions varied from 0.01%−17.2% of total carbon (TC). Emissions of CH_4−C presented a significant positive relationship with temperature, a negative relationship with slurry TS contents and no significant relationship with slurry pH ranging from 6.6−8.6. Length of storage (more than 30 days) had no significant influence on cumulative gas emissions from slurry storage. This study provides new emission factors of NH_3, N_2O and CH_4 in the percentage of TN or TC from dairy slurry storage. Our results indicate the potential interactive effects of slurry characteristics and storage conditions on gaseous emissions from slurry storage. Farm-scale measurements are needed to accurately estimate nutrient losses from liquid manure storage.

Keywords: ammonia; methane; nitrous oxide; liquid manure storage; manure characteristics; length of storage; nitrogen loss

3.1 Introduction

Liquid manure storage is known to represent an important source of ammonia (NH_3) and methane (CH_4) emissions (Chadwick et al., 2011; Smith et al., 2007). These gas emissions not only lower the fertilizer value of liquid manure but also pose a threat to environmental quality. It was reported by Oenema et al. (2007) that almost 30% of the excreted nitrogen (N) was lost during storage, approximately 19% of the excreted N via NH_3 emissions. The CH_4 emissions from stored dairy manure accounted for 8% to 15% of total CH_4 emissions from cows (Külling et al., 2002) and was estimated as the second largest source of CH_4 emissions (after enteric fermentation) on European dairy farming (Sneath et al., 2006). Due to high global warming potential, more attention has also been paid to nitrous oxide (N_2O) emissions from dairy slurry storage, since high fiber content in cow feces is more likely to form surface crust, creating intermittent aerobic and anaerobic environments for N_2O production (Petersen and Sommer, 2011).

A considerable number of studies have been conducted to measure NH_3, N_2O and CH_4 emissions from dairy slurry storage but the emission values were expressed in a wide variety of units. It was reported by Vigan et al. (2019) that the number of units used in literatures to report emission values from manure storage were 45, 31, 48 for NH_3, N_2O and CH_4 emissions, respectively. Standardization of the emission data is very important for comparing and compiling emission values from different studies, which could benefit emission inventories of livestock manure management systems.

A few recent studies on emissions from storage of liquid dairy manure provide flow-based gas emission factors in the percentage of total ammonium nitrogen (TAN) for NH_3 or N_2O or the percentage of volatile solids (VS) for CH_4 (Sommer et al., 2019; Kupper et al., 2020). These studies provide updated information for gas emission factors and contribute to more accurate national or regional emission inventories. However, few studies compiled these gaseous losses from dairy slurry storage in the percentage of total nitrogen or total carbon (% of TN or TC) that can be used to estimate carbon and nitrogen flows along manure management chains and to evaluate nutrient use efficiency in dairy farming systems. Therefore, one of the objectives of this paper is to collect measured NH_3–N, N_2O–N and CH_4–C losses during storage of liquid dairy manure and to update information on gaseous losses in the percentage of TN or TC.

Besides the units used to express gas emission factors, the magnitude of gas emissions from liquid manure storage varied from negligible to one of the largest sources in dairy farms highly depending on manure characteristics, environmental conditions, and storage management practices. The independent influence of slurry characteristics, environmental conditions, and management practices on gas emissions from liquid manure storage has been investigated. For example, van der Weerden et al. (2014a; 2014b) demonstrated inverse logarithmic relationships between slurry total solids (TS) contents and NH_3 and CH_4 emissions. Sawamoto et al. (2016) revealed that the effects of temperature and length of storage on CH_4 emissions from dairy slurry storage. Misselbrook et al. (2016) and Sommer et al. (2017) found lowering slurry pH by acidification can reduce emissions of NH_3 and CH_4. However, the integrated and interactive effects of these variables on gas emissions were rarely investigated due to a large effort in labor and costs. Based on accompanying parameters (e.g., slurry pH, TS contents, TN contents, TC contents, temperature etc.) in published studies measuring gas emissions from dairy slurry storage, a more advanced analysis on integrated effects of these variables on flow-based gaseous losses can help to identify the interactive effects on gas emissions and contribute to improving methods for calculating gas emission factors from liquid dairy manure storage.

Overall, the aims of this review are: (i) to provide quantitative information on flow-based losses of NH_3–N, N_2O–N and CH_4–C during storage of liquid dairy manure; (ii) to assess the integrated effects of slurry characteristics, temperature, and storage length on gaseous losses.

3.2 Materials and methods

3.2.1 Data source

We searched published literatures in English using the electronic databases: CAB Abstracts via Ovid, Web of Science and Scopus, by combining specific keywords covering cow, slurry or liquid manure, storage, and gas emissions (e.g., NH_3, N_2O, and CH_4). Articles were selected on the basis of following criteria: (i) the research focused on gas emissions from storage of liquid dairy manure, including raw slurry, liquid slurry after separation, anaerobic digested slurry and anaerobic digested slurry after separation; emissions from other livestock manure storage (e.g., pig slurry) were not included. (ii) the emission values were measured at laboratory, pilot or farm scales; data from published reviews or models were excluded. (iii) storage tanks did not have any physical barriers (e.g., covers or roofs), but tanks with natural crusts were included; (iv) the study reported at least one of the NH_3, N_2O and CH_4 gases; (v) cumulative gas emissions were reported; (vi) the storage period should be not less than 30 days; (vii) the reported various units of gas emissions from articles could be converted into an uniform unit in the percentage of initial TN for NH_3–N and N_2O–N losses and in the percentage of initial TC for CH_4–C loss. The unit conversion of emission data is shown in Supplemental Table S1. Data that were only graphically presented were digitized using the software WebPlotDigitizer (Burda et al., 2017).

3.2.2 Data extraction

A total of 12 publications were selected to extract data. Detailed information of selected literatures is given in Table 3.1. Gas emissions, including NH_3–N, N_2O–N and CH_4–C, from dairy slurry storage in selected literatures were collected. Potential variables influencing gas emissions were also recorded, including initial slurry pH, the TS content, TN content, TC content, average air temperature (T) during storage and length of storage (LOS). For studies without reporting air temperature, we extracted slurry temperature as a surrogate for air temperature (Rodhe et al., 2009).

3.2.3 Data analysis

Based on measurements that simultaneously reporting gas emissions, slurry characteristics, air temperature, and storage duration, we investigated the relationships among gaseous losses of N or C and potential influencing factors by linear mixed effects models using *lme4* package in R language (Bates et al., 2015). We firstly used *cor()* function to check correlations between influencing variables. Variables whose pairwise correlation coefficients were greater than 0.7 were not included simultaneously in one model. In this way, the slurry pH, TS content, T and LOS were selected as main explanatory variables of NH_3–N, N_2O–N and CH_4–C losses because of collinearity (correlation coefficients > 0.7) existing among TS content, TN content and TC content (Supplemental Tables S2–S4). Selected explanatory variables were as main fixed effects and literatures were considered as random effects in fitted linear mixed effects models (Table 3.2). Outliers that had significant effects on fitted results were removed using function *romr.fnc ()* from package *LMERConvenienceFunctions* (Tremblay and Ransijn, 2015). The number of extracted outliers were 5, 0 and 3 for datasets of NH_3–N, N_2O–N and CH_4–C, respectively (Table 3.2). Relationships between gas emissions and influencing factors were visualized using *visreg ()* function based on the results of linear mixed effects models. All statistical analyses were performed using R statistical software (version 3.6.1) (R core Team, 2017).

Table 3.2. The composition of fitted models to investigate the effects of slurry pH, TS content, T and LOS on gas emissions from dairy slurry storage.

Gas	Linear mixed effects models	Number of measurements before extracting outliers	Number of measurements after extracting outliers
NH_3–N	lmer (NH_3–N ~ pH + TS + T + LOS + (1|Reference))	34	29
N_2O–N	lmer (N_2O–N ~ pH + TS + T + LOS + (1|Reference))	24	21
CH_4–C	lmer (CH_4–C ~ pH + TS + T + LOS + (1|Reference))	24	24

Table 3.1. Detailed information of selected 12 articles for extracting data.

Reference	Country	Scale [1]	Number of measurements	LOS [2]	T [3]	pH
				days	°C	
Kröber et al., 2000	Switzerland	lab	3	49	20.0	7.0–8.3
Fangueiro et al., 2008	Portugal	lab	2	48	8.0	7.9
Dinuccio et al., 2011	Italy	lab	2	30	5.0	7.1
van der Weerden et al., 2014	New Zealand	lab	9	84–197	12.0–18.0	8.1–8.4
Wood et al., 2014	Canada	pilot	2	155	15.0	7.5
Rodhe et al., 2015	Sweden	pilot	4	90	1.0–16.0	7.2–7.9
Regueiro et al., 2016	Portugal	lab	6	60	15.0	5.5–7.2
Le Riche et al., 2016	Canada	pilot	6	173	18.0	7–7.4
Le Riche et al., 2017	Canada	pilot	2	207	18.0	6.6–7.1
Holly et al., 2017	America	pilot	5	182	9.5	6.6–7.9
Baral et al., 2018	Denmark	pilot	2	78–309	6.5–17.0	NA
Maldaner et al., 2018	Canada	farm	2	365	6.4–6.6	7.3–8.0

[1] lab: laboratory scale. [2] LOS: length of storage. [3] T: average air temperature during storage. [4] TS: total solids. [5] TN: total nitrogen. [6] TC: total carbon. [7] NH_3–N: ammonia–nitrogen. [8] N_2O–N: nitrous oxide–nitrogen. [9] CH_4–C: methane–carbon. [10] NA: not available.[1] lab: laboratory scale. [2] LOS: length of storage. [3] T: average air temperature during storage. [4] TS: total solids. [5] TN: total nitrogen. [6] TC: total carbon. [7] NH_3–N: ammonia–nitrogen. [8] N_2O–N: nitrous oxide–nitrogen. [9] CH_4–C: methane–carbon. [10] NA: not available.

3.3 Results and discussion

3.3.1 Gas emissions from dairy slurry storage

A total of 39 measurements on NH_3–N losses from dairy slurry storage, 28 measurements on N_2O–N losses and 25 measurements on CH_4–C losses were collected from selected publications. Descriptive statistics for reported NH_3–N, N_2O–N and CH_4–C emissions from studies conducted at laboratory, pilot and farm scales are shown in Table 3.3. Studies on quantifying flow–based gas emissions from dairy slurry storage were mainly conducted at laboratory and pilot scales, with few measurements at farm scale. The minimum and maximum gas emission values differ by one to several orders of magnitude for all gases.

initial TS [4]	initial TN [5]	initial TC [6]	NH_3–N [7]	N_2O–N [8]	CH_4–C [9]
g/kg			% of initial TN		% of initial TC
NA [10]	2.2–4.0	NA	10.7–23.7	NA	NA
45–63	2.5–2.6	19.1–27.6	0.8–1.3	0.01–0.02	1.5–2.0
51–75	3.3–3.6	19.8–31.1	4.2–5.8	0.002–0.003	0.6–0.9
76–169	4.9–5.6	29.8–71.2	14.0–47.7	0.01	0.1–0.9
117–142	3.7–3.8	50.0–62.0	0.9–3.0	0.186–0.214	NA
33–79	1.9–3.2	15.0–35.6	NA	0.00001	0.0087–3.4
47–63	1.4–2.5	NA	2.7–17.4	NA	NA
46–163	1.4–3.4	NA	8.7–19.5	NA	NA
39-66	1.6–2.4	NA	8.8–16.0	0.057–0.300	NA
21-61	1.8–2.3	11.0–23.0	3.92–9.17	0.038–0.069	0.4–1.4
58-76	2.8–3.2	21.8	0.09–2.6	0.0025–0.39	17.2
71-92	NA	24.1–33.0	NA	NA	4.0–12.8

Average NH_3–N loss from dairy slurry storage was 12.5% of initial TN with a higher emission from laboratory scale studies than from pilot studies. The emissions of N_2O–N from slurry storage were relatively small with an average value of 0.05% of TN. Laboratory-scale studies reported smaller N_2O–N emissions than pilot-scale studies. Average CH_4–C emission from slurry storage was about 2.04% of initial TC. Larger CH_4–C emissions from farm-scale studies were observed than pilot and laboratory studies.

The NH_3–N loss is the main loss pathway of N during slurry storage, significantly larger than N_2O–N loss. Sommer et al. (2006) estimated that the NH_3–N losses from slurry storage varied from 6% to 30% of TN in stored slurry, which is in the range of reported NH_3–N losses in this study. The default value recommended by the Intergovernmental Panel on Climate Change (2019) for NH_3 and NO_X emissions is 30% of TN for dairy slurry storage with natural crusts and 48% of TN for slurry storage without natural crusts. Vigan et al. (2019) reported that the NH_3–N losses from dairy slurry storage ranged from 3.6% – 43.2% of TN stored with an average value of 28.8%. These values are considerably larger than the mean NH_3–N emission factor found in this study. The large variability of NH_3–N emission factors in different publications indicates that emissions are affected by multiple factors and one simple estimated emission factors might not fit all slurry storage situations.

Table 3.3. Reported NH_3–N, N_2O–N and CH_4–C emissions from dairy slurry storage from studies conducted at laboratory, pilot and farm scales.

Gas emissions	Scale	Number of measurements	Mean	Standard deviation	Median	Minimum	Maximum
NH_3–N, % of TN	All	39	12.50	10.24	9.46	0.09	47.7
	Laboratory	22	15.59	11.73	15.45	0.78	47.7
	Pilot	17	8.51	6.18	8.68	0.09	19.5
N_2O–N, % of TN	All	28	0.05	0.10	0.01	0.00	0.39
	Laboratory	13	0.01	0.00	0.01	0.00	0.02
	Pilot	15	0.09	0.12	0.06	0.00	0.39
CH_4–C, % of TC	All	25	2.04	4.07	0.64	0.01	17.20
	Laboratory	13	0.68	0.56	0.61	0.10	2.01
	Pilot	10	2.53	5.24	0.63	0.01	17.20
	Farm	2	8.40	6.26	8.40	3.97	12.83

The N_2O emissions mainly occur in slurry storage with natural crusts and the amount is very small (Jayasundara et al., 2016). IPCC (2019) estimated that the average N_2O–N loss from slurry storage with natural crusts is 0.5% of TN stored, with an uncertainty range of 0.25% – 1% of TN, and there is no N_2O–N loss from slurry storage without natural crusts. The range of N_2O–N losses in this study (0% – 0.39% of TN) basically corresponds to the default values of N_2O–N emission factors in IPCC (2019). But the average N_2O–N emission factor is only one tenth of the default value of IPCC (2019). Vigan et al. (2019) and Kupper et al. (2020) reported the average N_2O–N loss from cattle slurry storage in published articles with 0.2% and 0.13% of TN, respectively, which is also higher than the mean value of this study. The distribution of reported N_2O–N losses in this study is left skewed by low values, with the maximum value five times higher than the median. Among selected studies, most studies reported the N_2O–N losses less than 0.1% of TN and only three studies reported the values higher than 0.1% of TN (Wood et al., 2014; Le Riche et al., 2017; Baral et al., 2018).

Methane emissions from slurry storage are highly related to volatile solids content of slurry (Sommer et al., 2004). Most publications reported CH_4 emission factors based on g CH_4 per kg volatile solids (Holly et al., 2017a; 2017b; Sawamoto et al., 2016). Studies on carbon flow based CH_4 emissions are limited, which prevents us from comparing the results with other studies.

3.3.2 Effects of slurry pH, TS content, temperature and storage duration on NH_3–N emissions

A total of 29 measurements from 7 publications were used to analyze relationships among NH_3–N emissions and slurry pH, TS content, ambient temperature, and length of storage (Table 3.4). Results showed that these influencing factors (i.e., fixed effects) explained about 51% of variance (marginal R^2) and 49% of variance was caused by selected publications (i.e., references) and residual error, of which variance of references accounting for 52%. Among references, measurements of NH_3–N emissions from studies of Dinuccio et al. (2011) and van der Weerden et al. (2014a) were relatively higher than measurements from studies of Fangueiro et al. (2008) and Le Riche et al. (2016) (Supplemental Figure S1).

Table 3.4. Parameter estimates of linear mixed effects model for NH_3–N emissions from slurry storage (lmer (NH_3–N~ pH + TS + T + LOS + (1|Reference))).

Predictors	Estimates	Confidence interval	p value
Intercept	-39.72	-68.68 – -10.76	0.015**
pH	4.57	0.91 – 8.24	0.023**
TS	0.00	-0.06 – 0.07	0.898
T	1.35	0.24 – 2.47	0.076*
LOS	-0.00	-0.07 – 0.06	0.905
Random effects [1]			
σ^2		18.53	
$\tau_{00\ Ref}$		19.73	
ICC		0.52	
Number of references		7	
Number of measurements		29	
Marginal R^2 / Conditional R^2		0.51 / 0.76	

[1] σ^2 means variance of residual error; $\tau_{00\ Ref}$ means variance explained by random effects of references; ICC means interclass correlation coefficient, indicating the proportion of random effects variance in the total error variance (the sum of random effects variance and residual error variance); The marginal R^2 means the variance of the fixed effects, while the conditional R^2 takes both the fixed and random effects into account. *Significant at the 0.1 probability level. **Significant at the 0.05 probability level. ***Significant at the 0.01 probability level.

3.3.2.1 Slurry pH

A significant positive relationship between slurry pH and NH_3–N emissions from slurry storage was observed (Fig. 3.1a). This result is in accordance with previous studies that demonstrated that lowering dairy slurry pH by acidification could effectively reduce NH_3–N emissions, with mitigation efficiencies varying from 20% to more than 90% (Misselbrook et al., 2016; Sommer et al., 2017; Regueiro et al., 2016). Ammonia emissions depend on the dynamic equilibrium between ammonium (NH_4^+) and NH_3 in aqueous systems, which is highly determined by slurry pH. When slurry pH is higher than 7, the release of NH_3 takes place, and at pH values above 11, all of the ammoniacal nitrogen is in the form of free NH_3 (Neerackal et al., 2017). When the pH is less than 7, NH_3 volatilization decreases, and all of the ammoniacal nitrogen is in the form of non-volatile NH_4^+ when pH values near to 6 (Neerackal et al., 2017; Vaddella et al., 2011).

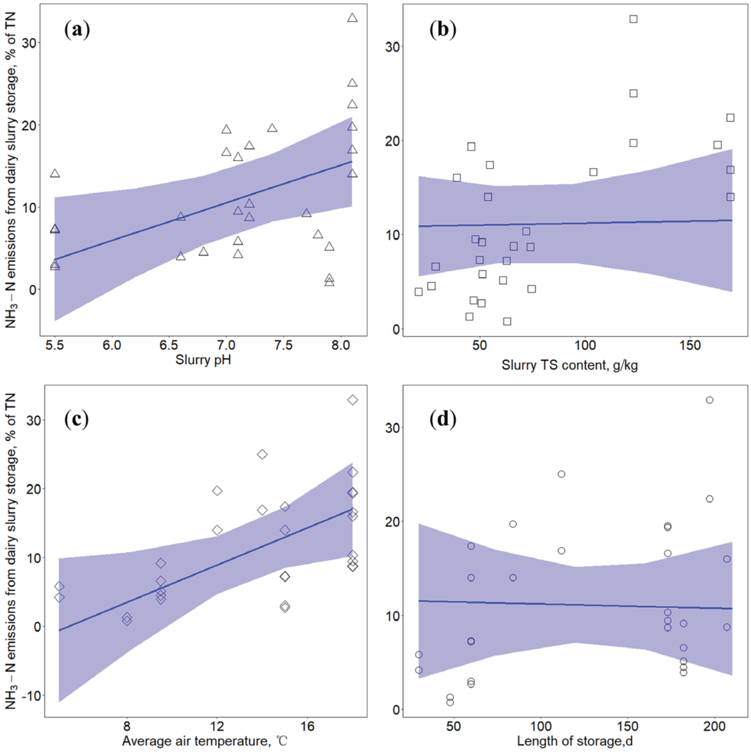

Figure 3.1. Relationships between NH_3–N emissions from dairy slurry storage with (a) slurry pH (in triangle), (b) total solids (TS) content (in square), (c) air temperature (in diamond) and (d) length of storage (in circle). Blue lines indicate prediction lines based on results of linear mixed effects model: lmer (NH_3–N ~ pH + TS + T + LOS + (1|Reference)). Blue shadows represent model predicted results with 95% confidence interval.

3.3.2.2 Total solids

Results of linear mixed effects model showed that when slurry TS contents varied from 21 to 169 g kg^{-1}, there was no significant relationship between NH_3–N emissions and the TS contents (Fig. 3.1b). The study of Wood et al. (2012) reported that NH_3 emissions increased linearly with slurry TS contents varying from 3 to 95 g/kg. van der Weerden et al. (2014a) found the inverse logarithmic relationships between cumulative NH_3 emissions versus the TS contents ranging from 76 to 399 g/kg. These results from different studies indicated that the relationship between NH_3–N emissions and slurry TS contents highly depends on the investigated range of TS contents. Slurry TS content is an indicator of substrate availability for NH_3 emissions and is related to crust formation (Wood et al., 2012). When the TS content of slurry is low, the availability of mineralized NH_4^+ from organic N is small, which lead to the reduced potential for reactive N to be lost (Sommer et al., 2006; Wood et al., 2012). When the TS content increases, the available N content increases, but high TS content will contribute to the formation of natural crust, acting as a physical barrier against gas exchange (van der Weerden et al., 2014b; Olesen et al., 1993). The trade-off between availability of substrate N for NH_3–N production and the crust formation in regulating gas emissions from slurry storage finally influences the relationship between NH_3–N emissions and slurry TS contents.

3.3.2.3 Temperature

Temperature showed a positive relationship with NH_3–N emissions from slurry storage (Fig. 3.1c), but the relationship was only significant at 0.1 significance level. Previous study also demonstrated that the N loss from slurry storage in summer (average air temperature 21.4 °C) was much greater than storage in winter (average air temperature 6.5 °C) (Perazzolo et al., 2017). Ammonia release from slurry storage tank is a combination of diffusion and convective movements over the slurry surface, which could be parameterized by the dissociation constant and the overall mass transfer coefficient (Sommer et al., 2006; Ni et al., 1999; Koirala et al., 2014). Increased liquid temperature enhances ionic activity, which ultimately increases the dissociation of ammonium ions (Koirala et al., 2014; Vaddella et al., 2013; Montes et al., 2009). Most studies reported ambient temperature during measurement periods instead of slurry temperature, which might lead to significant relationship between temperature and NH_3–N emissions only at 0.1 significance level. Also, the interactions between temperature and other potential influencing factors might complicate the relationship between temperature and NH_3–N emissions.

3.3.2.4 Length of storage

Length of storage did not significantly influence NH_3–N emissions from slurry storage (Fig. 3.1d). The storage duration of all selected measurements in this study are longer than 30 days. Ammonia volatilization losses from manure are most intensive at the initial stage of storage (Lee et al., 2011) and decrease with storage time (Külling et al., 2003). This agrees with rapid hydrolysis of urea that is generally considered to be the major source of NH_3 emissions (Bussink et al., 1998). It was reported that during the initial 10 days of manure storage, the loss of urinary–N was significant, accounting for 90% of NH_3–N loss, while the contribution of fecal–N was relatively low (Lee et al., 2011). Therefore, extending storage duration did not have significant effect on NH_3–N losses. In order to reduce nitrogen loss from manure storage, mitigation practices should be used and workable at the early stage of storage.

3.3.3 Effects of slurry pH, TS content, temperature and storage duration on N_2O-N emissions

Results of linear mixed effects model showed that the N_2O–N emissions from slurry storage were not significantly affected by slurry pH, TS content, temperature, and storage duration (Table 3.5). Errors of estimates of model were mainly from reported literatures (marginal R^2 = 0.00 and conditional R^2 = 0.99 in Table 3.5). The minimum N_2O–N emission was reported by Rodhe et al. (2015), who measured the N_2O–N losses from stored undigested and digested dairy slurry. The low TS contents of slurry (33–79 g/kg) did not benefit the formation of surface crusts, resulting in negligible N_2O emissions. Wood et al. (2014) measured the maximum N_2O–N emission from slurry storage with a value of 0.21% of TN. The TS content of the stored slurry was 117 g/kg, leading to the formation of natural crust on the slurry surface. Besides, the high air temperature (15 °C) mutually contributed to the persisting dry natural crust, resulting in the higher N_2O–N emissions. This also highlights the potential interactive effects of variables on N_2O–N emissions from slurry storage.

Table 3.5. Parameter estimates of linear mixed effects model for N_2O–N emissions from slurry storage (lmer (N_2O–N ~ pH + TS + T + LOS + (1|Reference))).

Predictors	Estimates	Confidence interval	p value
Intercept	0.06	-0.05 – 0.18	0.298
pH	-0.00	-0.01 – 0.01	0.732
TS	0.00	-0.00 – 0.00	0.933
T	-0.00	-0.00 – 0.00	0.979
LOS	0.00	-0.00 – 0.00	0.901
Random effects [1]			
σ^2		0.00	
$\tau_{00\ Ref}$		0.01	
ICC		0.99	
Number of references		6	
Number of measurements		24	
Marginal R^2 / Conditional R^2		0.00 / 0.99	

[1] σ^2 means variance of residual error; $\tau_{00\ Ref}$ means variance explained by random effects of references; ICC means interclass correlation coefficient, indicating the proportion of random effects variance in the total error variance (the sum of random effects variance and residual error variance); The marginal R^2 means the variance of the fixed effects, while the conditional R^2 takes both the fixed and random effects into account. *Significant at the 0.1 probability level. **Significant at the 0.05 probability level. ***Significant at the 0.01 probability level.

3.3.3.1 Slurry pH

Due to the limited N_2O emissions from slurry storage, the effect of slurry pH on N_2O emissions was less investigated. The insignificant relationship between N_2O–N emissions and slurry pH in this study agrees with the result of Sommer et al. (2017) who presented no significant effect of lowering slurry pH on N_2O emissions.

3.3.3.2 Total solids

Previous studies demonstrated that the slurry TS content is an important indicator for N_2O emissions. High TS contents benefited the formation of thick surface crusts, which created intermittent environment of aerobic and anaerobic and promoted nitrification and denitrification, producing N_2O gas (Schmithausen et al., 2018). However, there is little knowledge about the extent of N_2O emissions from slurry crusts. Hansen et al. (2009) found that the dry organic crusts had higher potential for emitting more N_2O gas, because there was deeper oxygen penetration in dry crusts than in wet crusts, which promoted dissolved NH_4^+ to be oxidized to the nitrogen oxides (NO_2^- and NO_3^-), leading to higher emissions of N_2O under anoxic conditions. Besides, the thickness of natural crusts influences the amount of N_2O production. The N_2O emissions from young and thin natural crusts were usually lower than old and thick natural crusts (Nielsen et al., 2010). This was because in well-developed and thick natural crusts, there was high O_2 penetration due to the relatively long distance from the surface to the free slurry interface (Nielsen et al., 2010). The effect of TS contents on N_2O–N emissions is complexed by the formation and characteristics of surface crusts, which resulted in the insignificant relationship (Fig. 3.2b).

3.3.3.3 Temperature

We did not observe a significant effect of ambient temperature on N_2O–N emissions (Fig. 3.2c). Sommer et al. (2000) also found that there was no relation between N_2O emissions and air or slurry temperature. Pereira et al. (2012) compared the N_2O emissions from cattle slurry at 5, 15, 25 and 35 °C, finding the amount of N_2O released did not vary significantly with temperature. However, Petersen et al. (2013) observed a significant increase of N_2O emissions from pig slurry storage with natural crusts when temperature was higher than 10 °C. Temperature might be not the only one influencing factor for N_2O–N emissions. As previous discussed, the intermittent environment of aerobic and anaerobic in surface crusts is an essential prerequisite for N_2O emissions from slurry storage. At a high storage temperature, a crust might more easily form, thereby creating conditions for N_2O production (Wang et al., 2016). But the high temperature will also promote NH_3–N emissions and reduce the availability of NH_4^+ for nitrification, consequently resulting in smaller N_2O–N emissions (Pereira et al., 2012). Therefore, the relationship between N_2O emissions and slurry storage temperature remains unclear and need to be investigated further.

3.3.3.4 Length of storage

Storage duration did not directly influence N_2O–N emissions from slurry storage (Fig. 3.2d). van der Weerden et al. (2014b) found an increasing tendency of N_2O emissions from slurry storage when storage length extended from 3 months to 7 months. It was speculated that the increase of N_2O emissions was because, during the late stage of storage, the mineralization of organic matter provided more substrate (i.e., NH_4^+, NO_3^-, NO_2^-) for nitrification and denitrification. The extending storage length also benefited the formation of thick and dry surface crusts, enhancing N_2O emissions. Previous studies observed that the N_2O emissions started after several weeks of the formation of slurry crusts (Wood et al., 2012; Sommer et al., 2000). It is difficult to come into a coherent conclusion based on limited data in this study. There is a need to measure N_2O emissions at farm scale, and to better understand relationships among gas emissions and slurry characteristics and storage conditions.

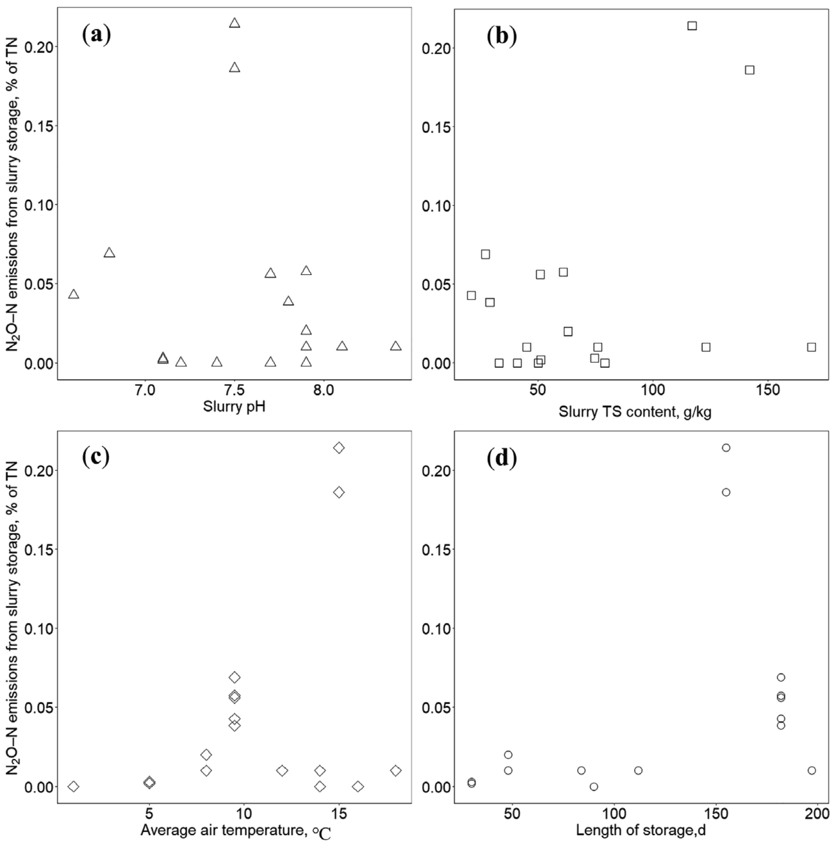

Figure 3.2. Relationships between N_2O–N emissions from slurry storage with (a) slurry pH (in triangle), (b) total solids (TS) content (in square), (c) air temperature (in diamond) and (d) length of storage (in circle).

3.3.4 Effects of slurry pH, TS content, temperature and storage duration on CH_4-C emissions

There were 21 measurements of CH_4–C emissions from 5 publications used for linear mixed effects model after removing three outliers from the dataset (Table 3.6). Results showed that all influencing factors explained 41.4% of variance (marginal R^2) and variance caused by references accounted for 20% of total error variance (Table 3.6).

3.3.4.1 Slurry pH

No significant relationship between slurry pH and CH_4–C losses from slurry storage was observed (Fig. 3.3a). The optimal pH of methanogen is near 7.0 and the CH_4 gas will be emitted in the range of pH values from 6.6 to 7.6 (United States Environmental Protection Agency (USEPA), 1994). Previous studies showed that lowering slurry pH by acidification could reduce CH_4–C losses, with mitigation efficiencies from 63% to 90% (Misselbrook et al., 2016; Sommer et al., 2017; Habtewold et al., 2018). Acidification could inhibit the growth and activity of methanogen, thereby leading to the reduction of CH_4 production (Habtewold et al., 2018). In this study, the range of slurry pH was 6.6–8.6, the small range of pH resulting in no significant changes in CH_4–C emissions from slurry storage.

Table 3.6. Parameter estimates of linear mixed effects model for CH_4–C emissions from slurry storage (lmer (CH_4–C ~ pH + TS + T + LOS + (1|Reference))).

Predictors	Estimates	Confidence interval	p value
Intercept	2.53	-2.29 – 7.36	0.326
pH	-0.21	-0.87 – 0.44	0.530
TS	-0.01	-0.02 – -0.00	0.025**
T	0.15	0.08 – 0.23	0.002***
LOS	-0.01	-0.01 – -0.00	0.093*
Random effects [1]			
σ^2		0.33	
$\tau_{00\ Ref}$		0.07	
ICC		0.19	
Number of references		5	
Number of measurements		21	
Marginal R^2 / Conditional R^2		0.41 / 0.52	

[1] σ^2 means variance of residual error; $\tau_{00\ Ref}$ means variance explained by random effects of references; ICC means interclass correlation coefficient, indicating the proportion of random effects variance in the total error variance (the sum of random effects variance and residual error variance); The marginal R^2 means the variance of the fixed effects, while the conditional R^2 takes both the fixed and random effects into account. *Significant at the 0.1 probability level. **Significant at the 0.05 probability level. ***Significant at the 0.01 probability level.

3.3.4.2 Total solids

A negative effect of TS contents on CH_4–C losses from slurry storage was observed (Fig. 3.3b), which agrees with the study of van der Weerden et al. (2014a) that observed an inverse logarithmic relationship between CH_4–C emissions and slurry TS contents. The liquid-based systems with low TS contents could promote an oxygen-free environment and anaerobic decomposition. Moreover, high water content is required for methanogen bacteria cell production and metabolism, thereby leading to larger CH_4–C emissions (USEPA, 1994). The increase of TS content is likely to increase the aeration status, which directly inhibits the activity of methanogen (Yamulki et al., 2006; Rotz, 2018). Besides, the increase of TS content benefits the formation of natural crust, in which CH_4 oxidation may occur due to the presence of methanotrophic bacteria (Ambus and Petersen, 2005; Petersen et al., 2005), thus leading to reduced CH_4–C emissions.

3.3.4.3 Temperature

Temperature had a significant influence on CH_4–C emissions from slurry storage (Fig. 3.3c), with increasing temperature leading to the increase of CH_4–C emissions. Previous studies demonstrated that temperature was the primary and dominant factor for CH_4 production (Sawamoto et al., 2016). High temperature could lead to high degradation rate of volatile solids and high CH_4 production (Baral et al., 2018; Sommer et al., 2004). Seasonal variations of CH_4 emissions further demonstrated the effect of temperature on CH_4 emissions. Jayasundara et al. (2016) and Cardenas et al. (2021) presented that the CH_4 emissions from liquid manure stored in warm seasons were considerably higher than in cold seasons. Temperature could influence the microbial activity and community development in the manure (Rennie et al., 2018). Low ambient temperature prevents the start of the methanogenesis process that optimally takes place at approximately 20 °C (Elsgaard et al., 2016; Huste, 1994; Sommer et al., 2006). Decreasing temperature might result in the abundance of psychrophilic methanogens and the decrease of mesophilic methanogens that have a higher CH_4 production rate than psychrophilic methanogens (Im et al., 2020), thereby affecting the total amount of CH_4 gas emitted.

3.3.4.4 Length of storage

It is well known that storage duration of liquid manure plays a decisive role in the production of CH_4 (Sommer et al., 2004). Previous studies demonstrated that extending slurry storage time could significantly increase the CH_4 emissions (Külling et al., 2002; van der Weerden et al., 2014b). This is because that the long storge duration could contribute to the establishment of sufficient methanogenic population (Wood et al., 2012), thereby leading to more CH_4 gas. However, we did not observe the significant influence of length of storage on CH_4–C losses from slurry storage at 0.05 significant level (Table 3.6 and Fig. 3.4d). In practical conditions, the CH_4 emissions might be complexed by multiple factors, the influence of which could be compensated. Temperature might be a more decisive factor in determining CH_4–C emissions from slurry storage than storage duration. Cardenas et al. (2021) presented that when the temperature was above 15 °C, even a short storage period could result in the emission of substantial amounts of CH_4 gas, while longer storage period under cold winter conditions emitted little CH_4 gas. These findings can be useful for designing CH_4 mitigation strategies, such as prolonging winter storage, shortening summer storage, cooling of slurry in the barn.

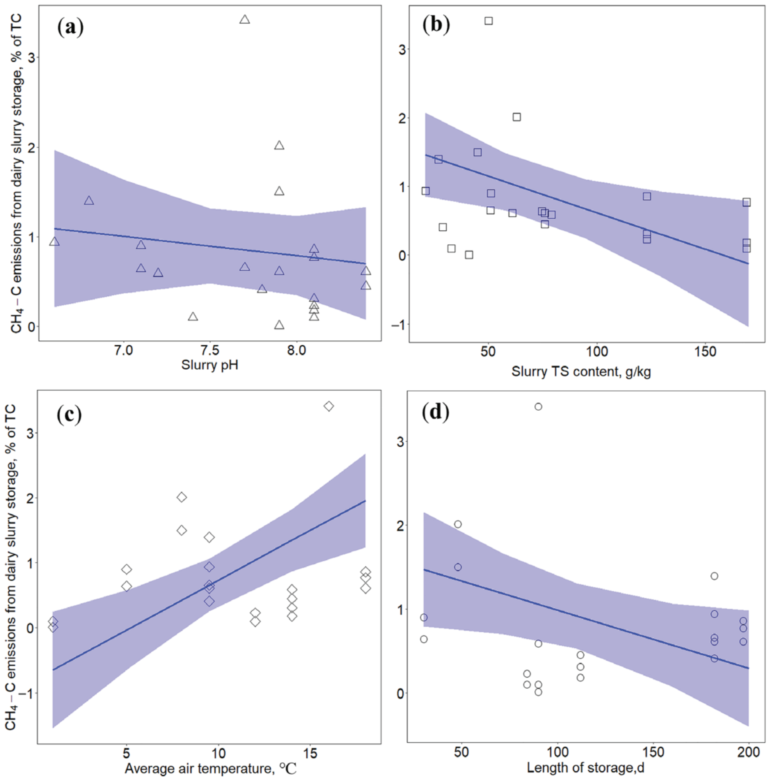

Figure 3.3. Relationships between CH_4–C emissions from slurry storage with (a) slurry pH (in triangle), (b) total solids (TS) content (in square), (c) air temperature (in diamond) and (d) length of storage (in circle). Blue lines indicate prediction lines based on results of linear mixed effects model: lmer (CH_4–C ~ pH + TS + T + LOS + (1|Reference)). Blue shadows represent model predicted results with 95% confidence interval.

3.4 Conclusions

Flow-based gas emission factors of slurry storage presented a large range and varied a lot among laboratory, pilot and farm scale studies. Slurry composition and storage conditions importantly define carbon and nitrogen transformations, and the resulting emissions of NH_3–N, N_2O–N and CH_4–C. Ammonia emissions were highly related to slurry pH. Lowering slurry pH significantly reduced NH_3–N losses. Ambient temperature also influenced NH_3–N losses. Storing slurry in warm seasons emitted more NH_3 than storing in cold seasons. No significant effect of TS contents on NH_3–N emissions was observed when slurry TS contents varied from 21−169 g/kg. Storing slurry more than 30 days did not significantly affect NH_3–N losses because NH_3–N emissions from slurry are most intensive at the initial stage of storage. The N_2O–N losses from slurry storage accounted for 0−0.39% of slurry TN and were not significantly affected by slurry pH, TS content, temperature, and length of storage. Methane emissions from slurry storage presented a significantly positive correlation with storage temperature, whereas the emissions decreased with the increasing of slurry TS content. Slurry pH ranging from 6.6–8.6 and length of storage had no significant influence on CH_4–C emissions. These information about relationships among gas emissions and slurry characteristics and storage conditions are useful for avoiding pollution swapping of mitigation strategies (e.g., acidification, additives, and separation etc.). Our results indicate that the inventory of gas emissions from slurry storage should consider the influence of temperature on emission estimates. More measurements on flow-based gas emission factors at farm scale are needed to better estimate carbon, nitrogen flows and cycles and nutrients use efficiency in dairy farming systems.

Supplementary materials

Supplementary materials to this article can be found online at https://www.mdpi.com/article/10 .3390/atmos12091156/s1.

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Chapter 4

Meta-analysis of greenhouse gas and ammonia emissions from dairy manure composting

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Abstract

In order to minimise nutrient losses, comprehensive overviews of the magnitude of gaseous emissions from manure composting processes and the factors that influence these losses are urgently needed. This study presents a meta-analysis of greenhouse gas (GHG) and ammonia (NH_3) emissions from four main dairy manure composting methods (static, turning, windrow and silo) based on 41 scientific articles (153 treatments). Gaseous emissions and secondary variables such as average composting temperature, initial moisture content, initial total carbon (TC) and initial total nitrogen (TN) content from each compost treatment were extracted and normalised to enable inter-study comparison. Six mitigation measures for composting were selected and mitigation efficiency (ME) of each measure on different gas emissions were calculated. Gaseous emissions from different composting methods showed large differences. Turning composting resulted in larger carbon and nitrogen losses compared to other composting methods. Although silo composting significantly promoted NH_3 emission, it reduced GHG losses by 82.84% compared with turning composting. Principal component analysis showed that the initial TC and TN content of the composted material were crucial in mediating gaseous emissions. Low TC and TN content can simultaneously reduce CH_4, CO_2 and N_2O emissions. Applying compost biofilters was the most effective way to reduce NH_3 emission with ME value of −97%. Adding sawdust or straw could reduce CH_4 and N_2O emissions by 66.3% and 44.0% respectively. Gaseous emissions from dairy manure composting varied a lot and were affected by physical characteristics of composted material and management practices of composting.

Keywords: nitrous oxide, methane, silo, turning, windrow, mitigation measure

4.1 Introduction

Manure composting technology has become the most popular form of manure management (Bernal et al., 2009; Onwosi et al., 2017), since it not only can reduce the volume of the accumulated faeces from intensive and specialised animal production but also produces a slow-release end-product, rich in humus, which could be used for crop fertilisation. However, manure composting also emits greenhouse gases such as N_2O, CO_2, CH_4 and NH_3 into the atmosphere, which could contribute to global warming, acidification of soil and formation of particulate matters in the air (NH_3). It has been reported that gaseous emissions from the manure composting process may account for 46% and 67% of the initial N and C content of the original manure, respectively (Shah et al., 2012). Most of the total N mass in the initial material can be lost by NH_3 emission throughout the composting process (Martins and Dewes, 1992; Sommer, 2001; Parkinson et al., 2004). Nitrous oxide emission accounted for about 0.1%~5% of total N losses (Sommer, 2001; Tamura and Osada, 2006; Maeda et al., 2013; Mulbry and Ahn, 2014), but it could cause more environmental concerns as N_2O has 265 times the global warming potential of CO_2 (IPCC, 2013). Carbon dioxide production was the principal pathway for C losses, and CH_4 emission may represent less than 10% (Hao et al., 2004; Mulbry and Ahn, 2014). These released gases are the major contributors to global warming (IPCC, 2013). Therefore, accurate estimation of gaseous emissions from dairy manure composting has great importance in mitigating nutrient losses and alleviating environmental pollution.

Currently, the calculation of GHG emissions from composting is based on default values recommended by the IPCC, with the values of 0.44−2.41 g CH_4/kg VS and 6−100 g N_2O−N/kg TN (IPCC, 2006). Nevertheless, these recommended values have been questioned because of large biases, with most of the GHG emission factors from literature of dairy manure composting (Sommer, 2001; Biala et al., 2016; Fillingham et al., 2017). For example, Sommer (2001) reported that 0.09 g CH_4−C/ kg DM and 2.00 g N_2O−N/kg TN were lost during the composting. Biala et al., (2016) believed gaseous emissions from composting were 0.004 g CH_4−C/kg DM and 0.01 g N_2O−N/kg TN, and Fillingham et al. (2017) reported gaseous emissions from composting as 0.85 g CH_4−C/kg DM and 10.40 g N_2O−N/kg TN. Even though much work in the literature has evaluated GHG and NH_3 emissions during composting processes on dairy farms, it is challenging to integrate and compare these results due to differences in composting methods (e.g., static, turning, windrow, silo) and functional units (e.g., emissions per kg fresh manure or per kg manure dry matter or other constituents). A systematic analysis can review and integrate quantitative results from publications, hence refining the emission factors with additional information about composting conditions. Previous systematic analyses of GHG and NH_3 emissions from dairy farms have mainly focused on anaerobic digestion processes (Ahn et al., 2011; Miranda et al., 2015; Miranda et al., 2016; Sajeev et al., 2018) and the whole manure management chain (Hou et al., 2015; Wang et al., 2018). A systematic analysis focused on gaseous emissions from dairy manure composting is still lacking.

This work addressed this by systematically reviewing and numerically combining studies of GHG and NH_3 emissions from dairy manure composting. We reviewed studies that analysed potential factors affecting gaseous emissions from the dairy manure composting process and the effects of mitigation measures during composting on CH_4, CO_2, N_2O, and NH_3 emissions. The specific objectives of this study were to: (1) compare differences in GHG and NH_3 emissions from different composting methods; (2) identify key environmental and chemical explanatory variables that might affect GHG emissions from dairy manure composting process; (3) analyse the impacts of mitigation measures on GHG and NH_3 emissions at the composting stage.

4.2 Materials and methods

4.2.1 Data sources and extraction

In order to include as many publications about dairy manure composting as possible, a systematic literature search and selection from bibliographic databases were performed. Keywords and logical connectors represented in Fig. 4.1 were used for searching scientific articles. The keywords aimed to identify papers focusing on the cattle husbandry sector, composting technologies and the emission indicators. Articles published before December 2018 were collected from Web of Science (WOS, http://apps.webofknowledge.com/) and the China Knowledge Resource Integrated Database (CNKI, http://www.cnki.net/). The language was set to English or Chinese. A total of 705 documents were obtained and imported to Endnote software.

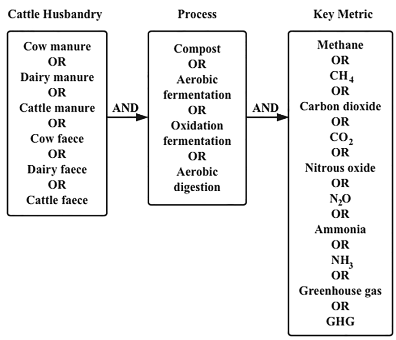

Figure 4.1. Keywords and logical connectors used in search for articles from the databases used in the literature search.

To ensure reliability of searched publications, a series of selection criteria was further set for a systematic selection of articles (Fig. 4.2). First the potential papers were selected based on the title and abstract. Then the full papers were inspected. Finally the papers included in this study were selected based on the following criteria: (1) studies focused on dairy manure composting; (2) studies included at least one of CH_4, CO_2, N_2O, and NH_3 emissions; (3) studies reported cumulative gas emission factors. With these criteria, 41 publications (27 from WOS and 14 from CNKI) with 153 compost treatments in total were selected. Basic information about publications such as authors, year of publication, composting methods, composting period, monitoring methods and reported gaseous emissions are presented in Table S1 for a complete overview. Four main categories of composting methods were identified: static (composting in piles with no turning for mixing); turning (composting in piles with regular turning for mixing); windrow (composting typically in a long narrow strip; the cross sections were trapezoidal or triangular with continuous mixing); silo (composted material within a closed container, continuous turning, computerised control of temperature and aeration). Data about mean GHG and NH_3 emissions from each compost treatment were extracted and data that were presented only in graphical form were digitised using WebPlotDigitizer (Burda et al., 2017). Compost physical indicators of initial total carbon (TC content), initial total nitrogen (TN content), initial moisture content and average temperature during composting were collected and recorded. Finally, a total of 153 treatments (69 for static composts, 27 for turning composts, 13 for windrow composts and 44 for silo composts) were used for further analysis.

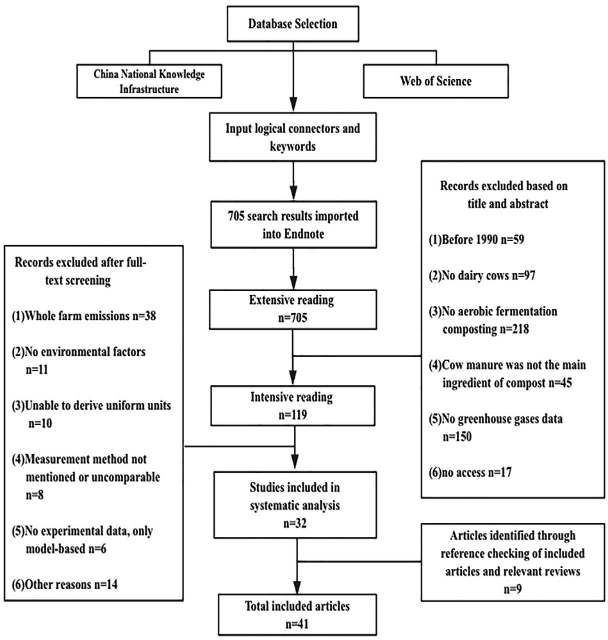

Figure 4.2. The procedure for selection of scientific publications.

4.2.2 Data analysis

4.2.2.1 Unit conversion

To perform statistical analysis, original reported data in various units for CH_4, CO_2, N_2O and NH_3 emissions from different composting treatments were converted to emission factors (EFs) such as g CH_4–C/kg DM, g CO_2–C/kg DM, g N_2O–N/kg TN and g NH_3–N/ kg TN based on a series of conversion equations (see Tables S2A, S2B, S2C and S2D in Supporting Data).

4.2.2.2 Normality test

A Shapiro-Wilk test was conducted to test the normality of data. The median of EF values was chosen to compare gaseous emissions from different composting methods since the data did not show a normal distribution. A nonparametric Wilcoxon Signed Rank test was performed to evaluate and compare significant differences of EFs among four composting methods. All statistical analyses were performed using R software (version: 3.5.1) (R core Team, 2017).

4.2.2.3 Calculation of the overall GHG emissions

To calculate the overall GHG emissions of each composting method, the various median EFs of gas emissions from different composting methods were first converted into kg/AU/year (1 AU refers to Animal Unit equaling 500 kg live weight) using the parameters presented in Supporting Data Table S2E. The CO_2 equivalents of CH_4 and N_2O emissions were obtained by using the Global Warming Potential of CH_4 and N_2O which are 28 and 265 respectively on the 100-year scale (IPCC, 2013). The CO_2 equivalents of GHG were calculated using the following formula:

$$E_{CO_2-eq} = 28E_{CH_4} + 265E_{N_2O}$$

Where E_{CO_2-eq} is the total CO_2 equivalent of GHG emission, kg CO_2-eq/AU/year; E_{CH_4} was the CH_4 cumulative emission of the composting, kg CH_4/AU/year; and E_{N_2O} was the N_2O cumulative emission, kg N_2O/AU/year.

4.2.2.4 Principal component analysis

In order to evaluate the effect of key environmental and chemical explanatory variables on gas emissions from dairy manure composting process, the datasets of static composting process were selected, which were not affected by human factors such as turning events.

Principal component analysis (PCA) was applied to reduce the dimensionality of the dataset and visualise the relationship between variables (Dray and Dufour, 2007; Lever et al., 2017). The ordinates were plotted using the 'ggbiplot' package (Vincent, 2011). The detailed information about loadings and correlation coefficients of PCA are presented in the Table S3 and S4. PCA for NH_3 emission and composition variables of interest was not performed because there were not enough treatments reporting NH_3 emission in the database.

4.2.2.5 Calculation of the mitigation efficiency of each measure

After systematically reviewing the 41 publications, six mitigation practices in the dairy manure composting process were identified, including "sawdust or straw additive", "microorganism additive", "phosphogypsum additive", "compressed and covered", "vermicomposting" and "compost biofilter". The efficiency of mitigation measures during composting were assessed by comparing the results of control and treatment emission factors using the following formula:

$$E_m = \left(\frac{EF_{trt}}{EF_{crt}} - 1 \right) \times 100\%$$

where E_m was the mitigation efficiency (%); EF_{trt} was the gas emission factor in the treatment group with mitigation measures and EF_{crt} was the gas emission factor in the control group without mitigation measures. Thus, a negative E_m value indicated a decrease in emissions due to the selected measure. Detailed information about the reported number of treatments, range, mean, SE (standard error), median and IQR (95% interquartile range) of mitigation efficiencies of each gas under different mitigation measures was presented in the Table S5, and the differences between the median E_m and 0 were also evaluated (P value).

4.3 Results and discussion

4.3.1 Gases emissions from different composting methods

Gaseous emissions from different composting methods are shown in Fig. 4.3. Basic statistics (including mean, median, range and 95% IQR) of emission factors of CH_4, CO_2, N_2O and NH_3 can be found in Table S6 in Supporting Data. A total of 65 treatments relating to CH_4 emission were reported in the database. Turning and windrow composting showed the highest median values of CH_4 EFs, followed by static composting, while silo composting had the lowest CH_4 emission (Fig. 4.3A). The high emissions for turning and windrow management were related to increased exposure to air, since frequent turning events disturb the surface crusting material, release the gas generated inside the compost and expose a new fraction of the substrate to oxygen (Amon et al., 2001). Silo composting with automated management (e.g. continuous turning, real-time monitoring and timely adjustment of equipment parameters of the composting process) ensured better conditions for composting compared to other systems, which were beneficial to the reduction of CH_4 emission (Fillingham et al., 2017; Guest et al., 2017).

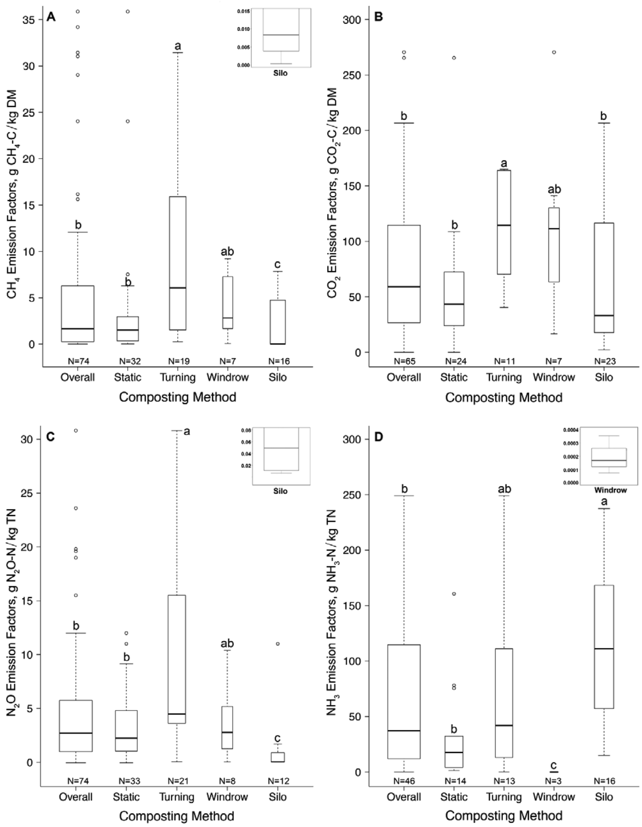

Figure 4.3. Gas emissions from different composting methods (static, turning, windrow and silo systems). Black solid lines in the boxplot represent the median quartiles, box boundaries indicate upper and lower quartiles. The whiskers indicate that values extend to 1.5 orders of the box length. o: outliers (>1.5 interquartile range). N = number of treatments. Different lower treatment letters within each boxplot indicate significant differences at P < 0.05.

The emissions of CO_2 from dairy manure composting were specified in 25 papers (65 treatments). The median CO_2 EF for turning was significantly higher than static and silo composting, while windrow composting had an intermediate emission rate (Fig. 4.3B). The increased CO_2 volatilisation in the storage methods with frequent turning could be mainly attributable to the greater pile porosity and the improved air permeability following turning events, thereby ensuring the oxygen supply to the microorganisms and promoting microbial breakdown of organic materials (Ahn et al., 2011; Arriaga et al., 2017).

The N_2O EFs of the composting process were extracted from 74 treatments. Results indicated that the median N_2O EF from turning composting was considerably higher than from other composting methods, although the difference from windrow composting was not significant (Fig. 4.3C). These larger losses might be attributable to turning operations which may promote N_2O losses through nitrification (aerobic) near the surface and denitrification (anaerobic) by mixing NO_3^-/NO_2^- accumulated on the surface into the inside of the pile (He et al., 2001; Hao, et al., 2001; El Kader et al., 2007; Ahn et al., 2011; Maeda et al., 2013; Chen et al., 2014). Other researchers have also thought the increase might be because of the role of enzymes. Denitrification enzymes are in an equilibrium state under anoxic conditions (Morley et al., 2008). When re-exposed to O_2 by turning, nitrous oxide reductases that can catalyse the transformation of N_2O to N_2 are clearly more severely inhibited by O_2 than the other reductases, resulting in a stronger N_2O emission (Nicholas et al., 2010; Burgin and Groffman, 2012; Saggar et al., 2013). The median of N_2O EFs from silo composting was the lowest among the composting methods. This might be due to the fact that denitrification (anaerobic process) was limited in the silo composter because the computer-controlled ventilation system could draw air into the composter, so the silo create better aerobic conditions for composting compared to other systems (Fillingham et al., 2017). Another reason was that silo composting emitted more NH_3 and therefore there might be less substrate for N_2O emission.

4

Results for the CO_2 equivalents of total GHG emissions from the four composting methods are presented in Table 4.1. Among these methods, turning composting resulted in the highest total GHG emissions. In contrast, silo composting exhibited the most favourable performance in terms of reducing total GHG emissions.

Table 4.1. Total GHG emissions from different composting methods (kg/AU/year).

Composting methods	Gases emissions		CO_2 equivalents of gases emissions		
	CH_4	N_2O	CH_4	N_2O	Total
Static	4.41	0.29	123.5	76.9	200.3
Turning	17.75	0.58	497.1	153.7	650.8
Windrow	8.23	0.36	230.6	95.4	326.0
Silo	0.03	0.006	0.8	1.6	2.4

In the database, a total of 20 papers (46 treatments) reported the NH_3 emission from dairy manure composting. The highest median value of NH_3 EFs was observed for silo composting with 111.07 g NH_3−N/kg TN. The high NH_3 emission may be attributed to high temperature and continuous turning. The average temperature in silo composting is about 65°C (Fillingham et al., 2017), which was beneficial to shift the equilibrium between ammonium (NH_4^+) and NH_3 towards gaseous NH_3. On the other hand, continuous turning in the silo composter would benefit the NH_3 release (Fillingham et al., 2017). As shown in Fig. 4.3D, windrow composts had the lowest NH_3 emission. The extreme low values may be related to the low temperature. The study of Arriaga et al. (2017) was conducted in northern Spain with average composting temperature 17.10 °C during 64-day composting period. Hao (2011) measured NH_3 emission from windrow composting in a cold region for 60 days, during which the composting temperature above 50 °C only lasted one day. The low composting temperature inhibits microbial ammonisation which could explain the lower NH_3 emissions (El Kader et al., 2007). In addition, the lowest NH_3 emission could also be attributed to large compost volume and wrapped wall around the pile in windrow composting, which is usually unfavourable for NH_3 volatilisation and can more easily form anaerobic environment (Beck-Friis et al., 2000).

4.3.2 Comparison with IPCC recommended emission factors

The Intergovernmental Panel on Climate Change (IPCC) recommended emission factors of CH_4 and N_2O from manure composting management with the values of 0.44−2.41 g CH_4/kg VS and 6−100 g N_2O−N/kg TN (IPCC, 2006). In order to compare with our data, the values of "0.44−2.41 g CH_4/kg VS" was converted to "0.36−2.01 g CH_4/kg DM" based on the VS/DM ratio of 5/6 in dairy manure (ASAE, 2005). The IPCC-recommended CH_4 and N_2O EFs had large biases with the median CH_4 and N_2O EFs obtained from the meta-analysis (Figs. 4.3A and 4.3C). These large differences were probably due to the facts that default value of IPCC were based on a limited number of studies in the literature and derived from various kinds of manure, while our study only focused on gaseous emissions from dairy manure composting (Chen et al., 2015; Chen et al., 2016; Wang et al., 2018). Therefore, accurate definition of IPCC default values from dairy manure composting would benefit a lot from distinguishing between manure types and composting methods. Our results from data synthesis can refine and supplement the EFs of IPCC-recommended values for different dairy manure composting methods.

4.3.3 Factors affecting gaseous emissions

A total of 10 studies (24 treatments) that simultaneously reported cumulative CH_4 emission, average compost temperature, initial moisture, initial TC and TN content were selected to identify the main factors influencing CH_4 emission. The first two principal components (Fig. 4.4A) jointly explained 85.7% of the variance in CH_4 emissions. The results showed that CH_4 emission was positively correlated to the moisture content of the compost. Within a certain range of high moisture content in the piles, the anaerobic zones are larger (Xie et al., 2003; Maeda et al., 2013) and more CH_4 is emitted. The TC content of initial material also exhibited a positive correlation with CH_4 emission mainly because of the direct participation of carbon source in CH_4 production (Hao, 2011). We found TN content promoted the CH_4 emission, which is consistent with the results of Pattey et al. (2005), perhaps because ammonium (NH_4^+) is a primary source of N for methanogenic bacteria (Knowles, 1993). The CH_4 yield was uncorrelated with temperature, which was in contrast to observations in other studies, which found that CH_4 yield was positively correlated to the average temperature of the composting (Sommer, 2001; Pattey et al., 2005; Yamulki, 2006; Qin et al., 2010; Sánchez-Monedero et al., 2010; Fan et al., 2017; Yang et al., 2017). The increased amount of CH_4 produced at higher temperatures was attributed to a faster rate of the methanogenesis process (Sommer, 2001; Pattey et al., 2005; Yamulki, 2006; Qin et al., 2010). Besides, the oxygen consumption rate of microorganisms is much higher than the oxygen supplement rate of the pile as the temperature rose, thus creating more anaerobic spaces in the compost (Sánchez-Monedero et al., 2010; Yang et al., 2017). However, in this study, the average temperature in 19 composting treatments of 24 treatments did not reach the optimum temperature for methanogens, about 35 °C (Wu et al., 2014), which could inhibit the activities of methanogens and lead to the effect of temperature on CH_4 emission being less obvious.

Only 9 studies (19 treatments) simultaneously reported CO_2 emission, average compost temperature, initial moisture, initial TC and TN content. The first two principal components of these variables are shown in Fig. 4.4B. Results showed TC and TN content were the main factors affecting CO_2 emission, because higher carbon and nitrogen content are more conducive to microorganism respiration. Microbial metabolism requires TC and TN content to provide the carbon and nitrogen source and the necessarily energy of survival (Gao, 2016). Qin et al. (2010) also indicated that changes in TC content would be linearly related to CO_2 emission. Temperature also exhibited a positive correlation with CO_2 yield. This could be due to the higher rate of decomposition of organic matter at higher temperature (Pattey et al., 2005; Arriaga et al., 2017). The PCA also showed CO_2 emission was negatively correlated with moisture content. The increase of moisture content promotes metabolism of microbial mass but could also expand the anaerobic area of the heap.

Principal component analysis of a total of 26 treatments (12 studies) relating to N_2O emission in the database are presented in Fig. 4.4C. The first principal component explained 45.3% of variance and the second component explained 30.2% of variance. The emission of N_2O

was positively related to the TN content of composting material, because nitrification and denitrification are promoted by high nitrogen content of the substrate (higher NH_4-N and NO_3-N content), resulting in an increase in N_2O emission (He et al., 2001; Huang et al., 2004). The N_2O production was also positively related to the TC content. Previous studies have shown that denitrifiers as heterotrophic microorganisms strongly rely on carbohydrates (and thus carbon) for energy (Burford and Bremner, 1975; Cabrera et al., 1994; El Kader et al., 2007). Similarly, temperature was positively related to N_2O emission, which was consistent with the results of El Kader et al. (2007) and Qin et al. (2010). This was because N_2O is produced by nitrification and denitrification whose rates could increase with increasing temperature (Granli, 1994). Our analysis showed that moisture content was not significantly correlated to N_2O emission. However, previous studies have shown that when the water content of the compost pile was in a certain range, the emission of N_2O would increase with water content. Since in a semi-humid environment aerobic and anaerobic regions would simultaneously exist, the nitrification and denitrification could be promoted concurrently and N_2O emission flux could become relatively high (Yan et al., 2013). However, the N_2O emission decreased when the material became very moist because of the inhibition of N_2O nitrification (Hwang and Hanaki, 2000). Research has demonstrated that the optimum conditions for minimising N_2O emissions during the composting were a moisture content of between 50% and 70% (Richard et al., 2002).

4.3.4 Effects of mitigation measures on gaseous emissions

Six mitigation practices in dairy manure composting process were identified and the mitigation efficiency (E_m) for different gaseous emissions was evaluated (Fig. 4.5). There were 6 papers (14 treatments) that studied the impact of sawdust or straw as compost additives on gaseous emissions from dairy manure composting. The results showed that the additives can be effective in both CH_4 and NH_3 mitigation, with ME values of −66.3% ($p = 0.002$) and −44.0% ($p<0.001$), but they may increase CO_2 emission. The changes of CH_4 and CO_2 may be because the presence of sawdust or straw would increase the porosity of the compost (Van Ginkel et al., 2002; Barrington et al., 2003; Hao et al., 2004), thereby increasing aerobic decomposition (Fig. 4.6: reaction ① could be improved and reactions ②, ③ could be inhibited). Besides, these additives can absorb ammonium (NH_4^+) in the faeces (and urine) after excretion, and enhance microbial assimilation of ammonium (NH_4^+) in the pile, thus decreasing NH_3 emission during composting (Fig. 4.6: reaction ⑤ could be improved and reaction ④ could be inhibited) (Chadwick, 2005; Steiner et al., 2010; Chowdhury et al., 2014a; 2014b).

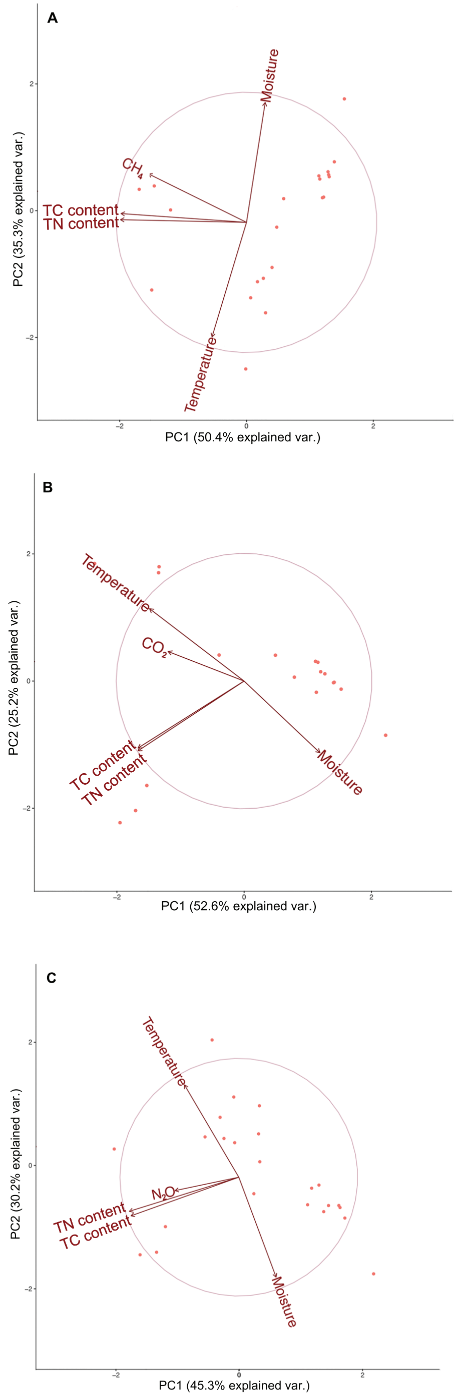

Figure 4.4. First and second principal components plots from the PCA for effects of composition and environmental factors on CH_4 (A), CO_2 (B) and N_2O (C) emission during composting. The circle indicates the 95% confidence interval.

Microorganism additives were mainly used for NH_3 mitigation. The microbial community CC-E (a complex bacterial community and *alcaligenes faecalis* is the main advantageous strain) and EM (Effective Microorganisms, a kind of commercial microbiological agent) are the main exogenous microbial additives in dairy composting currently (Shan and Shao, 2008; Chen et al., 2016). These additives delivered a mitigation potential for NH_3 emission during manure composting with ME of −9.15% ($p = 0.005$). Microorganisms could mineralise a large amount of organic nitrogen contained in the cow manure into ammonium nitrogen (NH_4^+-N) which could be transformed to nitrate by nitrification and eventually to N_2 by denitrification. Besides, the ammonium nitrogen (NH_4^+-N) can also be fixed as microbial protein under the action of fungi. By prompting nitrogen conversion to nitrate nitrogen and protein nitrogen, the ammonium nitrogen (NH_4^+-N) which was used in ammonisation could be reduced and eventually inhibit the NH_3 emissions (Fig. 4.6: reactions ⑤, ⑥ could be improved and reaction ④ could be inhibited) (Steiner et al., 2010; Chowdhury et al., 2014a; Chowdhury et al., 2014b; Chadwick, 2005).

Using phosphogypsum additives in composting process, the emission of NH_3 can be reduced by 55.7% ($p = 0.013$). The reduction of NH_3 volatilisation was because of the decrease of pH when phosphogypsum rate increases (Tubail et al., 2008). But the impact of phosphogypsum on GHG emissions from dairy manure composting was unclear because of limited studies. Previous studies found phosphogypsum could inhibit GHG emissions, especially CH_4 and N_2O emissions, during swine and cattle feedlot manure composting (Hao et al., 2005; Luo et al., 2013). The high sulphide concentrations and acidic conditions due to the use of phosphogypsum could inhibit methanogenesis, thus reduce CH_4 emission by 97.14% (Fig. 4.6: reaction ③ could be inhibited) (Hao et al., 2005). Liu et al. (2010) also indicated that phosphogypsum could result in lower pH and inhibit the action of N_2O reductase (Fig. 4.6: reaction ⑥ could be reduced). However the high hydrogen sulphide concentrations may also pose a health risk, which would need to be considered when applied this mitigation method.

Compressing and covering were also used to reduce CO_2 and NH_3 emissions during the composting process. The analysis showed that the mitigation efficiencies of this method could reach 10.1% for CO_2 emission and 24.3% for NH_3 emission. This was mainly because a poor supply of O_2 leads to lower microbial decomposition rates and ammonisation (Jungbluth et al., 2001; Chadwick, 2005). Moreover, covering can increase the resistance to gaseous diffusion into the air. If the cover was sawdust or straw, it could effectively absorb a part of CO_2 and NH_3 (Fig. 4.6: reactions ①, ④ could be reduced) (Zhu et al., 2015). If the cover was a plastic film, the water film under plastic film could function to block gas exchange, thereby slowing CO_2 and NH_3 dissipation (Zhu et al., 2017; Huang et al., 2018). However, compressing and covering can also strengthen the anaerobic conditions in the pile, thus leading to higher CH_4 emission (Fig. 4.6: reactions ②, ③ could be promoted) (median =29.08%, p = 0.013) (Chen et al., 2015; Chen et al., 2016; Wang et al., 2018). The measure had no significant impact on N_2O emission, although the mechanism is unclear and requires further investigation (Chen et al., 2015; Wang et al., 2018).

Vermicomposting was also applied to reduce NH_3 emission and a total of 6 treatments specified gas emissions during vermicomposting. Results showed vermicomposting can mitigate NH_3 emission with a ME median value of −33.5% (p = 0.002). This was attributed to the large specific surface area and loose texture in vermicomposting. These attributes create strong adsorption capacity and finally reduce production of NH_3 (Chen et al., 2015; Wang et al., 2018).

The impacts of biofilters on NH_3 emission from composting process were also investigated. Results showed NH_3 emission can be reduced significantly by 97% by using compost biofilters (p<0.001). Biofilters are thus highly effective in capturing NH_3 by adsorption and microbial degradation (Hong and Park, 2005). But this method has been applied less because the discharged air from composting is not being collected in most dairy farms and biofilters can only be used in a closed environment with collection equipment (Wang et al., 2018). No research was available for evaluating the effects of compost biofilters on CH_4, CO_2 and N_2O emissions.

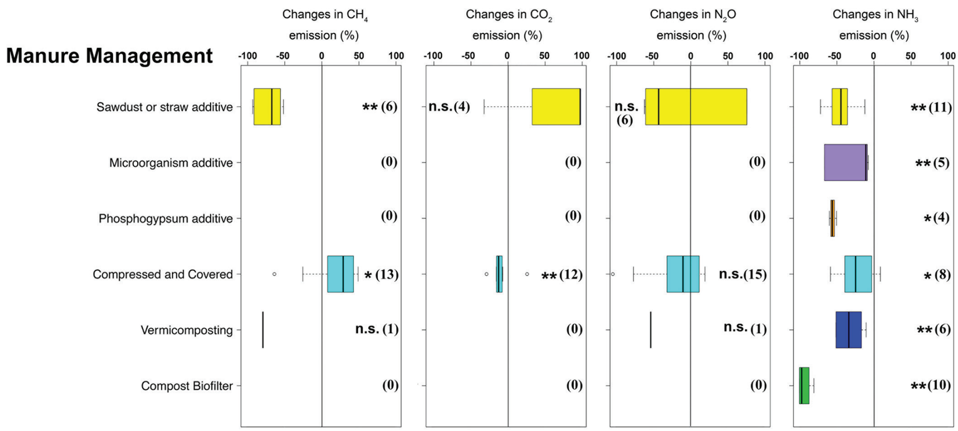

Figure 4.5. Box-plots showing the mitigation efficiencies of various measures for CH_4, CO_2, N_2O, and NH_3 emissions (see Tables S4 for numeric data). Black solid lines in the boxplot represent the median quartiles, box boundaries indicate upper and lower quartiles. The whiskers indicate that values extend to 1.5 orders of the box length. Values in the square brackets indicate outliers (>1.5 interquartile range). The number to the right of each box in parentheses represents the values of treatments. Wilcoxon Signed Rank test: *: P < 0.05; **: P < 0.01; n.s., not significantly different from zero. The lack of effects of various measures on CH_4, CO_2 and N_2O mean that no relevant measured values are obtained from current publications.

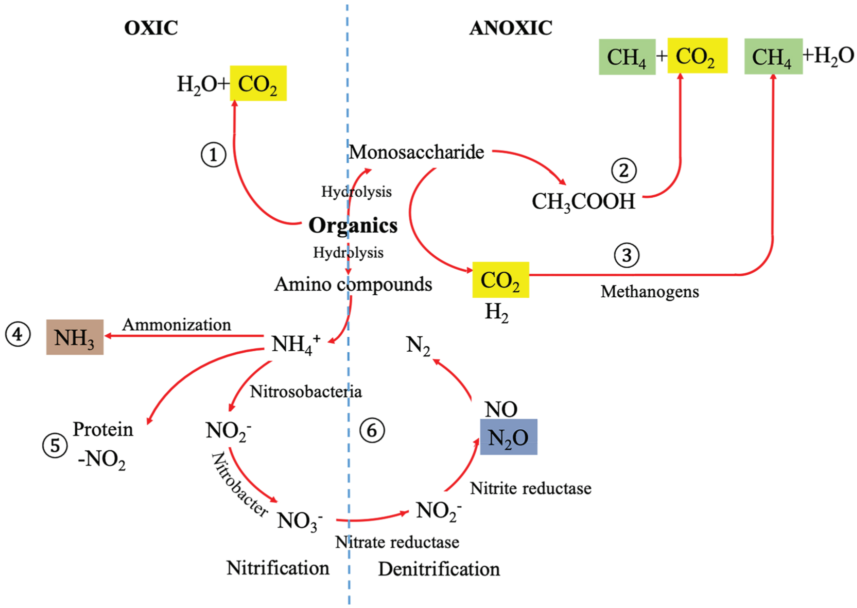

Figure 4.6. Schematic diagram of C and N cycles and reactions during one cycle of composting.

4.4 Conclusions

The emissions of GHG and NH_3 from four common composting methods (static, turning, windrow and silo composts) were compared. Turning composting would cause the highest gas emissions compared to other composting methods, whereas silo composts had the best performance in reducing GHG emissions though they significantly promoted N losses through NH_3 emission. The EFs from data synthesis in this study can refine and supplement the EFs of IPCC recommended values for dairy manure composting and improve estimation accuracy of gas emissions from composting. Reducing the initial TC and TN content of composting manure was a more effective way to minimise gaseous losses than adjusting the environmental conditions of composting. Of the mitigation measures for gas emissions, applying compost biofilters was the most effective way to reduce NH_3 emissions during composting. Adding sawdust or straw could significantly reduce CH_4 and N_2O emissions during composting.

Our results also showed that the number of studies quantifying NH_3 emission from dairy manure aerobic composting was limited. More attention should be paid to reducing NH_3 losses and improving nitrogen retention in composted products from dairy manure composting process in the future. Besides, limited data were available for principal component analysis due to the lack of reporting of compost physical and chemical characteristics in many publications. Therefore, we suggest that researchers should quantify physical and chemical properties of composted material along with the gas emissions from composting, to be able to meaningfully contribute to determining the main factors affecting gas emissions, and to optimise composting conditions. Overall, the results of this study have important implications for optimising the composting process, minimising nutrient loss and reducing environmental pollution.

Supplementary materials

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biosystemseng.2020.02.015.

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Chapter 5

A modular approach for quantification of nitrogen flows and losses along dairy manure management chains of different complexity

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Abstract

Nitrogen (N) loss from manure management chains (MMCs) of dairy farms are substantial and variable. The complexity of MMCs varies dependent on the method of collection and the number of subsequent storage and processing technologies used. We developed a modular approach that quantifies total ammoniacal N (TAN) and organic N flows along MMCs with different complexity. Emission factors of different N species for manure treatment facilities in MMCs were established based on published values. Simulated N losses from various MMCs from nine published case studies ranged from 20% to 50% of excreted N. Comparisons between simulated and reported N losses showed deviations ranging from 0.05% to 40% for the whole MMCs. Larger deviations were caused by uncertainty about emission factors for open lots, grazing lands and anaerobic lagoons. Using the modular approach, we could identify the elements that caused the deviations and could revise the reported emission factors, resulting in reduction of deviations in N losses to less than 10%. Simulated N losses were within the range of estimates obtained from the application the present Tier2/Tier 3 methodology, which further validates the reliability of our estimates. We conclude that the modular approach allows adequate estimations of N losses along MMCs and can help to identify the sources of differences or deviations of N loss estimations by different approaches. Further research should focus on N losses from open lots, anaerobic lagoons and N transformation during storage, which could further enhance the accuracy of estimations for N flows and losses in MMCs.

Keywords: dairy cows; TAN pool; organic N; gaseous emission; leaching and runoff; manure treatment facility

5.1 Introduction

Nitrogen (N) losses from manure management chains (MMCs) of dairy farms are substantial, varying from 20% to 70% of total N excreted (Mishima, 2002; Environmental Protection Agency (EPA) 2011; Bai et al., 2016). The pathways of N loss mainly include gaseous N emissions (e.g., NH_3, N_2O, NO and N_2), leaching and runoff, which not only lead to low N use efficiency but also pose a threat to the environment, such as greenhouse gases (GHG) and NH_3 emissions (Oenema and Tamminga, 2005; Steinfeld, 2006), and groundwater and surface water pollution (Oenema et al., 2007; Zhao et al., 2017).

Models to quantify N flows and losses associated with livestock manure management systems have been developed using either mechanistic process-oriented approaches, empirical equations, or emission factors, varying in the complexity and accuracy of application. Process-based mechanistic models, such as Manure-DNDC (Li et al., 2012) and Integrated Farm System Model (IFSM) (Rotz et al., 2012), consider biochemical and biophysical processes that govern the transport and transformation of nutrients in the manure life cycle. These mechanistic approaches offer robustness and flexibility of use in different manure management systems. However, simulations based on these models require large sets of data for parameterisation and are sometimes too complex for assessments at the farm scale. Whole farm models, such as DairyWise (Schils et al., 2007), $SIMS_{DAIRY}$ (Del Prado et al., 2011), MELODIE (Chardon et al., 2012), FarmDESIGN (Groot et al., 2012), NUTGRANJA 2.0 (Del Prado et al., 2014), FarmAC (Hutchings and Kristensen, 2015) have integrated emission factors or empirical equations (Table 5.1). These models are intended to simulate N flows and losses in different components of the farm and to identify the effects of changes in individual farm components on the whole-farm system. Moreover, some of farm models are intended primarily to estimate GHG emissions with comparatively crude assumptions about the losses of NH_3 and the flows of N from MMCs. Webb and Misselbrook (2004) and Dämmgen and Hutchings (2008) developed mass-flow methods to estimate gaseous N emissions along MMCs using emission factors. These approaches are much simpler than mechanistic models due to limited data requirements. They have been applied individually to estimate gaseous N emissions from parts of the MMCs, and were also integrated in whole farm models (e.g., SIMSDAIRY (Del Prado et al., 2011) and NUTGRANJA 2.0 (Del Prado et al., 2014)) or as a Tier 2/ Tier 3 methodologies in national emission inventory guidelines (IPCC, 2019; EEA, 2019) to simulate N flows and losses from MMCs.

However, this simplicity in representing emission processes using emission factors may lead to lower accuracy if the effects of diverse manure management facilities and technologies on emission of different N species are not considered (Petersen et al., 2013). Most existing models and approaches represent processes of N flows and losses during manure management facilities of excretion, storage, and application (Table 5.1). However, manure management facilities in intensive dairy production systems are becoming more complicated since a series of new manure management facilities and technologies (e.g., covering, anaerobic digestion, solid-liquid separation, and composting) have been developed to improve resource use efficiency and reducing the environmental burden (Chadwick et al., 2015; Holly et al., 2017; Rotz, 2017). To the best of our knowledge, there is no single model or approach that includes all these potential manure management facilities in dairy production systems.

Table 5.1. Overview of methods used by whole farm models to simulate N flows and losses from manure management systems.

Manure management facilities	Methods used by models					
	DairyWise (Schils et al., 2007)	$SIMS_{DAIRY}$ (Del Prado et al., 2011)	MELODIE (Chardon et al., 2012)	FarmDESIGN (Groot et al., 2012)	NUTGRANJA 2.0 (Del Prado et al., 2014)	FarmAC (Hutchings and Kristensen, 2015)
Manure excretion at housing facilities	NH_3, N_2O: empirical equations;	NH_3, N_2O, NO_x and N_2: emission factors;	NH_3, N_2O, N_2: existing empirical equations and emission factors;	NH_3, N_2O: emission factors;	NH_3, N_2O, NO_x, N_2: emission factors;	NH_3: emission factors;
Manure excretion on pasture	NH_3, N_2O: emission factors; NO_3^-: dynamic models;	NH_3: emission factors; N_2O, NO_x, N_2: dynamic models;	NH_3, N_2O, N_2 and NO_3^-: existing empirical equations and emission factors;	NH_3, N_2O: emission factors;	NH_3, N_2O, NO_x, NO_3^-: empirical models;	NH_3, N_2O, N_2: emission factors;
Manure storage facilities	NH_3: empirical equations; N_2O: emission factors;	NH_3, N_2O, NO_x and N_2: emission factors;	NH_3, N_2O, N_2: existing empirical equations and emission factors;	NH_3, N_2O: emission factors;	NH_3, N_2O, NO_x, N_2: emission factors;	NH_3, N_2O, N_2: emission factors;
Manure application	NH_3, N_2O: emission factors;	NH_3: emission factors; N_2O, NO_x, N_2: dynamic models;	NH_3, N_2O, N_2 and NO_3^-: existing empirical equations and emission factors;	NH_3, N_2O: emission factors;	NH_3, N_2O, NO_x, N_2, NO_3^-: empirical models;	NH_3, N_2O, N_2: emission factors;

Besides the various manure management facilities that can be incorporated into MMCs, accurate estimates of N flows and losses along MMCs must strictly follow mass balance principle, which means all relevant N flows and losses at manure management facilities must be considered. Most methods estimate manure gaseous N emissions, with few methods including potential N losses by runoff and leaching while the magnitude of these loss pathways might be significant for manure excreted on unpaved yards or lots and solid manure storage (IPCC, 2019). Hence, there is a need to develop a flexible and simple approach that considers complexities of manure management facilities at farm level and focuses on quantifying all possible N losses and flows along MMCs. This approach will contribute to improving the accuracy of modelling approaches based on mass flow analysis and emission factors.

Although emission factors of different N species from different manure management facilities (e.g., dairy barns, open lots, grazed lands, storage tanks, lagoons, solid storage, etc.) have been measured separately (Mosquera et al., 2006; McGinn et al., 2008; Borhan et al., 2011; Ngwabie et al., 2014; Wang et al., 2016; Aguirre-Villegas and Larson, 2017; Baldé et al., 2018), it is difficult to integrate these results to evaluate N flows and losses in whole MMCs, because these studies usually reported emission factors with different units (e.g., g/cow/d, g/m, g/kg N etc.). Therefore, standard emission factors measured by percentage of total ammoniacal N (TAN) or total nitrogen (TN) for all potential N species are needed to establish to facilitate the estimation of N flows and losses along the whole MMCs.

The objectives of this study were to develop a modular approach for estimating N flows and losses from MMCs of different complexity based on mass flow analysis; to collect and summarize emission factors of different N species in percentage of TAN or percentage of TN for different manure management facilities; to simulate and compare the N flows and losses from different MMCs based on the developed modular approach and examples from literatures; and to compare N losses from MMCs estimated by different approaches.

5.2 Materials and methods

5.2.1 Model Development

The stages of MMCs in dairy farms include manure excretion, manure handling and collection, manure storage, manure treatment and manure field spreading or manure output (e.g., export or recycling). The main manure handling practices and technologies applied at different stages in manure management of dairy farms are summarized in Fig. 5.1. Using a modular concept, the N flows and losses at each manure management facility were quantified by calculating the N input from previous stages, added N, conversions of organic N into inorganic N (mainly in TAN) and vice versa, and losses and outputs of N (Fig. 5.2). The main loss pathways of N include gaseous emissions, runoff and leaching into soils from the solid storage of manure at outdoor areas, in open lots and in pastures. Gaseous N emissions (e.g., NH_3, N_2O, NO and N_2) occur directly or indirectly from the TAN pool in manure (Dämmgen and Hutchings 2008). Leaching and runoff of N could also originate from the organic N pool in manure. Consequently, in this approach, we distinguish the N flows and losses between TAN and organic N (Fig. 5.2). By combining different manure management facilities, this modular approach enables to simulate TAN and organic N flows and losses under different complexity of MMCs.

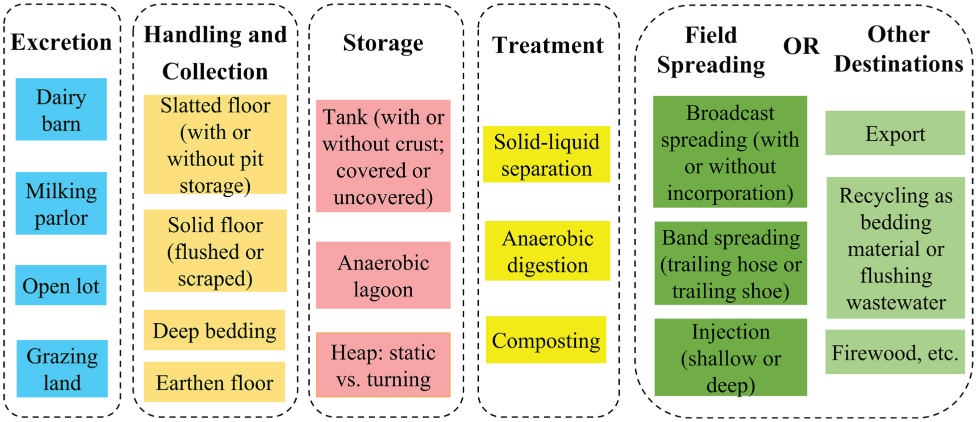

Figure 5.1. An overview of potential manure handling practices and technologies applied at different stages in manure management of dairy farms.

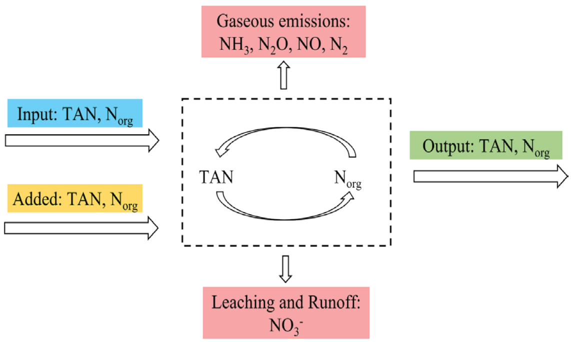

Figure 5.2. Flows of total ammonium nitrogen (TAN) and organic nitrogen (Norg) in each manure treatment facility. The blue box represents N input from previous facilities. The orange box indicates N added by bedding straw, flushing (waste)water or co-digestion material. Red boxes mean N losses by gaseous N emissions, leaching and runoff. The green box indicates the N output from this treatment facility in products.

5.2.2 Calculation procedure

The flows and losses of TAN and organic N along MMCs of dairy farms were quantified in twelve consecutive steps. A detailed calculation procedure is given in Supplementary Material A. The calculation procedure begins with the estimate of annual N excretion for each type of cows and the distribution of TAN and organic N excretion at different facilities (e.g., dairy barn, milking parlour, open lots and grazing lands). Additional TAN and organic N from bedding material and flushing (waste)water are estimated and added to initial excretion pools of TAN and organic N. Losses of TAN and organic N from each facility are estimated using emission factors of different N species and subtracted from the TAN and organic N pools which then flows to the next manure management facilities (e.g., aerobic/anaerobic storage, separator, biogas digester and compost). For each manure management facility, the transformations and losses of N are estimated. Finally, outputs of TAN and organic N to different destinations (e.g., field application, export, recycling as bedding material or flushing wastewater, etc.) are quantified by the proportions of manure allocated to each destination. For manure spreading to fields, the NH_3-N loss during spreading are estimated and subtracted from the TAN pool to field application.

5.2.3 Emission factors

Emission factors of different N species depend on manure characteristics and conditions in manure management facilities in dairy farms. We summarized reported emission factors in TAN or TN from published literature. As presented in Table S2 – S8 in Supplementary Material A, gaseous N emission factors were mainly expressed as a percentage of TAN, while N losses by leaching and runoff were estimated as a percentage of TN. Specially, a clear difference on emission factors for grazing lands was observed between urine and dung. Thus, emission factors were given separately based on TAN content of urine and organic N of faeces (Table S3). For manure spreading, the losses of NH_3-N highly depend on manure characteristics and application methods. We categorized manure into liquid manure, slurry and solid manure based on total solids content (Aguirre-Villegas and Larson, 2017; Rotz, 2017). Emission factors of NH_3-N were given for each manure type with different application methods (Table S6 – S8). Emission factors expressed in g per day per animal were converted to the percentage of TAN or TN lost based on estimated N excreted and the percentage of TAN in TN. For each emission factor, if possible, we calculated the mean, median and minimum and maximum values.

In addition, the transformations of TAN and organic N by mineralization or immobilization during storage were summarized and listed in Table S9 in Supplementary Material A. The mineralization rate of organic N during slurry storage ranged from less than 10% to more than 50%, with higher mineralization rate (averaged about 30% of organic N) under anaerobic storage conditions. We estimated that about 10% and 5% of organic N could be mineralized to TAN for solid manure storage without adding straw and for solid manure stored on open lot for a long period, respectively, as more aerobic conditions during solid manure storage might enhance the decomposition of organic matter. The immobilization rate of TAN in solid manure added with straw varied from 25% to 40% depending on the C/N ratios of solid manure (Kirchman and Witter, 1989; Menzi et al., 2003; Webb and Misselbrook, 2004; Dämmgen and Hutchings, 2008; Velthof et al., 2012).

5.2.4 Model verification and validation

In order to verify whether the modular approach can accurately estimate N flows and losses from MMCs, we found nine examples from publications (Table 5.2). These examples covered the most frequently used manure management facilities, such as dairy barn, open lot, milking parlour, grazing land, slurry aerobic/anaerobic storage, solid manure storage, separator, biogas digester. For each example, we simulated N flows and losses through the MMC using the developed approach and reported averaged emission factors. The simulated results were firstly inspected by the N balance, which was calculated by the input of N including the total amount of N excreted by different cow types, the N added from bedding, from flushing (waste)water and from co-digestion material, the N lost by gaseous N emissions, leaching and runoff and the N leaving the MMC. In anaerobic lagoons, a part of N might be retained in sludge after a long period of storage.

Comparisons with literature reported results were further conducted to validate the reliability of calculation and to identify the deviations between simulated and reported results. We also compared the calculated results of developed methods with estimates by approaches recommended by the Intergovernmental Panel on Climate Change (IPCC, 2019) and European Environment Agency (EEA, 2019). Both IPCC (2019) and EEA (2019) developed a Tier 3 or Tier 2 approach based on mass flow analysis to estimate emissions from manure management facilities. The default values of emission factors based on total N content entering manure management facilities (IPCC, 2019) or default emission factors based on the flow of TAN in the Tier 2 approach of EEA (2019) were given. These methods facilitate comparing the calculation procedure and results with the developed approach.

Table 5.2. Manure management practices in selected nine examples from publications.

Example	Manure management practices	Reference
1	Dairy barn -- flushing system -- separation -- holding pond -- application	Van Horn et al., 2003
2	Dairy barn -- flushing system -- anaerobic lagoon -- application	
3	Open lot -- scrape -- application	
	Open lot -- runoff -- holding pond -- application	
	Milking parlor -- flushing system -- anaerobic lagoon -- application	
4	Grazed land--application	
	Milking parlor -- flushing system -- anaerobic lagoon -- application	
5	Dairy barn – flushing system -- anaerobic lagoon -- application	Li et al., 2012
6	Milking parlor -- flushing system -- lagoon -- application	Leytem et al., 2018
	Open lot -- scrape -- solid manure storage -- application	
7	Dairy barn -- collection -- anaerobic digester -- application	Hoang et al., 2019
8	Dairy barn -- scrape -- open-air storage -- application	Fang et al., 2020
	Dairy barn -- urine flow -- lagoon -- application	
	Open lot -- scrape -- open-air storage -- application	
9	Dairy barn -- slatted floor -- deep pits -- application	
	Open lot -- scrape -- open-air storage -- application	

5.2.5 Sensitivity analysis

The sensitivity of the simulated results to variation in emission factors of different N species at different manure treatment facilities was investigated by changing the value of the single emission factor and holding all others at baseline values. A hypothetical dairy farm holding with 100 dairy cows was established. The annual N excretion was about 14019 kg, of which 50% as TAN. Several manure management scenarios for this dairy farm were assumed:

- Scenario 1: Dairy cows were held in a free-stall barn with concrete floor. All of the manure was flushed into a large anaerobic lagoon and the effluent from the lagoon was used for irrigation of cultivated crops.
- Scenario 2: Dairy cows were held in a deep bedding barn. Manure and bedding straw (1898 kg N/year) were mixed and collected twice per year. Produced farmyard manure was stored in a heap and applied to fields later.

- Scenario 3: Dairy cows were held on open lots. Manure excreted on earthen open lot was cleaned twice a week and was placed into open-air storage. Solid manure was spread onto the surface of surrounding farmland.
- Scenario 4: Dairy cows were maintained on pastures. All of the manure was dropped on grazed lands.

For each scenario, the baseline of N losses from MMCs was estimated using emission factors established in this study. The effect of changes in emission factors was assessed by assuming that emission factors were 20% and 50% smaller and larger, relative to the baseline values.

5.3 Results

5.3.1 Simulated N flows and losses from MMCs

Based on manure management practices in selected publications (Table 5.2), we simulated the flows and losses of TAN and organic N in each manure management facility along MMCs using the developed modular approach. As illustrated in Fig. 5.3, taking Example 2 as an example, one hundred dairy cows were held in a dairy barn equipped with solid floor and flushing system. Annual N excretion was about 14019 kg, of which 50% was TAN. About 14% of TAN was lost by NH_3-N volatilization from the barn and the remaining TAN and organic N were flushed into an anaerobic lagoon. We estimated that 30% of organic N was mineralized to TAN during lagoon storage, resulting in 8131 kg TAN and 4907 kg organic N. In the TAN pool, a large amount of TAN was lost by NH_3-N emission with a small percentage lost through N_2O, NO and N_2 emissions. In addition, due to the long retention time of slurry in the lagoon, about 10% of N was retained in sludge in which 10% was in TAN and 90% in organic N. After the lagoon storage, about 6278 kg N per year could be applied to fields. Simulated N flows and losses for other examples are listed in Supplementary Material B. The nitrogen balances of simulated results for selected examples are presented in Table S10 in Supplementary Material A.

5.3.2 Comparing simulated N loss with reported results

Comparisons between simulated and reported N losses from MMCs is shown in Fig. 5.4. There were large deviations in Examples 2, 3, 4 and 7 ranging from 27% to 50%. These differences were caused by discrepancies in emission factors for anaerobic lagoons, open lots, grazing lands and digested slurry storage.

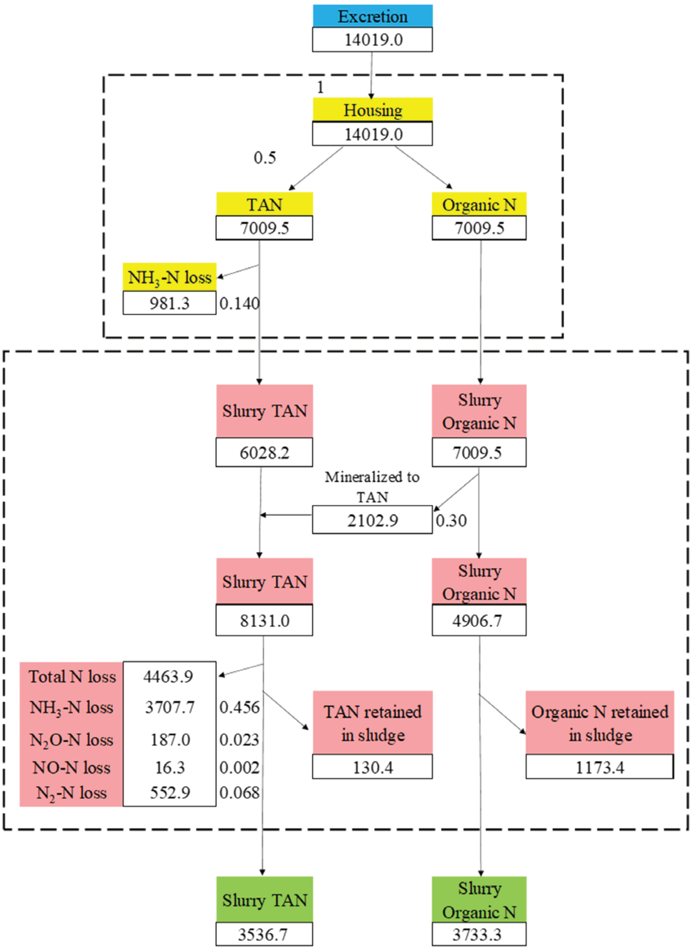

Figure 5.3. Nitrogen flows and losses along the manure management chain of Example 2 (Van Horn et al. 2003). Cells with different colours represent N flows and losses at different stages of manure management chain, i.e., blue for N excretion, yellow for dairy barn, red for lagoon, green for application. Numbers in cells indicate the annual amount of N flows and losses (kg/year) at each treatment facility, and numbers outside cells represent fraction of N distribution and transformation, and emission factors (kg/kg TAN) of different N species (TAN: total ammoniacal nitrogen).

As shown in Fig. 5.5a, the large difference of simulated and reported N losses in Example 2 was caused by the difference in emission factors for anaerobic lagoon. The reported N loss from anaerobic lagoon was 7991 kg N per year accounting for 60% of N entering the lagoon, which is 80% larger than the calculated results based on an emission factor 55% of TAN in lagoon. Revising the reported emission factor from 60% to 35% of TN (IPCC, for lagoon storage would result in a reduction of the deviation of N loss between simulated and reported to less than 1% (Fig. 5.4).

In Example 3, the reported N loss from open lot was twice the calculated amount, which led to the large difference in available applied N (Fig. 5.5b). When the reported N loss rate from open lot was adjusted from 65% to 30% of TN (IPCC, 2019), the deviation between simulated and reported N loss could be reduced to 5.5% (Fig. 5.4). The difference in emission factors for grazing lands resulted in a considerable difference between calculated and reported results of Example 4 (Fig. 5.5c). The reported N loss from grazing land was 6308 kg N per year, two times higher than the calculated N loss. If we reduced the N loss rate from 60% to 46.6% of TN for grazing lands (IPCC, 2019), the deviation could be reduced from 39% to 29% (Fig. 5.4).

In Example 7, there was a large difference in N loss from digested slurry storage (Fig. 5.5d). About 8909 kg N per year was estimated to emit from digested slurry storage, which was three times the reported N loss. Digested slurry usually has high pH and TAN content and might emit more NH_3. When the reported emission factor for digested slurry storage was revised from 12% to 42% of TAN, the deviation could be reduced to less than 10% (Fig. 5.4).

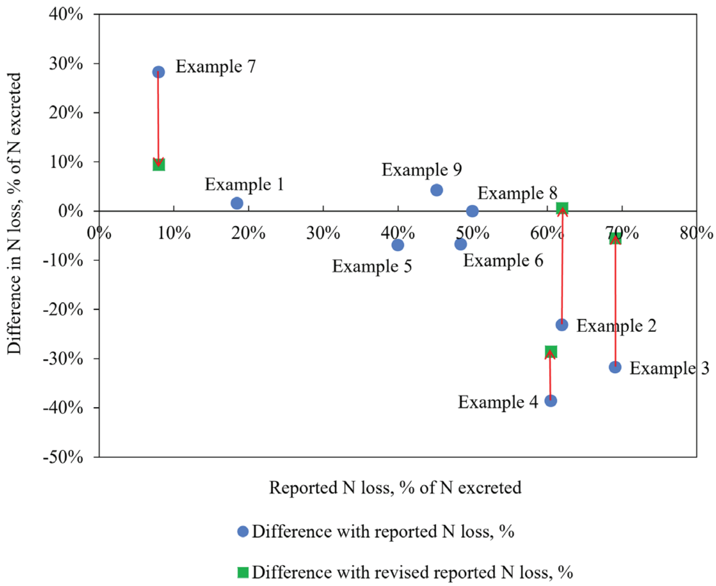

Figure 5.4. Deviations between simulated and reported N losses from manure management chains. Red arrows indicate the reduced difference in N losses after revising emission factors.

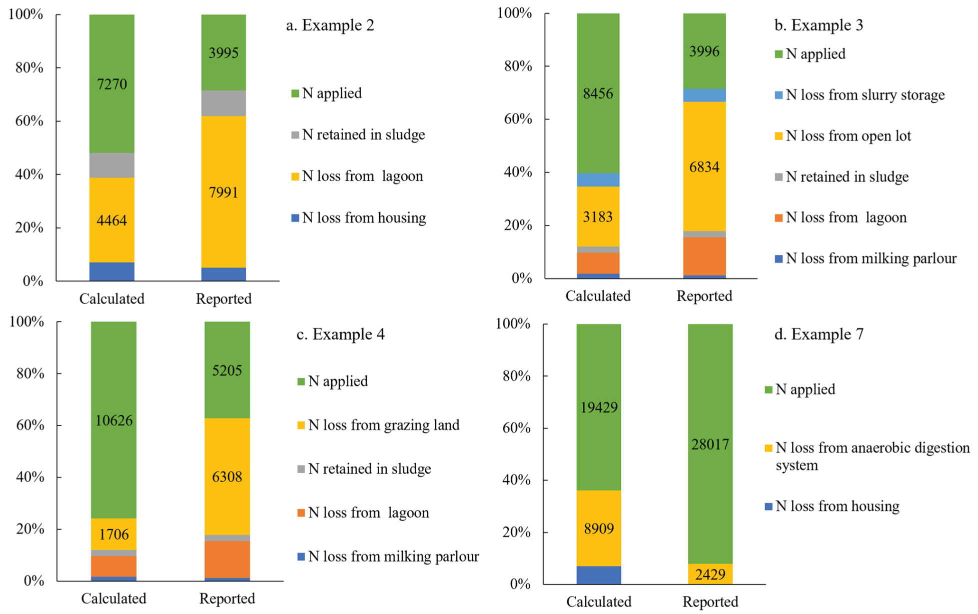

Figure 5.5. Comparison between calculated and reported N losses from manure management chains in Examples 2, 3, 4 and 7. All outflows of N in the manure management chain are presented, the sum of which equals the N inputs. Numbers in columns represent the amount of N lost from treatment facilities and the amount of N available for application.

5.3.3 Comparison with IPCC and EEA estimates

For each of the nine examples, we compared N losses estimated by the modular approach with calculations using the IPCC (2019) and EEA (2019) procedures. As shown in Fig. 5.6, the N losses from MMCs calculated by the modular approach were within the range of N losses estimated by IPCC (2019) and EEA (2019) approaches, except Examples 1, 8 and 9. For Example 1, there was a small difference in the N loss from the manure management chain among different approaches, ranging from 18% to 29% of N excreted and with higher loss estimated by EEA (2019) approach and emission factors. The higher estimate from EEA (2019) in Example 1 was due to the fact that the slurry was stored for a short time and the default emission factors in EEA (2019) might overestimate N loss. The higher N losses from Examples 8 and 9 estimated by the developed approach were due to the higher N losses on earthen open lots. We assumed that almost all urine N (about 60% of TN) excreted on earthen yards would be lost by NH_3 volatilization, leaching and runoff, which were 18% and 42% higher respectively than estimated N losses using the IPCC (2019) and EEA (2019) approaches.

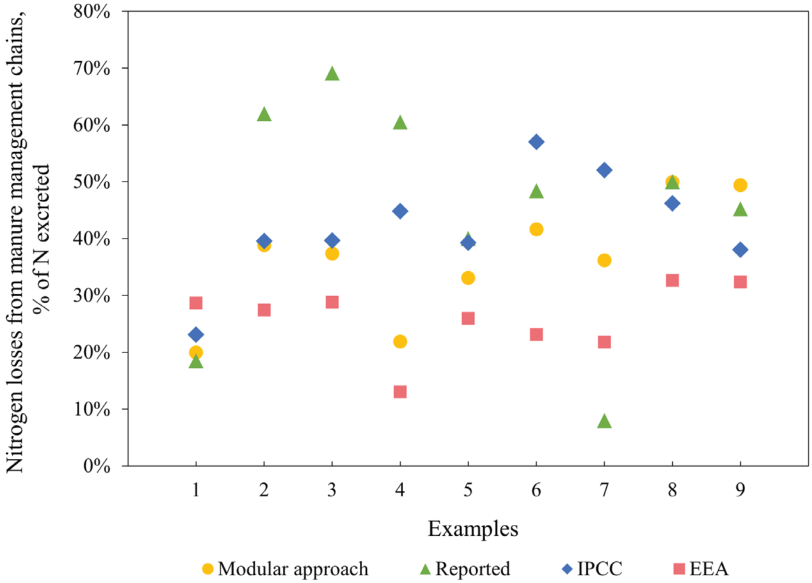

Figure 5.6. Estimates of N losses from manure management chains as reported in publications, calculated by the modular approach and by IPCC and EEA procedures (IPCC, 2019; EEA, 2019).

5.3.4 Sensitivity analysis

As illustrated in Fig. 5.7, for Scenario 1, the estimated N loss from the MMC was highly sensitive to NH_3-N emission factor for anaerobic lagoon. Changing emission factors of NH_3-N with 20% and 50% led to about 5% to 14% changes in the total N losses, respectively. The estimates of mineralization rate of organic N in anaerobic lagoon had little effects on N loss from the MMC, resulting in less than 5% change in total N loss (Fig. 5.7). For the deep litter bedding system in Scenario 2, only when the immobilization rate in deep litter and the emission factor of NH_3-N for farmyard manure storage were changed by 50%, changes of estimates of N loss from the MMC were larger than 5% (Fig. 5.7). The emission factor of N_2-N during solid manure storage had little effect on the total N loss from the MMC (Fig. 5.7). In Scenario 3, changing the NH_3-N emission factors for open lot with 20% and 50% resulted in 5% to 14% changes in N losses, while the total N loss from the MMC was less sensitive to the mineralization rate of organic N during solid manure storage. For the grazing system in Scenario 4, the effect of emission factor of N leaching of urine on the total N loss could be significant (more than 5%) when the variation of factors was larger than 50% (Fig. 5.7). The N loss by NH_3-N volatilization from grazing land was small and had little effect on the total N loss from the MMC.

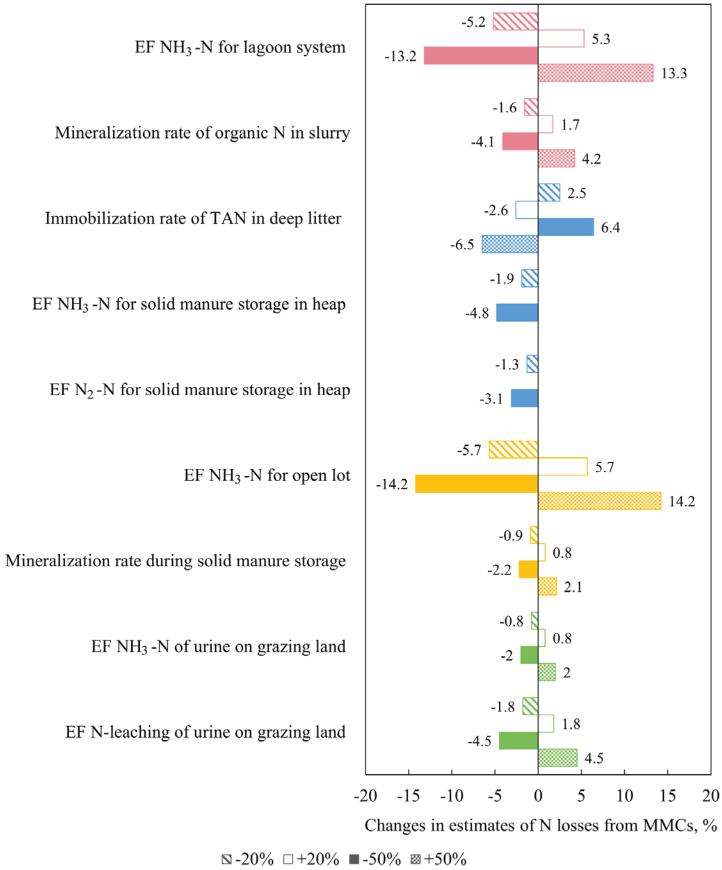

Figure 5.7. Effects of changing emission factors on N losses from manure management chains (MMCs) based on manure management scenarios 1 (red), 2 (blue), 3 (orange) and 4 (green) (EF: emission factor, TAN: total ammoniacal nitrogen).

5.4 Discussion

5.4.1 Main findings

The proposed modular modelling approach was able to capture N flows and losses along MMCs with different complexity in dairy farms. It allowed to identify the most sensitive stages in the MMCs and the emission factors that were most uncertain. Our results revealed that large deviations of estimates were mainly caused by the uncertainty about emission factors for open lots, grazing lands and anaerobic lagoons. Sensitivity analysis verified that changing emission factors of NH_3-N for open lots and anaerobic lagoons could result in considerable changes in the estimates of total N losses from the MMCs.

This modular approach distinguished TAN and organic N pools based on mass flow analysis and allowed to quantify all possible N loss pathways in different manure management facilities (e.g., slurry storage, lagoon, solid-liquid separator, biogas digester, solid manure storage etc.). In addition to the objectives of the study, this approach was able to identify shortcomings in the quantification of emission factors in the selected validation studies, which could help correct over- or under-estimations from modelling approaches.

5.4.2 Modelling approaches

Modelling approaches based on emission factors need fewer parameters and are much simpler in principle for estimating N losses from MMCs when compared with mechanistic and empirical models and tools (Petersen et al., 2013). However, the trade-off might exist between the complexity and accuracy of modelling approaches. The simplicity of calculations with emission factors comes with a trade-off of reduced accuracy and flexibility for application in contrasting manure management systems if the effects of various manure management practices on emission factors of N losses are not considered.

Existing modelling approaches presented emission factors of different N species for various manure management facilities, with differences in the detail of the effects of manure management practices on emission factors. Webb and Misselbrook (2004) distinguished NH_3 emission factors for slurry and farmyard manure during housing buildings, slurry storage facilities, farmyard manure storage and manure spreading. Dämmgen and Hutchings (2008) and EEA (2019) extended emission factors of N species by including N_2O, NO and N_2 emissions for slurry and litter-based manure produced at barns, yards, grazing lands, storage facilities and field application. The national emission inventory guideline developed by IPCC (2019) considered the effects of manure management facilities on emission factors and recommended default values of N_2O emission, NH_3 and NO_X volatilization, and leaching N for various manure management facilities with or without mitigation measures.

However, few of these approaches included and estimated N flows and losses from separation and anaerobic digestion systems. Pardo et al. (2017) developed a new modelling approach ($SIMS_{WASTE-AD}$) to calculate NH_3 and N_2O emissions from anaerobic digestion processes, including pre-anaerobic digestion, anaerobic digestion, post-anaerobic digestion and solid-liquid separation of digested slurry. While it is primarily designed to be applied within the $SIMS_{DAIRY}$ modelling framework, this new approach is also expected to interact with other models in integrated approaches. With more available information about emission factors of N losses from various manure management practices, an improvement on the accuracy of quantifying N losses from MMCs using modelling approaches based on emission factors could be achieved by integrating as much as potential manure management facilities to mass flow analysis of MMCs and by compiling experimental emission factors of N losses from different manure management facilities.

5.4.3 Strengths of this modular approach

In this modular approach, we firstly extend flexibility and application of this modelling approach by taking each potential manure management facilities as additional, candidate modules and by integrating contrasting manure management facilities to cover various MMCs in on-farm settings. This modular concept makes this approach highly adaptable and possible for farmers to build simulations tailored to their manure management systems. This modular approach is also more adequate than previously proposed methodologies as all potential N loss pathways are accounted for. The parsimonious simulation method avoids too much and detailed parameters estimation and has enough accuracy, "getting the right answers for the right reasons" (Keating, 2020). Simulated results of nine examples from publications validated the feasibility of our approach to evaluate N flows and losses along different MMCs.

5.4.4 Emission factors

The uncertainty of emission factors for open lots, anaerobic lagoons and grazing lands could lead to large deviations in estimates of N losses from the MMCs using different approaches. The magnitude of N losses from these facilities had a large range and was highly influenced by manure management practices (e.g., exposed manure surface area, manure mixing, manure characteristics etc.) and climate variables (e.g., temperature, wind speed, rainfall etc.) (Leytem et al., 2011; Leytem et al., 2018). These processes are difficult to parameterize, especially in on-farm settings. Future emission factor estimates need to capture the temporal and spatial variabilities in emissions and to include the key influencing variables driving the losses of different N species. Moreover, measurement data must be collected over time periods long enough to capture these variations in emissions for accurate estimates of annual emission factors.

Effects of mitigation strategies on emission factors of different N species in manure management facilities should also be considered. We established emission factors for slurry storage with covers, for which the NH_3-N loss reduced about 90% compared with the loss from slurry storage without natural crust. For solid manure storage, Pardo et al. (2015) demonstrated that covering and compaction of solid manure composting substantially decreased NH_3 emissions by 61% and 54%, respectively. Additives could also reduce NH_3 emissions and mitigation efficiencies varied with additives type, with high mitigation efficiencies by sawdust or straw (44%), and phosphogypsum additives (56%), low reduction efficiencies by microorganism additives (9%) (Ba et al., 2020). However, it should be noted that a certain degree of pollution swapping and compensatory loss of mitigation options have been observed by previous studies (Reidy et al., 2008; 2009). Sajeev et al. (2018) presented that although the use of covers (e.g., straw, granules) for slurry storage could significantly reduce NH_3 emissions, they might lead to large increases in N_2O emissions due to the development of both aerobic and anaerobic zones in the surface layer. Shah et al., (2013) demonstrated in a modelling study that the reduction of N losses of mitigation practices (e.g., additives and cover) during solid manure storage were compensated by extra emissions after application. These results further highlight that an accurate estimate should be based on a detailed knowledge of partial emission factors and conversion processes in different facilities as affected by mitigation strategies.

5.4.5 Implications

Our study will contribute to understanding the effects of contrasting management-related technical measures on N flows and losses in whole MMCs. Further, it will allow to evaluate N use efficiency of different manure management systems and optimize manure management practices at farm level. Besides applied individually to estimate N flows and losses along MMCs with different complexity, it can also be integrated with or linked to whole-farm models to assess the role of MMCs in farm nutrient management and to identify points for improvement in farm configuration and management.

5.5 Conclusions

We proposed a modular approach to estimate TAN and organic N flows, and to quantify different N species loss (e.g., NH_3, N_2O, NO and N_2 emissions, N leaching and runoff) from MMCs with different complexity in dairy farms. Estimated N losses from various MMCs were within the range of estimates of IPCC (2019) and EEA (2019), which validates the reliability of our estimates. Emission factors for open lots and anaerobic lagoons had a large uncertainty and could have significant effects on N losses from MMCs. This developed modular approach could be used to identify the sources of differences or deviations of N losses from MMCs estimated by different approaches and can help correct over- or under-estimations. This approach improves the flexibility of whole-farm modelling approaches in estimating N flows and losses along contrasting manure management practices and will contribute to understanding of the effects of various management-related technical measures on manure N flows and losses.

Future studies should focus on more accurately quantifying N emissions of N_2O, NO and N_2, and losses by leaching and runoff N from open lots, anaerobic lagoons and solid manure storage. Developing a modelling approach for estimating emission factors of N losses as functions of management practices and environmental conditions will allow for better modelling of temporal and regional differences in N losses from MMCs. In addition, more research on integrated measurement of N flows along various MMCs in practical dairy farms is needed for comparing the difference of N losses between measured and simulated results and for further validation of modelling estimates.

Supplementary materials

Supplementary data to this article can be found online at https://doi.org/10.1007/s10705-021-10183-0.

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Chapter 6

A model to identify entry points to curb emissions from complex manure management chains

Under revision at *Journal of Cleaner Production* as: Qu, Q., Zhang, K., Groot, J.C.J. A model to identify entry points to curb emissions from complex manure management chains.

Abstract

Livestock operations deploy increasingly complex facilities and technologies in manure management to reduce negative environmental impacts and to improve the agronomic value of manures. To capture and quantify processes of degradation, conversion and emission of manure constituents in these complex systems, this study presented a newly developed modular manure management (FarmM3) model. Using this model, we simulated flows and losses of manure organic matter (OM), carbon (C), nitrogen (N), phosphorus (P) and potassium (K) from manure management chains (MMCs) with deep litter, anaerobic lagoon, solid-liquid separation (SLS), anaerobic digestion (AD), and combinations of SLS and AD. The sensitivity of degradation and losses of manure constituents to changes in the configuration and parameters of MMCs was assessed. Results showed the MMCs with deep litter and AD led to higher OM degradation, C losses and greenhouse gas (GHG) emissions due to the substantial amounts of straw added to bedding and the digester. A trade-off between GHG and ammonia emissions was identified in the MMCs with deep litter. Application of SLS could reduce GHG emissions by 40% to 60% due to reduced methane and nitrous oxide emissions from separated liquid fraction storage. A stronger reduction of ammonia emission was observed when applying SLS to digested slurry than to raw slurry. Sensitivity analysis showed that the N loss was most sensitive to N transformation in the MMC with deep litter, and was most affected by the loss coefficients of ammonia during liquid manure storage and application in MMCs with SLS and AD. Losses of P and K from MMCs with SLS were influenced by separation efficiencies from SLS and loss coefficients from solid fraction storage. The impact of model input parameters on GHG emissions highly depended on the selected manure management facilities. This study shows that manure management facilities have a strong influence on the fate of manure constituents. The FarmM3 model can be used to quantify the degradation and losses of different manure constituents in complex MMCs and the effects of manure treatment facilities, and to identify the most important parameters determining these losses.

Keywords: modelling, manure constituent, greenhouse gas emission, solid-liquid separation, anaerobic digestion, winding stairs sensitivity analysis

6.1 Introduction

The intensification and specialization of dairy production resulted in the decoupling of crop and dairy farming. With a substantial amount of produced manure but few available lands, these intensive dairy farming systems posed detrimental impacts on the environment, such as gaseous emissions, groundwater and surface waters pollution, and excessive use of feed additives (e.g., heavy metals, antibiotics, micronutrients) (Oenema et al., 2007; Kuppusamy et al., 2018). To reduce the environmental risk of gaseous emissions and other nutrient losses and to increase operational flexibility in manure management, various emerging manure management facilities are available and prioritized in dairy farms with high animal density (Hou et al., 2018; Tan et al., 2021; Niles et al., 2022). For example, solid-liquid separation (SLS) can be used to separate slurry into a diluted liquid fraction and a nutrient-rich solid fraction using different types of mechanical separators, which not only could increase manure fertilizer value but also facilitate exporting solid fractions to avoid nutrient surpluses within farms (Hjorth et al., 2010; Sommer et al., 2013). Anaerobic digestion (AD) has been used to produce biogas (a mixture of methane and carbon dioxide) as a source of alternative energy by breaking down manure OM in the absence of oxygen (Foged et al., 2011), and has been proven to reduce GHG emissions (Aguirre-Villegas and Larson, 2017; Holly et al., 2017).

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Application of these management facilities may induce changes in physical, chemical and/or biological properties of manure and hence influence the decomposition of OM and carbon (C), and the fate of nutrients within manure management chains (MMCs) (Hou et al., 2014; Khalil et al., 2016). Aguirre-Villegas et al. (2019) found that applying SLS could retain more total solids and volatile solids but much less total ammoniacal nitrogen and total potassium in separated solid fractions, which resulted in both greenhouse gas (GHG) and ammonia (NH_3) reductions from storage and land application compared to a scenario without SLS. AD alone and combined AD and SLS could reduce GHG emissions due to reduction of the quantity of volatile solids in liquid manure storage but could also lead to increased NH_3 emissions due to the increased total ammoniacal nitrogen from mineralization of organic nitrogen during digestion.

Given the possible interactive effects of manure management facilities on emissions, the importance of integrated modelling approaches in estimating gaseous emissions and nutrient flows from a whole chain perspective has been pointed out (Hou et al.,2014; Sajeev et al., 2017; Wei et al., 2021). Table A6.1 provides a list of existing integrated modelling approaches. However, most of these approaches mainly focus on traditional manure management facilities, i.e., a linear process of manure excretion, manure storage and application. Few of them allow to integrate the emerging on-farm manure management facilities (e.g., SLS, AD, composting, etc.) and enable to evaluate the impacts of these new manure management facilities on nutrient losses along the whole MMC. Pardo et al. (2017) designed a module ($SIMS_{WASTE-AD}$) to calculate gases emissions from AD processes, but the new module was aimed to be applied within the $SIMS_{DAIRY}$ modelling framework (Del Prado et al., 2011) to account for potential effects of AD on nutrient flows. Dairy-CropSyst developed by Khalil et al. (2019) allowed to evaluate the effects of diverse manure management facilities (AD, separation and nutrient recovery) on nutrient fate through MMCs with liquid manure handling systems with lagoons, while not addressing solid manure handling systems in MMCs. Sefeedpari et al. (2019) introduced a process-based analysis model that can be used to calculate degradation and losses of manure constituents through MMCs with different manure management facilities. However, Sefeedpari et al. (2019) only quantified the quality of final products to applied fields without estimating the losses of manure constituents from manure application, which might not be able to fully capture the interactive effects of manure management facilities on nutrient losses since the reduced losses before application might lead to increased losses after application (Shah et al., 2013). The approaches listed in Table A6.1 focus on only one or a few manure constituents or gaseous emissions (NH_3, N_2O, CH_4, GHG, etc.), and conversions and losses of manure OM and C, phosphorus (P) and potassium (K) from MMCs are sparsely considered. It was reported that over 50% of the excreted manure P and K could be lost from MMCs (Bai et al., 2016). Knowledge of degradation of OM and losses of P and K along various MMCs could contribute to a more comprehensive assessment of the performance of MMCs.

As discussed above, there is lack of a model that has more flexibility of integrating various emerging manure management facilities and could comprehensively evaluate flows and losses of different manure constituents (OM, C, N, P and K) of diverse MMCs. We address this issue by introducing a newly developed modular manure management (FarmM3) model. It has the advantage of being able to integrate more alternative manure management facilities from excretion to application to cover complex MMCs in a modular way. It can assess the effects of emerging manure treatment technologies on different manure constituents (OM, C, N, P and K).

In this paper, we firstly describe this newly developed, flexible and extendable FarmM3 model for quantifying conversions and losses of OM, C, N, P and K along dairy MMCs with different complexity. Then with the FarmM3 model, we assess different MMCs with diverse manure management facilities and evaluate environmental impacts by comparing nutrient losses, NH_3 and GHG emissions. Finally, a global sensitivity analysis is performed to investigate how the system level losses of OM, C, N, P and K from MMCs are influenced by variations in configurations of manure management facilities and loss coefficients.

6.2 Materials and methods

6.2.1 Model description

A modular approach developed by Qu et al. (2022) allows to estimate N flows and losses along diverse MMCs in a flexible way. In this study, we further extended this approach to quantify degradation of OM and C, losses of P and K by integrating loss coefficients from the literature and developed a modular manure management (FarmM3) model. Fig. A6.1 shows the main data window of FarmM3 model. This model includes four types of components: Inputs, Pools, Separators and Applications. The Inputs components specify the quantities of materials added such as the excreted manure, amendments of bedding materials or crop residues for co-digestion with a given composition (dry matter (DM) content, ash content, and C, N, P, K contents). Conversion and loss coefficients of OM, C, N, P and K are established from experiments or literature reviews for each manure management Pool in which manure is deposited or maintained. Separators split a Pool in two new Pools and for each nutrient the fraction allocated to the new Pools can be specified. Applications should be the endpoints of the MMC, in which the loss coefficients of manure nutrients during application are specified. Table A6.2 lists the input parameters of different components in the FarmM3 model.

Any number of these types of components can be combined into MMCs that start with Inputs and finish with Applications of manure fractions in fields or barns. Within each component, flows and losses of OM, C, organic N, inorganic N, P and K are quantified using a mass balance approach by calculating input from the previous component, conversion and loss within the component and output to the next component. The accumulated degradation and losses of manure constituents from the whole MMC are derived by summing up the losses in different types of components. Also, losses of different N species (i.e., NH_3-N, N_2O-N, NO-N, N_2-N, leaching and runoff N) and different C species (CO_2-C and CH_4-C) through the MMC are presented. These losses are also expressed per cow and per unit of area by dividing by the number of cows and the total surface area of the farm. The detailed calculation procedures are presented in the supplementary material.

6.2.2 Model visualization

The FarmM3 modelling tool was developed in MS Visual Studio using the C# programming language. The flows of manure constituents along the whole chain of MMC were visualized using DOT language in Graphviz software that allows to create diagrams with code and have them automatically drawn (Ellson et al., 2004). As shown in Fig. 6.1 with a flow diagram of inorganic N through an example MMC as generated by the FarmM3 model. The nodes were labelled with different types of inputs, manure management facilities, or loss pathways of manure constituents along the MMC. The amounts of flows and losses of manure constituents were added to vertices between nodes.

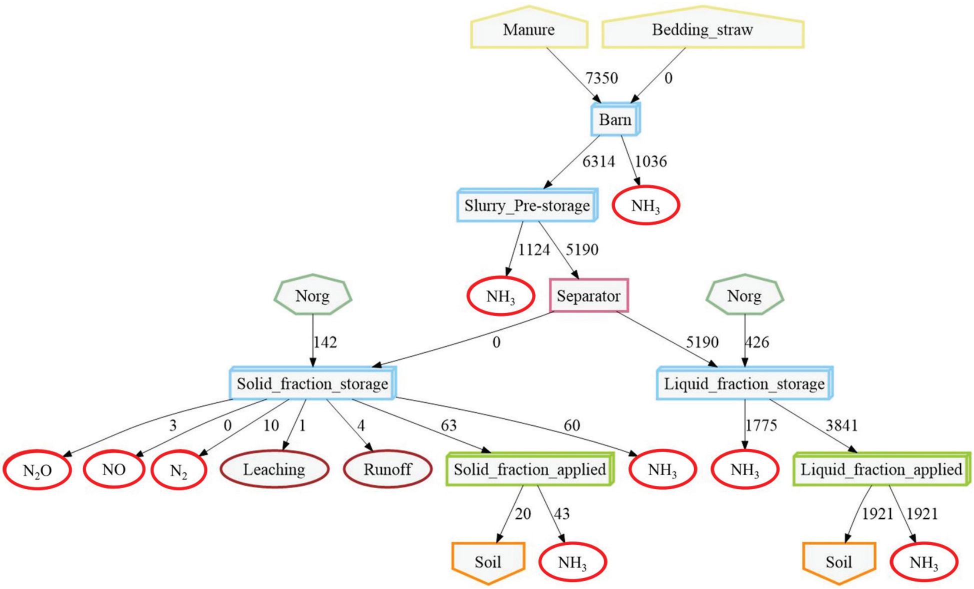

Figure 6.1. The flow of inorganic nitrogen (Nmin) through the manure management chain (MMC) with a mechanical solid-liquid separation (SLS). The golden house shapes represent Inputs, the blue boxes denote Pools, the pink rectangle indicate Separators and the green boxes represent Applications.

6.2.3 Analysis of manure management scenarios

A hypothetical dairy farm with 100 cows was developed with several contrasting manure management scenarios as shown in Table 6.1. Manure management facilities included deep litter with farmyard manure (FYM) storage, anaerobic lagoon, exercise yard, SLS, AD and combinations of SLS and AD. The annual amount of manure excreted is 1,201,018 kg/year, with different quantities of added straw for bedding or co-digestion, as shown in Table A6.3.

Table 6.1. Description of manure management scenarios. The arrows indicated the flows of manure constituents among management facilities.

Scenarios	Scenario description	Manure management chains
S1	Deep litter and FYM storage	Dairy barn → Deep litter → Farmyard manure storage → Broadcast spreading
		Milking parlor → Flushing system → Slurry storage tank → Broadcast spreading
		Grazing pasture
S2	Anaerobic lagoon storage	Dairy barn → Combined scraping and flushing system → Anaerobic lagoons → Broadcast spreading
S3	Anaerobic lagoon and yard manure storage	Dairy barn → Combined scraping and flushing system → Anaerobic lagoons → Broadcast spreading
		Exercise yards → Scraping system → Solid manure in heap → Broadcast spreading
S4	SLS	Dairy barn → Flushing system → Slurry pre-storage → Separator → Separated liquid fraction storage tank → Broadcast spreading
		Dairy barn → Flushing system → Slurry pre-storage → Separator → Separated solid fraction composting → Broadcast spreading
S5	AD	Dairy barn → Scraping system → Slurry pre-storage → Anaerobic digester → Digested slurry storage → Broadcast spreading
S6	AD and SLS	Dairy barn → Scraping system → Slurry pre-storage → Anaerobic digester → Digested slurry storage → Separator → Separated liquid fraction storage tank → Broadcast spreading
		Dairy barn → Scraping system → Slurry pre-storage → Anaerobic digester → Digested slurry storage → Separator → Separated solid fraction composting → Broadcast spreading
S7	SLS and AD	Dairy barn → Flushing system → Slurry pre-storage → Separator → Separated liquid fraction storage tank → Anaerobic digester → Digested slurry storage tank → Broadcast spreading
		Dairy barn → Flushing system → Slurry pre-storage → Separator → Separated solid fraction composting → Broadcast spreading

Note: FYM represents farmyard manure; SLS represents solid-liquid separation; AD represents anaerobic digestion.

6.2.4 Winding Stairs sensitivity analysis

Variations in manure management facilities, and the inherent uncertainties associated with emission factors, can have substantial implications for estimated results. A variance-based Winding Stairs sensitivity analysis was performed to assess the effects of variations in loss coefficients on the expected degradation of OM, nutrient flows and emissions from MMCs. Given the complexity of MMCs and potential interactions between loss coefficients of manure management facilities, in this study, we selected manure management scenarios

1, 4 and 6 in Table 6.1 as examples to analyse the effects of variations in loss coefficients on output variables. Results of the sensitivity analysis of scenarios 2, 3, 5 and 7 are presented in Tables A6.4 to A6.7.

The Winding Stairs algorithm incorporated in the FarmM3 modelling tool is based on Monte-Carlo sensitivity analysis but performs a systematic sampling of random parameter values within user-defined ranges. Output variables Y_i are decided by input parameters X_1, X_2, …, X_k based on a developed deterministic function f ($Y_i = f(X_1, X_2, …, X_k)$), here represented by the model calculations as described in the supplementary material. Sampling of parameter values occurs in a cyclical order. In the first step of cycle 1, X_{11} is randomly adjusted, in the second X_{21}, etc. thereby producing new values $\{X_{11}, X_{21}, …, X_{k1}\}$. Thus, each cycle contains K steps that constitute one Winding Stairs sample or 'winding' (Jansen et al., 1994). The number of random Winding Stairs samples generated (R) can be set as a parameter of the algorithm. The total number of observations generated is N = K × (R+1), where 1 represents the original parameter set that is used at the start of the first cycle. For each sample of parameter values the model output variables are calculated using the function f. This results in a matrix with K columns and R+1 rows, see Fig. A6.2 for an example from Chan et al. (2000) with K=3 and R=4.

In our case, the system level losses of OM, inorganic N, P, K and GHG emissions from MMCs were selected as output variables. We excluded NH_3 emissions and total C losses as output variables due to strong correlations between NH_3 emissions and inorganic N losses (Pearson correlation coefficients = 0.858 to 0.999), and between total C losses and OM degradation (Pearson correlation coefficients = 0.954 to 1.000). The Pearson correlation coefficients among output variables in different MMCs are listed in Table A6.8. The original parameter values, minimum and maximum values of selected input parameters were specified based on empirical values from publications. These input factors were sampled randomly within the set ranges through 5000 windings (R=5000).

The variance of model output variables was decomposed into the first-order sensitivity index (FSI) and total sensitivity index (TSI). The FSI, also called top marginal variance, is defined as the variance reduction due to fixing factor X_k while varying the other factors. Conversely, the TSI, also denoted as bottom marginal variance, is the variance caused when only X_k is uncertain (Jansen, 1999; Chan et al., 2000). These indices can be used to evaluate the main effects (FSI values) and the total effects (TSI values), including main and interactive effects, of these parameters on the system level losses of OM, inorganic N, P, K and GHG emissions. Small differences between FSI and TSI values indicate that there is no interaction between parameters.

6.3 Results

6.3.1 Degradation and losses of manure constituents from MMCs

As shown in Table 6.2, the amounts of OM degradation, C and nutrient losses, NH_3 and GHG emissions from seven manure management scenarios were compared. For the hypothetical dairy farm with 100 cows, the MMC with deep litter and FYM storage (Scenario 1) had the lowest NH_3 emissions and total N loss. In contrast, the amounts of OM degraded and C lost in Scenario 1 were five to seven times higher than losses in Scenarios 2, 3 and 4. Additionally, the GHG emissions from Scenario 1 were much higher than emissions from Scenarios 2, 3 and 4. The results indicated the pollution swapping of NH_3 emissions and OM degradation, C loss and GHG emissions from the MMC with deep litter. The substantial amount of added straw to deep litter provided more substrate for degradation of OM but could absorb urine N quickly and promoted the immobilization of inorganic N to organic N, thereby reducing NH_3 emissions.

We compared the degradation and losses of manure constituents in Scenarios 2, 3 and 4 because they all have the same amounts of manure and bedding straw input (Table A6.3). In comparison with Scenarios 2 and 3, the larger amounts of degraded OM degradation and C loss in Scenario 4 were caused by the higher degradation rate of OM under aerobic conditions during separated solid manure storage. The SLS in Scenario 4 helped reduce GHG emissions by more than 50%, due to lower total CH_4 and N_2O emissions compared to Scenarios 2 and 3. Small differences in N losses and NH3 emissions in Scenarios 2 and 4 were observed. Scenario 3 presented higher environmental risk of losing nutrients (N, P and K) by leaching and runoff, compared to Scenarios 2 and 4 (the MMCs without exercise yards). In Scenarios 5, 6 and 7, the OM degradation, C losses, and GHG emissions were relatively higher compared to other scenarios, due to the substantial amounts of straw added to AD. Applying SLS after AD (Scenario 6) reduced GHG emissions by 44% compared to applying AD only (Scenario 5). This was due to the lower CH_4 emissions from separated liquid fraction storage in Scenario 6 (Fig. A6.3). Additionally, Scenario 6 resulted in slightly lower N loss and NH_3 emissions. Changing the sequence of manure management facilities might influence flows and losses of manure constituents. Applying SLS before AD (Scenario 7) resulted in lower OM degradation, C loss, but higher GHG emissions than applying SLS after AD (Scenario 6). The main differences between the GHG emissions in Scenarios 6 and 7 were due to the higher CH_4 emissions from the digested slurry storage, compared to the separated liquid fraction from digestate in Scenario 6 (Fig. A6.3). This can be further explained by the larger quantity of volatile solids in the digested slurry storage in Scenario 7 due to a large amount of straw added to the digester with the separated liquid fraction. Scenario 6, on the other hand, only had a small percentage of volatile solids from digestate which can be retained in the liquid fraction after separation. Although the total losses of P and K from both Scenarios 6 and 7 were small, about 60% lower P loss was observed from Scenario 7. This was because the higher separation efficiency of P in digested slurry (Scenario 6) than in raw slurry (Scenario 7) led to more P retained in the solid fraction, thereby increasing leaching and runoff losses of P. Different from P, the higher separation efficiency of K in raw slurry than in digested slurry led to 40% more K loss in Scenario 7, compared to Scenario 6. These results show that there are contrasting effects of separation efficiency of SLS on total P and K losses.

Table 6.2. Amounts of OM degradation, nutrient losses and GHG emissions from different manure management scenarios. The added straw for animal bedding and anaerobic digester were also included in material flows.

Scenarios	Description	OM degraded	Total C loss	GHG emissions	Total N loss	NH_3 emissions	Total P loss	Total K loss
		kg/cow		kg CO_2-eq /cow	kg/cow	kg/cow	g/cow	g/cow
1	Deep litter and FYM storage	2297.1	1148.5	5765.5	44.6	30.5	24.9	3273.7
2	Anaerobic lagoon storage	291.2	145.6	2561.7	63.7	57.3	0.0	0.0
3	Anaerobic lagoon storage and yard manure storage	370.4	185.2	2507.0	72.4	60.0	3088.9	62956.3
4	SLS	422.3	211.1	1060.0	60.2	59.6	7.5	721.7
5	AD	5760.1	2880.0	1974.9	120.3	120.3	0.0	0.0
6	AD and SLS	5724.8	2862.4	1101.4	110.1	109.7	19.8	514.1
7	SLS and AD	5380.0	2690.0	1768.3	117.9	117.4	7.5	721.7

Note: FYM represents farmyard manure; SLS represents solid-liquid separation; AD represents anaerobic digestion.

6.3.2 Sensitivity of system level losses from MMCs to variations of loss parameters

6.3.2.1 Degradation of organic matter

The most important variables influencing degradation of OM differed among scenarios and depended on applied manure management facilities in MMCs. The Winding Stairs sensitivity analysis showed that degradation rates of OM under aerobic conditions in deep litter and in FYM storage contributed more than 80% of variance to total OM degradation from Scenario 1, whereas the contribution of degradation rates of OM in slurry storage was almost negligible (Table 6.3). In Scenario 4, both the degradation rate of OM in separated solid manure storage and in separated liquid fraction storage were influential, with two times higher FSI value for degradation rate of OM in separated solid manure storage than in separated liquid manure storage (Table 6.3). In Scenario 6 with a digester and a separator, the degradation of OM was the most sensitive to the degradation rate of OM in the digester (Table 6.3). The effects of these input parameters on degradation of OM from MMCs in Scenarios 1, 4 and 6 are presented in Fig. 6.2, with higher parameter values leading to more degradation of OM from MMCs. It should be noted that the contribution of input parameters to total variance of OM degradation from MMCs depends on the range of input parameters and the degree of influence of these parameters represented by regression coefficients (Fig. 6.2). A small range but large value of regression coefficient might lead to a small fraction of explained variance, and consequently a small value of coefficient of determination (R^2) and FSI and TSI values (Figs. 6.2d and 6.2i).

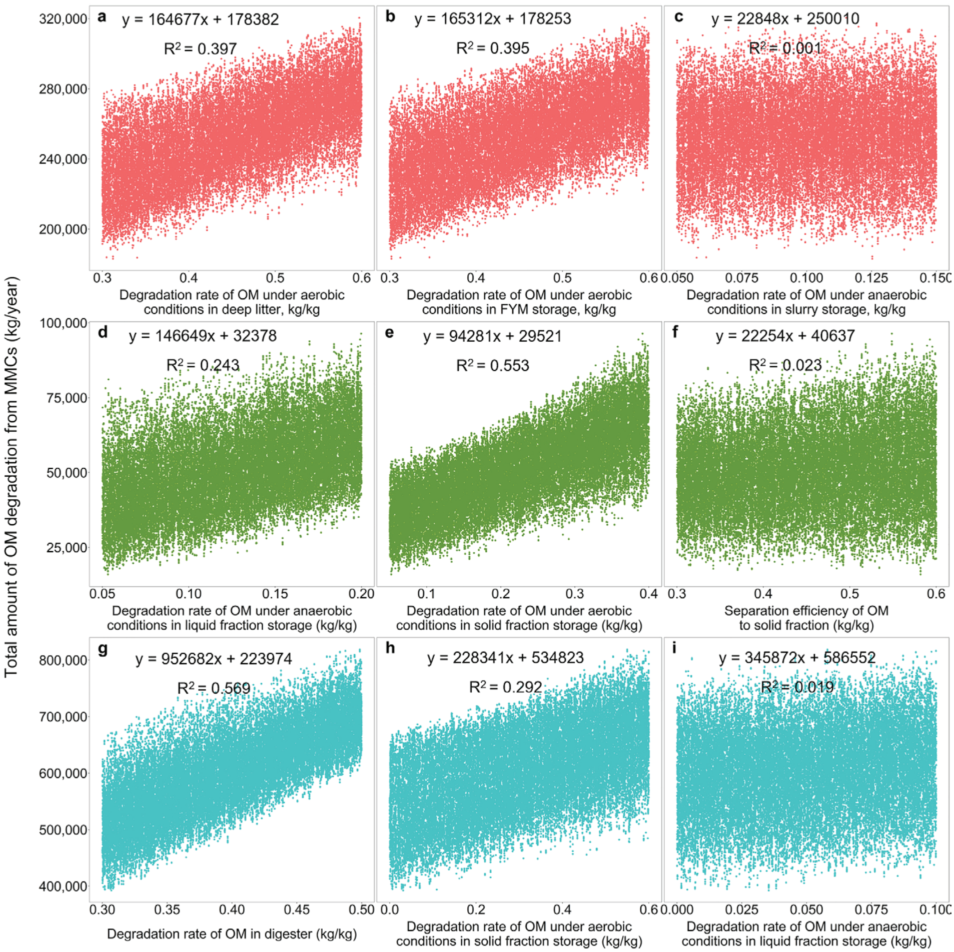

Figure 6.2. Relationships between input parameters of different manure management facilities and total organic matter (OM) degradation from manure management chains (MMCs). Different colors represent different MMCs of Scenarios 1 (red, a-c), 4 (green, d-f) and 6 (blue, g-i).

Table 6.3. Sensitivity index (%), including first-order sensitivity index (FSI) and total sensitivity index (TSI), of input parameters on total OM degradation from MMCs of Scenarios 1, 4 and 6. The parameters with FSI and TSI values higher than 10% are indicated in bold.

Manure management facility	Parameters	Scenario 1			Scenario 4			Scenario 6		
		Range	FSI	TSI	Range	FSI	TSI	Range	FSI	TSI
Deep litter	Fraction of substrate stored under oxic conditions	0.5–1.0	2.9	6.5						
	Degradation rate of OM under oxic conditions	0.3–0.6	**40.5**	**42.1**						
	Degradation rate of OM under anoxic conditions	0.2–0.4	-0.3	2.6						
FYM storage	Fraction of substrate stored under oxic conditions	0.5–1.0	5.3	6.6						
	Degradation rate of OM under oxic conditions	0.3–0.6	**41.3**	**42.7**						
	Degradation rate of OM under anoxic conditions	0.2–0.4	-0.3	2.6						
Slurry storage	Fraction of substrate stored under oxic conditions	0.0–0.1	-1.6	0.0	0.0–0.2	-0.8	0.2	0.0–0.2	0.0	0.0
	Degradation rate of OM under oxic conditions	0.0–0.1	-1.6	0.0	0.0–0.1	-0.8	0.4	0.00–0.05	-1.2	0.0
	Degradation rate of OM under anoxic conditions	0.05–0.15	-1.3	0.2	0.00–0.05	5.6	6.4	0.00–0.05	-1.3	0.1
Anaerobic digester	Degradation rate of OM under anoxic conditions							0.3–0.5	**56.1**	**56.2**
Digested slurry storage	Fraction of substrate stored under oxic conditions							0.0–0.1	0.1	0.0
	Degradation rate of OM under oxic conditions							0.0–0.1	1.4	0.0
	Degradation rate of OM under anoxic conditions							0.0–0.1	5.0	5.3
Liquid fraction storage	Fraction of substrate stored under oxic conditions				0.0–0.1	0.9	0.1	0.0–0.1	0.0	0.0
	Degradation rate of OM under oxic conditions				0.0–0.1	0.8	0.0	0.0–0.3	0.1	0.1
	Degradation rate of OM under anoxic conditions				0.05–0.20	**25.2**	**25.9**	0.0–0.1	3.0	2.1
Solid fraction storage	Fraction of OM to Solid fraction				0.3–0.6	1.6	4.8	0.3–0.6	2.9	3.7
	Fraction of substrate stored under oxic conditions				0.5–1.0	3.7	7.6	0.5 – 1.0	1.5	2.8
	Degradation rate of OM under oxic conditions				0.05–0.40	**53.9**	**59.7**	0.0 – 0.6	**29.7**	**31.4**
	Degradation rate of OM under anoxic conditions				0.00–0.15	2.1	1.6	0.0 – 0.2	0.6	0.5

6.3.2.2 Total inorganic N losses

In Scenario 1, changes of mineralization rate of organic N in FYM storage contributed more than 50% variance of total inorganic N losses, with higher mineralization rate leading to larger inorganic N losses (Fig. 6.3b). Conversely, the immobilization rate of inorganic N in deep litter had a negative effect on total inorganic N losses (Fig. 6.3a). In total, these two parameters contributed more than 70% of variance of total inorganic N losses, much higher than contributions of loss coefficients of NH_3-N during manure storage and application (Table A6.9). On the contrary, in Scenarios 4 and 6, the total inorganic N losses were more sensitive to changes in loss coefficients of NH_3-N during storage and application than mineralization rates of organic N. In total, loss coefficients of NH_3-N during liquid fraction storage and application resulted in more than 60% of variance, which was four times higher than the contribution of mineralization rates of organic N (Table A6.9). We did not observe significant contribution of separation efficiency of N to variance of total inorganic N losses in Scenarios 4 and 6 because only a small percentage of inorganic N (less than 10%) would be allocated to the solid fraction.

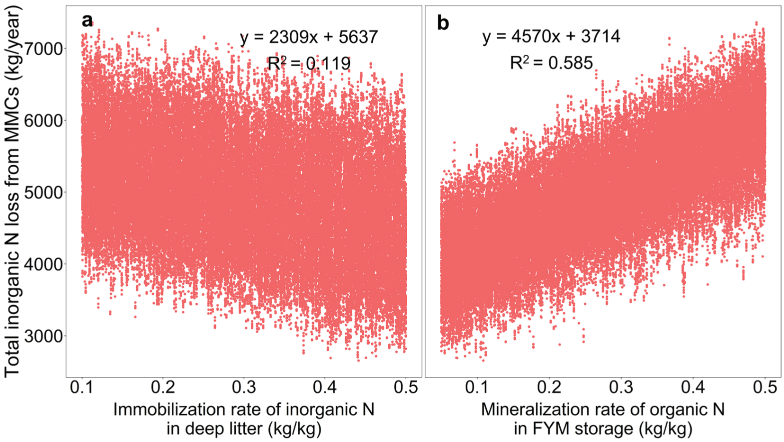

Figure 6.3. Influence of the immobilization rate of inorganic N in deep litter (a, with FSI and TSI values of 13.6% and 12.6%, respectively) and the mineralization rate of organic N in farmyard manure storage (b, with FSI and TSI values of 57.5% and 58.8%, respectively) on total inorganic N losses from the MMC of Scenario 1.

6.3.2.3 Total P and K loss

Runoff and leaching losses during solid manure storage are the primary loss pathways of P and K from MMCs. We observed significant contribution of loss coefficients of leaching and runoff from solid manure storage to variance of total P and K losses from the MMCs, with FSI values ranging from 40% to 99%. In MMCs with SLS (Scenarios 4 and 6), both separation efficiency of P of separator and loss coefficient of P from solid fraction storage were influential, with two times higher FSI value for loss coefficient of P from solid fraction storage than separation efficiency of P (Table A6.10). The separation efficiency of K was as important to total K losses as the loss coefficient of runoff and leaching from solid fraction storage. The differences between FSI and TSI values of separation efficiency of P or K and loss coefficient of P or K from solid fraction storage indicated interactions between these two parameters, contributing more than 10% of variance to total P or K losses from MMCs with SLS. The amounts of P or K in solid fractions are the prerequisite for the losses of P or K, with the more P or K staying in solid fractions leading to the more P or K losses by runoff and leaching during storage. The parameters relating to AD in Scenario 6 did not influence system level losses of P and K.

6.3.2.4 GHG emissions

The emissions of GHG from MMCs are related to C losses by emissions in the form of CH_4, and to N_2O emissions. Configurations of the manure management facilities in MMCs had important effects on GHG emissions, with stronger effects of parameters in earlier facilities of MMCs than in later facilities. In Scenario 1, we observed important effects of degradation of C in deep litter on GHG emissions from the whole MMC. About more than 70% of variance of total GHG emissions could be explained by changes of degradation rate of OM under aerobic conditions in deep litter and changes of fraction of CH_4-C in total C loss from deep litter. In Scenario 4, a larger contribution of loss coefficient of N_2O during separated liquid fraction storage to total variance of GHG emissions from the MMC was observed even with a small varying range from 0.0 to 0.1 for the loss coefficient (Table A6.11). The degradation rate of C under anaerobic conditions in separated liquid fraction and the fraction of C lost as CH_4-C emissions were also influential, in total contributing about 40% of variance to total GHG emissions from the MMC. The AD in Scenario 6 played the most important role in determining the uncertainty of GHG emissions from the MMC. The highest TSI was observed for the effect of the fraction of CH_4 produced in a digester that would be combusted (Table A6.11), with higher fraction of CH_4 combusted resulting in lower GHG emissions (Fig. A6.4b). Besides, the degradation rate of C in AD could be influential, with higher values resulting in larger variations of GHG emissions from the MMC (Fig. A6.4a).

6.4 Discussion

6.4.1 Impacts of manure management facilities on the fates of manure constituents

With the newly developed FarmM3 model we quantified the degradation and losses of OM, C, N, P and K from several contrasting manure management scenarios. Comparisons among these manure management scenarios demonstrated the impact of choice and configurations of manure management facilities on flows and losses of manure constituents throughout the whole MMCs. The MMCs with deep litter and AD had higher OM degradation, C losses and GHG emissions because of the added straw. Application of SLS could reduce GHG emissions by 40 to 60%. These reductions were due to the lower CH_4 and N_2O emissions from separated liquid fraction storage as less volatile solids entered the liquid fraction, so no natural crust formed during the storage (Aguirre-Villegas et al., 2014; Holly et al., 2017). The influence of SLS on NH_3 emissions from MMCs was affected by manure management facilities before and after SLS. We observed a greater reduction in NH_3 emissions when applying SLS to digested slurry than to raw slurry. The decrease of NH_3 emissions from MMCs with SLS might be because of less NH_3 emissions from separated solid fraction storage and from separated liquids application due to the quick infiltration of ammoniacal N in liquids to the soil (Aguirre-Villegas et al., 2014). In contrast, Kupper et al. (2020) found that SLS caused higher losses for NH_3 due to the absence of a surface crust during separated liquid fraction storage (Baldé et al., 2018). The balance between increased NH_3 emissions from separated liquid fraction storage and reduced NH_3 emissions from solid fraction storage and from liquid manure application resulted in different effects of SLS on NH_3 emissions from the whole MMC. This study also showed that, compared to the MMCs without SLS, separated solid fraction storage might increase the risks of P and K losses through leaching and runoff. This highlights the importance of improving management of solid manure storage to reduce nutrient losses.

6.4.2 The important parameters of determining losses of manure constituents in complex MMCs

The FarmM3 model enabled the identification of the most important parameters determining losses of manure constituents in complex MMCs through WS sensitivity analysis. The degradation rates of OM under aerobic conditions during solid manure storage and under anaerobic conditions during liquid manure storage were the most important influencing parameters. The important parameters for determining GHG emissions varied among MMCs, indicating the effects of configurations of manure management facilities in MMCs on GHG emissions.

In the MMCs with solid manure storage (e.g., the deep litter system), the immobilization and mineralization rate between inorganic and organic N were more influential than the loss coefficients of NH_3-N. Various studies have highlighted the importance of considering N transformations in solid manure storage when using a inorganic N flow approach to estimate N losses from MMCs (Dämmgen and Hutchings, 2008; Velthof et al., 2012). But the quantitative effects of immobilization and mineralization rates of organic N and inorganic N on total N losses from MMCs were rarely investigated, which prevents us from comparing with other studies. For liquid manure management systems with SLS or AD, the loss coefficients of NH_3-N during liquid manure storage and application contributed more to the variance of total N losses, which is in accordance with the study of Aguirre-Villegas et al. (2014). The separation efficiencies of organic N and inorganic N did not influence the total N losses from MMCs, which agrees with the study of Perazzolo et al. (2017). This was mainly because of the low separation efficiency (0 to 10%) of inorganic N to solid fraction and the negligible inorganic N losses from solid fraction storage (Aguirre-Villegas et al., 2019). Different from N, total losses of P and K from MMCs were sensitive to changes of separation efficiencies of P and K from SLS, with more contribution of separation efficiency of K to total losses than separation efficiency of P.

6.4.3 Comparison with other modelling approaches

The FarmM3 model in this study was developed based on a modular concept and a mass balance approach. Compared to existing modelling approaches that only considered the traditional manure management strategies (e.g., storage and application), this model has more flexibility that allows to integrate more alternative manure management facilities in a desired and feasible sequential order from excretion to application to cover complex MMCs. The results of quantifying flows of manure constituents along MMCs with contrasting manure management facilities in this study verified the feasibility of applying the FarmM3 model in various MMCs in on-farm settings. Besides, this model is able to quantify flows of different manure constituents (i.e., OM, C, N, P and K) throughout MMCs, which might make it as a helpful tool for comprehensively evaluating the effects of manure management options on degradation and losses of different manure constituents and for identifying trade-offs among these manure constituents.

6.4.4 Limitations

The developed FarmM3 model quantified the flows and losses of manure OM, C, N, P and K throughout MMCs based on empirical values of input parameters from publications. However, in practical situations, these input parameters, including loss coefficients, emissions factors and performance parameters of manure management facilities might be affected by management practices and environmental factors (i.e., temperature, rainfall etc.). In this regard, this model still has its limitations regarding selecting suitable parameters to specifically meet the on-farm situation although the simplicity of calculations with emission factors might make it easy to use. These limitations can be reduced further by identifying the most influential input parameters through sensitivity analysis, and by improving the accuracy of these important parameters by developing mechanistic estimation models or by validating estimated results using data from farm measurements.

6.4.5 Implications

The developed FarmM3 model can be used as a helpful tool for quantifying degradation and losses of different manure constituents (OM, C, N, P and K) in complex MMCs. The results can contribute to understanding the effects of various manure management facilities on the flows and losses of manure OM, C, N, P and K through the whole MMCs. It can support farm managers in decision making on designing and optimizing manure management strategies by assessing trade-offs among these different environmental indicators. By identifying the most important parameters determining losses from various MMCs, it also helps researchers to identify future research priorities in estimating loss coefficients of different manure constituents from various manure management facilities.

6.4.6 Future research

To further validate and improve the accuracy of model estimates, on-farm measurements on degradation and losses of different manure constituents from whole MMCs might be necessary and helpful. For future works, the FarmM3 model can be integrated into whole-farm models such as the FarmDESIGN model (Groot et al., 2012) to improve flexibility of model application in dairy farming systems with complex MMCs. In addition to evaluating environmental performance in terms of degradation of OM, nutrient losses and gaseous emissions from MMCs, this model could be extended to allow a multi-criteria decision-making analysis for design and optimize manure management scenarios considering economic aspects, such as investment and operation costs of management facilities.

6.5 Conclusions

This study developed the FarmM3 model, and quantifiably compared diverse manure management facilities, and the relative degradation and losses of OM, C, N, P and K throughout their MMCs. The results showed that, compared to other MMCs, the MMCs with deep litter and AD yielded higher OM degradation, C losses and GHG emissions due to the more added straw. This implied the positive relationships between the quantity of manure dry matter and OM degradation, and GHG emissions. Further, the MMC with deep litter showed reduced NH_3 emission, but increased GHG emissions due to the pollution swapping caused by adding straw. Application of SLS could reduce GHG emissions, but its effect on NH_3 emissions varied depending on the characteristics of the separated slurry. A larger reduction in NH_3 emissions was observed when applying SLS to digested slurry than to raw slurry. The sequence of manure management facilities in MMCs influenced the flows and losses of constituents. For example, our results showed greater reductions in GHG and NH_3 emissions when applying SLS after AD than applying SLS before AD.

Results of WS sensitivity analysis showed the most important parameters for determining GHG emissions varied among MMCs, indicating the effects of configurations of manure management facilities in MMCs on GHG emissions. For N losses, the immobilization and mineralization rates between inorganic and organic N were more influential than loss coefficients of NH_3-N in the MMC with deep litter. In liquid manure systems, the loss coefficients of NH_3-N from liquid manure storage and application were more influential than from solid fractions. The separation efficiencies of organic N and inorganic N did not influence total N losses from MMCs with SLS. In contrast, the separation efficiencies of P and K from SLS were influential to total losses of P and K from MMCs with SLS.

Our modelling approach could contribute to understanding the role of manure management facilities in farm nutrient management planning and could be helpful for farmers, researchers and policy makers to decide how to improve manure management systems at farm level.

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Supplementary material

1. Model calculations

The flows and losses of constituent X (including OM, C, Organic N, inorganic N, P and K) throughout MMCs commences with a specified quantity of produced manure (m_{Manure}, kg/year). The amounts of constituent X in manure are calculated using the contents of DM, ash, C, N, P and K. The content of C is assumed as a fraction of OM. The amount of inorganic N is calculated using the fraction of inorganic N in total N.

$$m_{Manure\_OM} = m_{Manure} \times DM\_content \times (1 - Ash\_content)$$

$$m_{Manure\_C} = m_{Manure\_OM} \times C\_content$$

$$m_{Manure\_N} = m_{Manure} \times DM\_content \times N\_content$$

$$m_{Manure\_Inorganic\_N} = m_{Manure\_N} \times N\_inorganic$$

$$m_{Manure\_Organic\_N} = m_{Manure\_N} \times (1 - N\_inorganic)$$

$$m_{Manure\_P} = m_{Manure} \times DM\_content \times P\_content$$

$$m_{Manure\_K} = m_{Manure} \times DM\_content \times K\_content$$

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Similarly, additional amounts of constituent X from bedding straw or co-digestion straw are calculated by multiplying the quantity added (m_{Added}, kg/year) with contents of OM, C, organic N, inorganic N, P and K.

The flows of constituent X between different manure management facility Y (i.e., dairy barn, deep litter (DL), slurry storage tank, solid manure heap, composting, SLS, AD etc.) of MMCs are estimated using fractions of constituent X flowing to the facility (*Fraction_X_Y*, kg/kg). The total amount of constituent X into facility Y is calculated by summing up the amounts of constituent X from manure and added materials.

$$m_{Manure\_X\_Y} = m_{Manure\_X} \times Fraction\_X\_Y$$

$$m_{X\_Y} = m_{Manure\_X\_Y} \times m_{Added\_X\_Y}$$

The amounts of constituent X split to two new pools (i.e., Pool 1 and Pool 2) by separators are estimated using the fractions of constituent X allocated to the new pools (*X_to_Pool_1*, kg/kg).

$$m_{X\_Pool\_1} = m_{X\_Y} \times X\_to\_Pool\_1$$

$$m_{X\_Pool\_2} = m_{X\_Y} \times (1 - X\_to\_Pool\_1)$$

Degradation and losses of constituent X from different manure facility Y are estimated using the quantity in the facility and the conversion and losses coefficients of constituent X in the facility. For the degradation of OM and C, the fractions of oxic and anoxic conditions in the component and corresponding degradation rates under different conditions are used to calculate the degradation amounts of OM and C. The emissions of CH_4-C and CO_2-C from the degradation of C are calculated using the fraction of CH_4-C in the total C loss.

$$m_{OM\_Degraded\_Y} = m_{OM\_Y} \times Fraction\_Oxic\_Y \times OM\_Degradation\_Oxic\_Y + m_{OM\_Y} \times (1 - Fraction\_Oxic\_Y) \times OM\_Degradation\_Anoxic\_Y$$

$$m_{C\_Degraded\_Y} = m_{C\_Y} \times Fraction\_Oxic\_Y \times OM\_Degradation\_Oxic\_Y + m_{C\_Y} \times (1 - Fraction\_Oxic\_Y) \times C\_Degradation\_Anoxic\_Y$$

$$m_{CH_4-C} = m_{C\_Degraded} \times Fraction\_Methane\_C$$

$$m_{CO_2-C} = m_{C\_Degraded} \times (1 - Fraction\_Methane\_C)$$

The immobilization and mineralization rates between organic N and inorganic N are used to estimate the amounts of conversion between inorganic N and organic N in the facility. The losses of gaseous N, including NH_3-N, N_2O-N, NO-N and N_2-N, and leaching N are extracted from the amount of inorganic N. The loss by runoff N is estimated and extracted from both inorganic N and organic N pools.

$$m_{Inorganic\_N\_Y} = (m_{Manure\_Inorganic\_N\_Y} + m_{Added\_Inorganic\_N\_Y}) \times (1 - Immobilization\_N) + (m_{Manure\_Oragnic\_N\_Y} + m_{Added\_Organic\_N\_Y}) \times Mineralization\_N$$

$$m_{Organic\_N\_Y} = (m_{Manure\_Inorganic\_N\_Y} + m_{Added\_Inorganic\_N\_Y}) \times (1 - Mineralization\_N) + (m_{Manure\_Oragnic\_N\_Y} + m_{Added\_Organic\_N\_Y}) \times Immobilization\_N$$

$$m_{Inorganic\_N\_Lost\_Y} = m_{Inorganic\_N\_Y} \times \sum(Z\_Loss\_Coefficient\_Y)$$

In which Z represents NH_3-N, N_2O-N, N_2-N, NO-N, N_Leaching and N_Runoff;

$$m_{Organic\_N\_Lost\_Y} = m_{Organic\_N\_Y} \times N\_Runoff\_Loss\_Coefficient\_Y$$

Similarly, the losses of P or K by leaching and runoff from the facility Y are estimated by multiplying the amounts of P or K to the facility Y with loss coefficients of leaching and runoff.

$$m_{P\_Lost\_Y} = m_{P\_Y} \times P\_Leaching\_Runoff\_Y$$

$$m_{K\_Lost\_Y} = m_{K\_Y} \times K\_Leaching\_Runoff\_Y$$

The accumulated degradation and losses of constituent X from the whole MMC is derived by summing up the losses in different manure facilities Y. The output of constituent X from the MMC is calculated by extracting the amount of losses from total input amounts to the MMC.

$$m_{X\_Lost\_Y} = \sum m_{X\_Lost\_Y}$$

$$m_{X\_Output} = m_{X\_Y} - m_{X\_Lost}$$

Based on the emissions of CH_4 and N_2O, the total GHG emissions expressed in CO_2 equivalents is calculated by converting the CH_4 and N_2O emissions to $CO_{2\text{-eq}}$ emissions based on the Global Warming Potential (GWP) conversion factor of 28 for CH_4 and 265 for N_2O (Myhre et al., 2013). The total CO_2 equivalent of GHG emissions was calculated using the following equation:

$$E_{CO_2\text{-eq}} = 28 \times E_{CH_4,\ \text{non-combusted}} + 265 \times E_{N_2O}$$

where $E_{CO_2\ eq}$ is the total CO_2 equivalent of GHG emissions, kg CO_2-eq/year; $E_{CH_4,non-combusted}$ is the total CH_4 emissions that are not used as fuel, kg CH_4/year and E_{N_2O} is the total N_2O emissions from the MMC, kg N_2O/year.

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Table A6.1. Reported approaches and models for estimating degradation and losses of manure constituents from manure management chains.

Reference	Stages of manure management chains	Manure constituents
Webb and Misselbrook, 2004	manure excretion (housing/grazing/outdoors), storage and application	NH_3
Olesen et al., 2006	manure excretion (housing), storage and application	C and N
Dämmgen and Hutchings, 2008	manure excretion (housing/outdoor yards/feedlots/pasture), storage and application	N
Del Prado et al., 2011; Pardo et al., 2017	manure excretion (housing), storage, treatment (anaerobic digestion) and application	N, CH_4
Li et al., 2012	manure excretion (feedlot), treatment (compost, lagoon, anaerobic digestion) and application	C, N, P
Rotz et al., 2012	manure excretion (barn/pasture), storage, transport, treatment (anaerobic digestion, composting), application	C, N, P and K
Khalil et al., 2019	manure excretion, storage, treatment (anaerobic digestion, coarse fiber removal, fine solid removal separation and nutrient recovery) and application	C, N and P
Sefeedpari et al., 2019	manure excretion, treatment (anaerobic digestion, mechanical separation and composting)	C, N, P and K

Table A6.2. A summary of parameters in the FarmM3 model.

Modules	Parameters	Description
Input Module	DM_content	Dry matter content in fresh material
	Ash_content	Ash content in dry matter
	C_content	Carbon content in organic matter
	N_content	Nitrogen content in dry matter
	P_content	Phosphorus content in dry matter
	K_content	Potassium content in dry matter
	N_inorganic	Inorganic nitrogen as a fraction of total nitrogen
Pool Module	Mineralization	Fraction of organic nitrogen mineralized
	Immobilization	Fraction of inorganic nitrogen immobilized
	NH_3_N_Loss_Coefficient	Fraction of inorganic nitrogen lost as NH_3-N
	N_2O_N_Loss_Coefficient	Fraction of inorganic nitrogen lost as N_2O-N
	NO_N_Loss_Coefficient	Fraction of inorganic nitrogen lost as NO-N
	N_2_N_Loss_Coefficient	Fraction of inorganic nitrogen lost as N_2-N
	Fraction_Oxic	Fraction of substrate stored under aerobic conditions
	C_Degradation_Oxic	Fraction of organic matter or organic carbon that is degraded under aerobic conditions
	C_Degradation_Anoxic	Fraction of organic matter or organic carbon that is degraded under anaerobic conditions
	Fraction_Methane_C	Fraction of release carbon that is emitted in methane as opposed to CO_2
	Fraction_CH_4_Combusted	Fraction of emitted methane that is combusted
	N_Leaching	Fraction of inorganic nitrogen lost by leaching and runoff
	N_Runoff	Fraction of inorganic and organic nitrogen lost by run-off
	P_Leaching_Runoff	Fraction of phosphorus lost by leaching and runoff
	K_Leaching_Runoff	Fraction of potassium lost by leaching and runoff
Separator Module	DM_to_Pool_1	Fraction of dry matter to pool 1
	Ash_to_Pool_1	Fraction of ash to pool 1
	C_to_Pool_1	Fraction of carbon to pool 1
	N_to_Pool_1	Fraction of nitrogen to pool 1
	P_to_Pool_1	Fraction of phosphorus to pool 1
	K_to_Pool_1	Fraction of potassium to pool 1
	N_inorganic_to_Pool_1	Fraction of inorganic nitrogen to pool 1

Modules	Parameters	Description
Application Module	NH_3_N_Loss_Coefficient	Fraction of inorganic nitrogen lost as NH_3-N
	N_2O_N_Loss_Coefficient	Fraction of inorganic nitrogen lost as N_2O-N
	NO_N_Loss_Coefficient	Fraction of inorganic nitrogen lost as NO-N
	N_2_N_Loss_Coefficient	Fraction of inorganic nitrogen lost as N_2-N
	N_Leaching	Fraction of inorganic nitrogen lost by leaching
	N_Runoff	Fraction of inorganic and organic nitrogen lost by runoff
	P_Leaching_Runoff	Fraction of phosphorus lost by leaching and runoff
	K_Leaching_Runoff	Fraction of potassium lost by leaching and runoff

Table A6.3. Amounts and compositions of manure excreted and straw added to manure management chains.

Scenarios	Inputs	Amount, kg in FM	DM content, kg/kg FM	Ash content, kg/kg DM	C content, kg/kg OM	N content, g/kg DM	P content, g/kg DM	K content, g/kg DM	N inorganic, kg/kg N
1	Manure	1201018	0.15	0.124	0.5	68.0	7.6	86.0	0.6
	Bedding straw	401500	0.85	0.140	0.5	5.0	0.8	15.0	0.0
2	Manure	1201018	0.15	0.124	0.5	68.0	7.6	86.0	0.6
	Bedding straw	182500	0.85	0.140	0.5	5.0	0.8	15.0	0.0
3	Manure	1201018	0.15	0.124	0.5	68.0	7.6	86.0	0.6
	Bedding straw	182500	0.85	0.140	0.5	5.0	0.8	15.0	0.0
4	Manure	1201018	0.15	0.124	0.5	68.0	7.6	86.0	0.6
	Bedding straw	182500	0.85	0.140	0.5	5.0	0.8	15.0	0.0
5	Manure	1201018	0.15	0.124	0.5	68.0	7.6	86.0	0.6
	Bedding straw	182500	0.85	0.140	0.5	5.0	0.8	15.0	0.0
	Co-digestion straw	1209719	0.85	0.140	0.5	9.9	1.4	16.0	0.0
6	Manure	1201018	0.15	0.124	0.5	68.0	7.6	86.0	0.6
	Bedding straw	182500	0.85	0.140	0.5	5.0	0.8	15.0	0.0
	Co-digestion straw	1209719	0.85	0.140	0.5	9.9	1.4	16.0	0.0
7	Manure	1201018	0.15	0.124	0.5	68.0	7.6	86.0	0.6
	Bedding straw	182500	0.85	0.140	0.5	5.0	0.8	15.0	0.0
	Co-digestion straw	1209719	0.85	0.140	0.5	9.9	1.4	16.0	0.0

Note: FM represents fresh matter; DM represents dry matter; OM represents organic matter.

Table A6.4. Sensitivity index (%), including first-order sensitivity index (FSI) and total sensitivity index (TSI), of input parameters on total inorganic N loss, OM degradation and GHG emissions from the MMC of Scenario 2.

Parameters	Nmin		OM		GHG	
	FSI	TSI	FSI	TSI	FSI	TSI
Barn: Loss coefficient of NH_3-N	0.3	0.2	—	—	—	—
Anaerobic lagoon: Mineralization rate of organic N	**12.3**	**13.5**	—	—	—	—
Anaerobic lagoon: Loss coefficient of NH_3-N	**70.5**	**74.7**	—	—	—	—
Anaerobic lagoon: Loss coefficient of N_2O_N	1.6	1.5	—	—	**46.3**	**47.1**
Slurry application: Loss coefficient of NH_3-N	12	**15.1**	—	—	—	—
Anaerobic lagoon: Fraction of slurry stored under oxic conditions	—	—	1.9	5.1	0.6	1.1
Anaerobic lagoon: Degradation rate of C under oxic conditions	—	—	6.9	9.4	1.9	2.1
Anaerobic lagoon: Degradation rate of C under anoxic conditions	—	—	**88**	**89.3**	**17.5**	**20.0**
Anaerobic lagoon: Fraction of C lost in CH_4-C	—	—	-0.7	0	**28.9**	**31.4**

Notes: The parameters with FSI and TSI values more than 10% are indicated in bold.

Table A6.5. Sensitivity index (%), including first-order sensitivity index (FSI) and total sensitivity index (TSI), of input parameters on total inorganic N loss, OM degradation, GHG emissions, total P and K losses from the MMC of Scenario 3.

Parameters	Nmin		OM		GHG		TP		TK	
	FSI	TSI	FSI	TSI	FSI	TSI	FSI	TSI	FSI	TSI
Barn: Loss coefficient of NH_3-N	-0.7	0.1	—	—	—	—	—	—	—	—
Yard: Mineralization rate of organic N	2.3	2.5	—	—	—	—	—	—	—	—
Yard: Loss coefficient of NH_3-N	1.4	2.8	—	—	—	—	—	—	—	—
Anaerobic lagoon: Mineralization rate of organic N	**59.7**	**61.5**	—	—	2.8	6.9	—	—	—	—
Anaerobic lagoon: Loss coefficient of NH_3-N	4.5	6.0	—	—	—	—	—	—	—	—
Anaerobic lagoon: Loss coefficient of N_2O-N	0.8	1.2	—	—	**46.3**	**49.4**	—	—	—	—
Anaerobic lagoon: Loss coefficient of NO-N	1.1	1.2	—	—	—	—	—	—	—	—
Anaerobic lagoon: Loss coefficient of N_2-N	0.8	1.2	—	—	—	—	—	—	—	—
Solid manure storage: Mineralization rate of organic N	-1.3	0.7	—	—	—	—	—	—	—	—
Solid manure storage: Loss coefficient of NH_3-N	-1.5	0.6	—	—	—	—	—	—	—	—
Solid manure storage: Loss coefficient of N_2O-N	-2.2	0.1	—	—	—	—	—	—	—	—
Solid manure storage: Loss coefficient of N_2-N	-1.3	0.7	—	—	—	—	—	—	—	—
Slurry application: Loss coefficient of NH_3-N	**17.5**	**21.1**	—	—	—	—	—	—	—	—
Solid manure application: Loss coefficient of NH_3-N	2.9	4.4	—	—	—	—	—	—	—	—
Anaerobic lagoon: Fraction of slurry stored under oxic conditions	—	—	2.1	1.6	-0.7	0.4	—	—	—	—
Anaerobic lagoon: Degradation rate of C under oxic conditions	—	—	1.8	1.5	-0.5	0.4	—	—	—	—

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Parameters	Nmin		OM		GHG		TP		TK	
	FSI	TSI	FSI	TSI	FSI	TSI	FSI	TSI	FSI	TSI
Anaerobic lagoon: Degradation rate of C under anoxic conditions	—	—	**24.0**	**23.4**	4.0	6.6	—	—	—	—
Anaerobic lagoon: Fraction of C lost in CH_4-C	—	—	0.7	0	**30.4**	**32.2**	—	—	—	—
Yard: Fraction of solids stored under oxic conditions	—	—	1.2	1.5	-0.6	0	—	—	—	—
Yard: Degradation rate of C under oxic conditions	—	—	3.1	3	-0.5	0.1	—	—	—	—
Yard: Degradation rate of C under anoxic conditions	—	—	0.9	0.7	-0.6	0	—	—	—	—
Yard: Fraction of C lost in CH_4-C	—	—	0.4	0	-0.6	0	—	—	—	—
Solid manure storage: Fraction of solids stored under oxic conditions	—	—	1	2.5	-0.6	0.1	—	—	—	—
Solid manure storage: Degradation rate of C under oxic conditions	—	—	**64.6**	**64.6**	0.9	2.1	—	—	—	—
Solid manure storage: Degradation rate of C under anoxic conditions	—	—	2.8	2.4	-0.9	0.1	—	—	—	—
Solid manure storage: Fraction of C lost in CH_4-C	—	—	—	—	-1.7	0.3	—	—	—	—
Yard: Leaching and runoff loss coefficient of P	—	—	—	—	—	—	**99.7**	**100.8**	—	—
Solid manure storage: Leaching and runoff loss coefficient of P	—	—	—	—	—	—	-0.7	0.4	—	—
Yard: Leaching and runoff loss coefficient of K	—	—	—	—	—	—	—	—	**99.6**	**99.8**
Solid manure storage: Leaching and runoff loss coefficient of K	—	—	—	—	—	—	—	—	0.2	0.4

Notes: The parameters with FSI and TSI values more than 10% are indicated in bold.

Table A6.6. Sensitivity index (%), including first-order sensitivity index (FSI) and total sensitivity index (TSI), of input parameters on total inorganic N loss, OM degradation and GHG emissions from the MMC of Scenario 5.

Parameters	Nmin		OM		GHG	
	FSI	TSI	FSI	TSI	FSI	TSI
Barn: Loss coefficient of NH_3-N	-5.3	0	—	—	—	—
Slurry prestorage: Mineralization rate of organic N	2.2	0.6	—	—	-3.1	0
Slurry prestorage: Loss coefficient of NH_3-N	2.1	0.5	—	—	—	—
Slurry prestorage: Loss coefficient of N_2O-N	2.2	0.1	—	—	-2.9	0.3
Slurry prestorage: Loss coefficient of NO-N	1.9	0.1	—	—	—	—
Slurry prestorage: Loss coefficient of N_2-N	2.1	0.1	—	—	—	—
Digester: Mineralization rate of organic N	7.9	6.1	—	—	-3.2	0
Digested slurry storage: Mineralization rate of organic N	3.4	2.4	—	—	-3.1	0
Digested slurry storage: Loss coefficient of NH_3-N	**65.8**	**67.7**	—	—	—	—
Digested slurry storage: Loss coefficient of N_2O-N	-4.1	1.4	—	—	-1.7	1.1
Digested slurry application: Loss coefficient of NH_3-N	**16.9**	**20.4**	—	—	—	—
Slurry prestorage: Fraction of slurry stored under oxic conditions	—	—	-7.1	0	-1.6	0
Slurry prestorage: Degradation rate of C under oxic conditions	—	—	-7.1	0	-1.6	0
Slurry prestorage: Degradation rate of C under anoxic conditions	—	—	-6.8	0.4	-1.6	0
Slurry prestorage: Fraction of C lost in CH_4-C	—	—	-7.3	0	-1.6	0
Digester: Degradation rate of C under anoxic conditions	—	—	**73.7**	**76.2**	2.8	7.1
Digester: Fraction of C lost in CH_4-C	—	—	-7.3	0	3.3	7.2
Digester: Fraction of produced CH_4 Combusted	—	—	-7.3	0	**83.2**	**90.0**
Digested slurry storage: Fraction of slurry stored under oxic conditions	—	—	-7.3	0.1	-3.3	0
Digested slurry storage: Degradation rate of C under oxic conditions	—	—	-7.1	0.1	-3.3	0
Digested slurry storage: Degradation rate of C under anoxic conditions	—	—	**23.8**	**26.6**	-3.0	0.6
Digested slurry storage: Fraction of C lost in CH_4-C	—	—	-0.4	0	-2.8	0.6

Notes: The parameters with FSI and TSI values more than 10% are indicated in bold.

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Table A6.7. Sensitivity index (%), including first-order sensitivity index (FSI) and total sensitivity index (TSI), of input parameters on total inorganic N loss, OM degradation, GHG emissions, total P and K losses from the MMC of Scenario 7.

Parameters	Nmin		OM		GHG		TP		TK	
	FSI	TSI	FSI	TSI	FSI	TSI	FSI	TSI	FSI	TSI
Barn: Loss coefficient of NH_3-N	-1.1	0.1	—	—	—	—	—	—	—	—
Slurry prestorage: Mineralization rate of organic N	-2.9	1.1	—	—	—	—	—	—	—	—
Slurry prestorage: Loss coefficient of NH_3-N	-1	0.9	—	—	—	—	—	—	—	—
Separator: Separation efficiency of inorganic N to solid fraction	-2.9	0	—	—	—	—	—	—	—	—
Separator: Separation efficiency of organic N to solid fraction	-2.8	0.9	—	—	—	—	—	—	—	—
Liquid fraction storage: Mineralization rate of organic N	-5.1	0	—	—	—	—	—	—	—	—
Liquid fraction storage: Loss coefficient of NH_3-N	-0.7	0.2	—	—	—	—	—	—	—	—
Solid fraction storage: Mineralization rate of organic N	-4.2	0.8	—	—	—	—	—	—	—	—
Solid fraction storage: Loss coefficient of NH_3-N	-1	0	—	—	—	—	—	—	—	—
Solid fraction storage: Loss coefficient of N_2O-N	-4.7	0	—	—	—	—	—	—	—	—
Solid fraction storage: Loss coefficient of N_2-N	-4.7	0.1	—	—	—	—	—	—	—	—
Solid fraction application: Loss coefficient of NH_3-N	-5	0.2	—	—	—	—	—	—	—	—
Digester: Mineralization rate of organic N	0.3	5.7	—	—	—	—	—	—	—	—
Digested slurry storage: Mineralization rate of organic N	4.2	8.1	—	—	—	—	—	—	—	—
Digested slurry storage: Loss coefficient of NH_3-N	**47.3**	**52.3**	—	—	—	—	—	—	—	—
Digested slurry application: Loss coefficient of NH_3-N	**25.7**	**34.3**	—	—	—	—	—	—	—	—
Slurry prestorage: Fraction of slurry stored under oxic conditions	—	—	2.6	0	-0.4	0	—	—	—	—
Slurry prestorage: Degradation rate of C under oxic conditions	—	—	2.6	0	-0.4	0	—	—	—	—
Slurry prestorage: Degradation rate of C under anoxic conditions	—	—	7.7	0.2	-0.4	0	—	—	—	—
Separator: Separation efficiency of OM to solid fraction	—	—	3.9	1.5	-0.2	0	—	—	—	—
Separator: Separation efficiency of C to solid fraction	—	—	1.3	0	-0.2	0.4	—	—	—	—
Liquid fraction storage: Fraction of liquids stored under oxic conditions	—	—	1.8	0	-0.4	0	—	—	—	—
Liquid fraction storage: Degradation rate of C under oxic conditions	—	—	7.4	0	-0.4	0	—	—	—	—
Liquid fraction storage: Degradation rate of C under anoxic conditions	—	—	7.4	0	-0.2	0	—	—	—	—
Solid fraction storage: Fraction of solids stored under oxic conditions	—	—	1.6	0.6	-0.2	0	—	—	—	—
Solid fraction storage: Degradation rate of C under oxic conditions	—	—	**14**	**8.8**	-0.4	0	—	—	—	—
Solid fraction storage: Degradation rate of C under anoxic conditions	—	—	7.7	0.1	-0.4	0	—	—	—	—
Solid fraction storage: Fraction of C lost in CH_4-C	—	—	1.3	0	0.2	0	—	—	—	—

Parameters	Nmin		OM		GHG		TP		TK	
	FSI	TSI	FSI	TSI	FSI	TSI	FSI	TSI	FSI	TSI
Digester: Degradation rate of C under anoxic conditions	—	—	**80.1**	**72.7**	4.5	6.4	—	—	—	—
Digester: Fraction of C lost in CH_4-C	—	—	1.3	0	5.9	7.1	—	—	—	—
Digester: Fraction of produced CH_4 Combusted	—	—	1.3	0	**84.5**	**87.5**	—	—	—	—
Digested slurry storage: Fraction of slurry stored under oxic conditions	—	—	2	0.1	-0.2	0	—	—	—	—
Digested slurry storage: Degradation rate of C under oxic conditions	—	—	7.3	0	-0.4	0	—	—	—	—
Digested slurry storage: Degradation rate of C under anoxic conditions	—	—	**11.6**	**8.8**	0.3	0.1	—	—	—	—
Digested slurry storage: Fraction of C lost in CH_4-C	—	—	1.3	0	1.9	1.0	—	—	—	—
Separator: Separation efficiency of P to solid fraction	—	—	—	—	—	—	**33.4**	**44.7**	—	—
Solid fraction storage: Leaching and runoff loss coefficient of P	—	—	—	—	—	—	**56.9**	**68.4**	—	—
Separator: Separation efficiency of K to solid fraction	—	—	—	—	—	—	—	—	**35.4**	**61.7**
Solid fraction storage: Leaching and runoff loss coefficient of K	—	—	—	—	—	—	—	—	**40.0**	**66.3**

Notes: The parameters with FSI and TSI values more than 10% are indicated in bold.

Table A6.8. Pearson correlation coefficients among output variables from Winding Stairs sensitivity analysis of MMCs.

Manure management Scenario 1

	NH_3 emission	Total inorganic N loss	Total P loss	Total K loss	OM Degradation	Total C loss	GHG emission
NH_3 emission	1						
Total inorganic N loss	0.856	1					
Total P loss	0.009	0.007	1				
Total K loss	0.007	0.008	-0.001	1			
OM Degradation	-0.005	-0.010	0.003	0.003	1	1	
Total C loss	-0.005	-0.010	0.003	0.003	1	1	
GHG emission	0.052	0.138	-0.010	0.000	0.503	0.503	1

Manure management Scenario 2

	NH_3 emission	Total inorganic N loss	OM Degradation	Total C loss	GHG emission
NH_3 emission	1				
Total inorganic N loss	0.948	1			
OM Degradation	0.024	0.021	1	1	
Total C loss	0.024	0.021	1	1	
GHG emission	-0.008	0.186	0.440	0.440	1

Manure management Scenario 3

	NH_3 emission	Total inorganic N loss	Total P loss	Total K loss	OM Degradation	Total C loss	GHG emission
NH_3 emission	1						
Total inorganic N loss	0.907	1					
Total P loss	-0.017	-0.014	1				
Total K loss	-0.013	-0.011	0.006	1			
OM Degradation	0.007	0.003	0.002	0.009	1	1	
Total C loss	0.007	0.003	0.002	0.009	1	1	
GHG emission	-0.085	0.119	0.019	-0.001	0.147	0.147	1

Manure management Scenario 4

	NH_3 emission	Total inorganic N loss	Total P loss	Total K loss	OM Degradation	Total C loss	GHG emission
NH_3 emission	1						
Total inorganic N loss	0.914	1					
Total P loss	-0.012	-0.009	1				
Total K loss	0.008	0.004	0.015	1			
OM Degradation	-0.004	0.001	0.013	-0.005	1		
Total C loss	-0.004	0.001	0.013	-0.010	0.954	1	
GHG emission	-0.018	0.164	0.020	0.006	0.256	0.245	1

Manure management Scenario 5

	NH_3 emission	Total inorganic N loss	OM Degradation	Total C loss	GHG emission
NH_3 emission	1				
Total inorganic N loss	0.976	1			
OM Degradation	0.002	0.004	1	1	
Total C loss	0.002	0.004	1	1	
GHG emission	0.026	0.048	0.250	0.250	1

Manure management Scenario 6

	NH_3 emission	Total inorganic N loss	Total P loss	Total K loss	OM Degradation	Total C loss	GHG emission
NH_3 emission	1						
Total inorganic N loss	0.999	1					
Total P loss	-0.013	-0.014	1				
Total K loss	-0.012	-0.011	0.012	1			
OM Degradation	-0.008	-0.009	0.002	-0.011	1		
Total C loss	-0.005	-0.006	0.004	-0.012	0.964	1	
GHG emission	0.007	0.007	0.003	0.004	0.197	0.191	1

Manure management Scenario 7							
	NH_3 emission	Total inorganic N loss	Total P loss	Total K loss	OM Degradation	Total C loss	GHG emission
NH_3 emission	1						
Total inorganic N loss	0.999	1					
Total P loss	-0.017	-0.016	1				
Total K loss	-0.020	-0.019	0.000	1			
OM Degradation	0.008	0.007	0.010	-0.003	1		
Total C loss	0.009	0.008	0.007	-0.003	0.984	1	
GHG emission	0.011	0.012	0.010	0.007	0.191	0.194	1

Table A6.9. Sensitivity index (%), including first-order sensitivity index (FSI) and total sensitivity index (TSI), of input parameters on total inorganic N loss from MMCs Scenarios 1, 4 and 6.

Manure management facility	Parameters	Scenario 1			Scenario 4			Scenario 6		
		Range	FSI	TSI	Range	FSI	TSI	Range	FSI	TSI
Pasture	Loss coefficient of NH_3-N	0.06 – 0.18	1.3	0.4						
Milking parlour	Loss coefficient of NH_3-N	0.106 – 0.176	0.9	0.1						
Barn	Loss coefficient of NH_3-N				0.106 – 0.176	3.0	0.3	0.105 – 0.175	0.5	0.1
Deep litter	Immobilization rate of inorganic N	0.1 – 0.5	**13.6**	**12.6**						
	Loss coefficient of NH_3-N	0.155 – 0.259	1.0	0.1						
	Loss coefficient of N_2O-N	0.0 – 0.1	0.9	0.1						
FYM storage	Fraction of inorganic N to FYM storage	0.5 – 1.0	3.5	3.4						
	Fraction of organic N to FYM storage	0.9 – 1.0	0.9	0.2						
	Mineralization rate of organic N	0.05 – 0.50	**57.5**	**58.8**						
	Loss coefficient of NH_3-N	0.2 – 0.5	2.5	4.1						
	Loss coefficient of N_2O-N	0.0 – 0.1	-0.6	0.5						
	Loss coefficient of N_2-N	0.0 – 0.3	1.8	4.2						

Manure management facility	Parameters	Scenario 1			Scenario 4			Scenario 6		
		Range	FSI	TSI	Range	FSI	TSI	Range	FSI	TSI
Slurry storage	Mineralization rate of organic N	0.05 – 0.50	-0.9	1.1	0.0 – 0.3	11.6	9.0	0.0 – 0.3	2.7	1.0
	Loss coefficient of NH_3-N	0.158 – 0.474	1.6	2.7	0.1 – 0.3	4.9	3.8	0.1 – 0.3	1.0	1.2
Anaerobic digester	Mineralization rate of organic N							0.375 – 0.625	**12.2**	**11.5**
Digested slurry storage	Mineralization rate of organic N							0.0 – 0.1	1.3	0.5
	Loss coefficient of NH_3-N							0.1 – 0.3	6.7	6.3
Liquid fraction storage	Mineralization rate of organic N				0.05 – 0.50	**15.2**	**14.3**	0.0 – 0.5	9.7	9.9
	Loss coefficient of NH_3-N				0.158 – 0.474	**28.6**	**27.6**	0.158 – 0.474	**26.4**	**27.5**
	Loss coefficient of N_2O-N				0.0 – 0.1	4.7	2.9			
	Loss coefficient of NO-N				0.0 – 0.1	5.4	2.8			
	Loss coefficient of N_2-N				0.0 – 0.1	4.7	2.8			
Solid fraction storage	Fraction of inorganic N to Solid fraction				0.0 – 0.1	1.8	0.2	0.00 – 0.05	0.3	0.1
	Fraction of organic N to Solid fraction				0.05 – 0.30	1.7	0.4	0.05 – 0.25	0.2	0.2
	Mineralization rate of organic N				0.05 – 0.50	2.1	1.3	0.05 – 0.50	1.1	0.6
	Loss coefficient of NH_3-N				0.3 – 0.5	1.3	0.1	0.3 – 0.5	0.2	0.0
	Loss coefficient of N_2O-N				0.00 – 0.05	1.3	0.0	0.00 – 0.05	0.3	0.0
	Loss coefficient of N_2-N				0.0 – 0.3	1.4	0.1	0.0 – 0.3	0.3	0.0
Application	Leachate: Loss coefficient of NH_3-N	0.0 – 0.2	-0.7	0.1						
	Slurry: Loss coefficient of NH_3-N	0.05 – 0.50	3.1	4.9						
	FYM: Loss coefficient of NH_3-N	0.0 – 0.9	6.3	8.6						
	Solid fraction: Loss coefficient of NH_3-N				0.0 – 0.9	1.8	0.5	0.0 – 0.9	0.4	0.2
	Liquid fraction: Loss coefficient of NH_3-N				0.01 – 0.5	**35.0**	**34.3**	0.1 – 0.5	**40.1**	**40.8**

Notes: The parameters with FSI and TSI values more than 10% are indicated in bold.

Table A6.10. Sensitivity index (%), including first-order sensitivity index (FSI) and total sensitivity index (TSI), of input parameters on total P and K losses from MMCs Scenarios 1, 4 and 6.

Manure management facility	Parameters	Scenario 1			Scenario 4			Scenario 6		
		Range	FSI	TSI	Range	FSI	TSI	Range	FSI	TSI
		Total P loss								
FYM storage	Fraction of P to FYM storage	0.8 – 1.0	2.4	1.6						
	Leaching and runoff loss coefficient of P	0.0 – 0.1	98.5	97.7						
Solid fraction storage	Fraction of P to FYM storage				0.1 – 0.5	28.9	39.2	0.1 – 0.4	21.3	32.1
	Leaching and runoff loss coefficient of P				0.0 – 0.1	61.0	71.3	0.0 – 0.1	67.5	78.4
		Total K loss								
FYM storage	Fraction of K to FYM storage	0.5 – 1	11.8	12.5						
	Leaching and runoff loss coefficient of K	0 – 0.1	87.6	88.3						
Solid fraction storage	Fraction of K to FYM storage				0.0 – 0.2	42.3	55.2	0.0 – 0.1	42.8	58.3
	Leaching and runoff loss coefficient of K				0.0 – 0.1	44.6	57.5	0.0 – 0.1	42.1	57.6

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Table A6.11. Sensitivity index (%), including first-order sensitivity index (FSI) and total sensitivity index (TSI), of input parameters on GHG emissions from MMCs of Scenarios 1, 4 and 6.

Manure management facility	Parameters	Scenario 1			Scenario 4			Scenario 6		
		Range	FSI	TSI	Range	FSI	TSI	Range	FSI	TSI
Deep litter	Fraction of substrate stored under oxic conditions	0.5 – 1.0	4.8	6.4						
	Degradation rate of C under oxic conditions	0.3 – 0.6	**39.7**	**41.6**						
	Degradation rate of C under anoxic conditions	0.2 – 0.4	1.9	2.6						
	Fraction of C lost in CH_4-C	0.15 – 0.25	**35.4**	**36.3**						
	Loss coefficient of N_2O-N	0.0 – 0.1	5.3	5.6						
FYM storage	Fraction of substrate stored under oxic conditions	0.5 – 1.0	-0.4	0.0						
	Degradation rate of C under oxic conditions	0.3 – 0.6	-0.3	0.2						
	Degradation rate of C under anoxic conditions	0.2 – 0.4	-0.2	0.0						
	Fraction of C lost in CH_4-C	0.01– 0.03	0.2	0.5						
	Loss coefficient of N_2O-N	0.0 – 0.1	7.4	6.9						
Slurry storage	Fraction of substrate stored under oxic conditions	0.0 – 0.1	-0.4	0.0						
	Degradation rate of C under oxic conditions	0.0 – 0.1	-0.4	0.0						
	Degradation rate of C under anoxic conditions	0.05 – 0.15	-0.1	0.3						
	Fraction of C lost in CH_4-C	0.1 – 0.6	0.3	0.6						
Anaerobic digester	Degradation rate of C under anoxic conditions							0.3 – 0.5	5.8	6.7
	Fraction of C lost in CH_4-C							0.45 – 0.75	5.8	7.3
	Fraction of produced CH_4 Combusted							0.0 – 1.0	**85.1**	**88.6**
Digested slurry storage	Fraction of substrate stored under oxic conditions							0.0 – 0.1	-0.3	0.0
	Degradation rate of C under oxic conditions							0.0 – 0.1	1.2	0.0
	Degradation rate of C under anoxic conditions							0.0 – 0.1	-0.3	0.0
Liquid fraction storage	Fraction of substrate stored under oxic conditions				0.0 – 0.1	0.7	0.1	0.0 – 0.1	-0.2	0.0
	Degradation rate of C under oxic conditions				0.0 – 0.1	0.7	0.0	0.0 – 0.3	-0.2	0.0
	Degradation rate of C under anoxic conditions				0.05 – 0.20	**14.6**	**17.5**	0.0 – 0.1	1.4	0.2
	Fraction of C lost in CH_4-C				0.1 – 0.6	**22.5**	**24.9**	0.1 – 0.6	1.8	0.1
	Loss coefficient of N_2O-N				0.0 – 0.1	**52.3**	**53.0**			

Manure management facility	Parameters	Scenario 1			Scenario 4			Scenario 6		
		Range	FSI	TSI	Range	FSI	TSI	Range	FSI	TSI
Solid fraction storage	Fraction of C Solid fraction				0.3 – 0.6	1.9	3.0	0.3 – 0.6	-0.3	0.0
	Fraction of substrate stored under oxic conditions				0.5 – 1.0	0.7	0.1	0.5 – 1.0	-0.2	0.1
	Degradation rate of C under oxic conditions				0.05 – 0.40	1.4	1.0	0.0 – 0.6	-0.2	0.7
	Degradation rate of C under anoxic conditions				0.00 – 0.15	0.7	0.0	0.0 – 0.2	1.2	0.0
	Fraction of C lost in CH_4-C				0.0 – 0.1	2.5	1.7	0.0 – 0.1	1.7	0.1
	Loss coefficient of N_2O-N				0.00 – 0.05	0.3	0.1	0.00 – 0.05	-3.0	0.0

Notes: The parameters with FSI and TSI values more than 10% are indicated in bold.

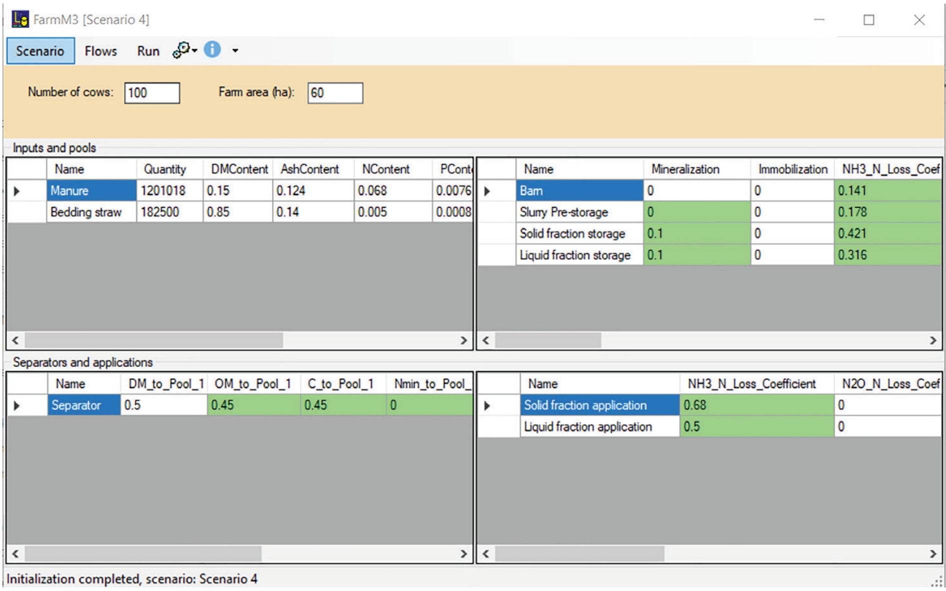

Figure A6.1. Main window of FarmM3 with four panels that allow parameterisation of the four types of components: Inputs, Pools, Separators and Applications. The parameters marked with a green colour have been selected for the sensitivity analysis.

$$
\begin{bmatrix} y_1 & y_2 & y_3 \\ y_4 & y_5 & y_6 \\ y_7 & y_8 & y_9 \\ y_{10} & y_{11} & y_{12} \\ y_{13} & y_{14} & y_{15} \end{bmatrix}
= \begin{bmatrix} f(x_{11}, x_{21}, x_{31}) & f(x_{11}, x_{22}, x_{31}) & f(x_{11}, x_{22}, x_{32}) \\ f(x_{12}, x_{22}, x_{32}) & f(x_{12}, x_{23}, x_{32}) & f(x_{12}, x_{23}, x_{33}) \\ f(x_{13}, x_{23}, x_{33}) & f(x_{13}, x_{24}, x_{33}) & f(x_{13}, x_{24}, x_{34}) \\ f(x_{14}, x_{24}, x_{34}) & f(x_{14}, x_{25}, x_{34}) & f(x_{14}, x_{25}, x_{35}) \\ f(x_{15}, x_{25}, x_{35}) & f(x_{15}, x_{26}, x_{35}) & f(x_{15}, x_{26}, x_{36}) \end{bmatrix}.
$$

Figure A6.2. Matrix with model results for a selected indicator Y for a Winding Stairs sensitivity analysis of a model with 3 parameters and using 4 sampling cycles (Chan et al., 2000).

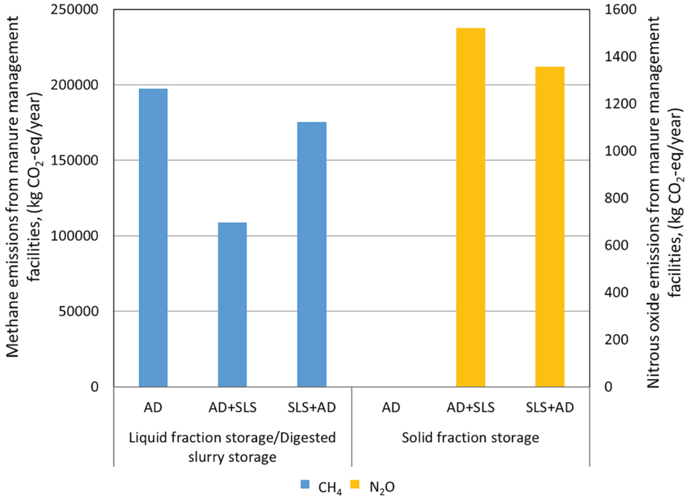

Figure A6.3. GHG emissions, including CH_4 and N_2O, from manure management facilities in the MMC applying AD alone (Scenario 5), the MMC applying AD before SLS (Scenario 6) and the MMC applying SLS before AD (Scenario 7).

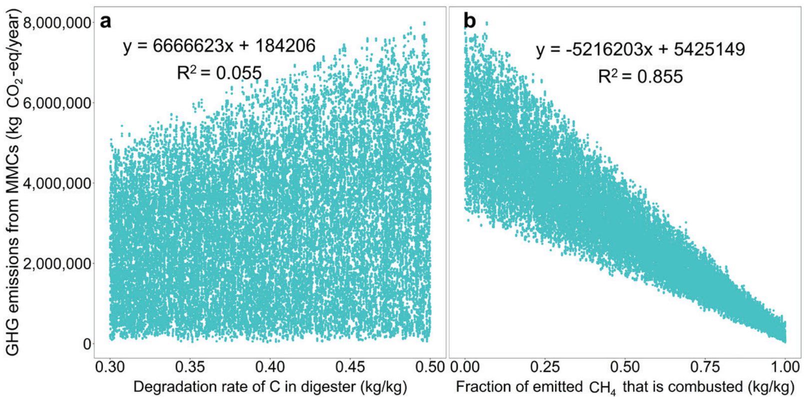

Figure A6.4. Influence of degradation rate of C in digester (a, with FSI and TSI values of 5.8% and 6.7%, respectively) and fraction of emitted CH_4 that is combusted (b, with FSI and TSI values of 85.1% and 88.6%, respectively) on GHG emissions from the MMC with AD and SLS of Scenario 6.

Chapter 7

Exploration of scenarios to increase nutrient circularity in an intensive mixed crop-dairy farm

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Abstract

Both improved manure management and integration of crop-dairy production have been proposed as alternative ways to reduce nutrient losses and improve sustainability of intensive dairy production. However, the potential interactive relationships between these two options are rarely investigated. In this study, a whole farm model (FarmDESIGN) extended with a manure management module (FarmM3) was used to simulate an intensive mixed crop-dairy farm with a herd of 66 cows and 9.6 ha of crop areas. The objectives were to investigate how different (combinations of) manure management technologies influence nutrient losses at manure management and farm levels and how manure management impacts farm multi-objective optimization results for more integrated crop-dairy production. The optimization aimed to improve farm environmental performance, increase feed self-sufficiency and food production. Our results showed that individual manure management technologies were insufficient to reduce N losses from manure management chains due to compensatory losses, whereas combinations of slurry solid-liquid separation, covered storage of solid and liquid fractions, and improved manure application could remarkably reduce N losses at manure management. Multi-objective optimization showed that improved manure management did not dissolve trade-offs or synergies among objectives but did affect the positions and the slopes of the solution frontiers between objectives. Differences between solution frontiers of alternative farm configurations in terms of N volatilization, soil N losses and soil organic matter (OM) balance indicated that MMCs could be designed effectively to optimize these objectives. The clear trade-off between feed self-sufficiency and dietary energy production indicated the food-feed competition of integrated crop-dairy systems. Our study confirmed the value of improved manure management in reducing N losses and improving the soil OM balance, and highlighted the potential to increase nutrient cycling without compromising food production by integrating crop and dairy production.

Keywords: FarmDESIGN, feed self-sufficiency, multi-objective optimization, manure management, integration of dairy-crop production

7.1 Introduction

7.1.1 Intensive dairy production

Dairy farming has intensified and specialized over the past few decades (FAO, 2018). With high stocking rates and large external feed inputs, intensive dairy farming systems are typically characterized by high productivity and profitability but also negative site-specific environmental impacts (Clay et al., 2020). Environmental problems, such as greenhouse gas and ammonia emissions, nutrient surpluses and eutrophication of ecosystems, have gained worldwide attention (Rotz et al., 2006; Oenema et al., 2007). Poor on-farm manure management and the spatial decoupling of crop and dairy farms were perceived as the major causes of nutrient losses from intensive confinement dairy farms (Bai et al., 2013; Chadwick et al., 2020). With a large amount of excreted manure but limited available cropland to utilize the produced manure, manure has become a burden for intensive confinement dairy farms. Additionally, due to the high livestock density per unit of area, the strong reliance on external feeds of these intensive confinement dairy farms has resulted in very low levels of nutrient circularity and use efficiency. Strategies to reduce nutrient losses and to improve nutrient use efficiency have been pointed out to involve improving manure management and by recoupling crop and dairy production (Oenema and Tamminga, 2005; Chadwick et al., 2020).

7.1.2 Improved manure management

Emission mitigation measures and manure treatment technologies have been developed to reduce nutrient losses from manure management facilities. For instance, covering of manure storages, slurry acidification and dilution, and injection of liquid manure were used to reduce ammonia emissions. Anaerobic digestion, solid-liquid separation and composting could contribute to greenhouse gases mitigation and facilitate manure nutrient management by producing alternative manure products such as anaerobic digestate, separated liquid and solid fractions, and compost (Foged et al., 2011; Sommer et al., 2013). However, devising a single mitigation measure on a single loss pathway has led to pollution swapping by increased losses of other compounds (De Vries et al., 2015). It was reported that the reduced ammonia emissions by covering slurry storage and by injection of liquid manure resulted in an increased nitrous oxide emission (Sommer and Hutchings, 2001; Berg et al., 2006). Moreover, the reduced gas emission at previous stages might lead to increased losses at later manure management stages (Shah et al., 2013). Considering the pollution swapping among nutrient loss pathways and compensatory losses among manure management technologies, a combination of different manure management technologies has been proven more efficient to reduce nutrient losses from the whole manure management chains (Rotz et al., 2006; Hou et al., 2014; Sajeev et al., 2018). However, the quantitative effects of improved manure management on nutrient use efficiency at farm level are rarely investigated. With more conserved nutrients in manure products, fewer nutrient inputs from synthetic fertilizers and more cropland might be needed to utilize these nutrients. Thus, for intensive confinement dairy farms, the recoupling of crop and dairy production could be a promising solution to reduce nutrient surplus and to increase nutrient circularity and use efficiency (Peyraud et al., 2014; Schut et al., 2021).

7.1.3 Reintegration of crop and dairy production

In integrated or recoupled crop and dairy systems, the manure produced by cows is used as a source of fertilizer for the crops, which in turn provide feeds for the livestock. This closed-loop system allows for the efficient use of nutrients, reducing the need for synthetic fertilizers and limiting the demand for imported feeds (Ryschawy et al.,2012; Marton et al., 2016). Although the value of integrated crop-livestock systems in terms of reducing detrimental environmental impacts has been confirmed, studies highlighted the necessity to consider food production of these systems (Lemaire et al., 2014; Puech et al., 2023), since the issue of increasing nutrient recycling by crop-livestock integration raises questions about the use of agricultural land and resource allocation between food crops, feed and animal products, particularly the role of intermediate resources such as fodder for animal feed (Barbieri et al., 2022).

Thus, a strategic plan and design for integrated crop-livestock systems is highly vital to improve nutrient use efficiency and circularity, to increase feed self-sufficiency without compromising food production. Given the strong interactions among different farm components (i.e., animals, manure, soils and crops) of integrated crop and dairy farms, whole farm models can be powerful means to redesign crop and dairy systems to balance supply and demand of feedstuff and manure, and to provide ex-ante assessments of performance of integrated crop-dairy systems.

7.1.4 Whole farm models

In this study, we will apply a whole farm model, the FarmDESIGN model, developed by Groot et al. (2012) to redesign farming systems by balancing crop-livestock interactions. This model can identify complicated interactions among farm components, support the exploration of alternative farm configurations using a Pareto-based multi-objective optimization algorithm, and provide redesign plans for improving farm nutrient use efficiency, increasing self-sufficiency of feed and guarantee food production. With a linked external manure management module (Chapter 6), it also allows to investigate the effects of improved manure management practices on nutrient losses and farm nutrient use efficiency.

7.1.5 Objectives

Overall, this study aims to investigate alternative farm management practices to improve nutrient use efficiency of intensive confinement dairy farms. Using the extended FarmDESIGN model, we first evaluated the impacts of various improved manure management practices on nutrient losses and farm nutrient use efficiency. Then we explored alternative farm configurations to further increase nutrient circularity by integrating crop and dairy production based on Pareto-based multi-objective optimization. Lastly, the potential impacts of improved manure management chains on designing and optimizing plans for integrated crop and dairy farms were investigated.

7.2 Materials and methods

7.2.1 FarmDESIGN model

FarmDESIGN is a bio-economic whole farm model that supports evaluation of mixed crop-livestock farm performance comprehensively with various agronomic, environmental and economic indicators (Groot et al., 2012). It can be used to simulate flows of organic matter, carbon, nitrogen, phosphorus and potassium to, through and from farm components (crop-animal-manure-soil) on an annual basis. Based on a Pareto-based multi-objective optimization algorithm, this model also enables to redesign towards sustainable farming systems by exploring alternative farm configurations of balancing different farm components interactions.

We used an extended version of the model with a flexible modular manure management model (FarmM3) (Chapter 6). FarmM3 quantifies processes (flows, conversions, losses) in manure management chains based on the quantities and qualities of materials provided such as the excreted manure, bedding materials or crop residues for co-digestion. The detailed calculation procedure of FarmM3 can be found in Chapter 6. The amounts and characteristics of manure available for application are fed back from the FarmM3 module to FarmDESIGN. Adding FarmM3 as an external manure module improved the flexibility of FarmDESIGN model in estimating nutrient losses from manure management chains with different manure treatment technologies.

7.2.2 Case study farm

A mixed dairy-crop farm in an agri-environmental scientific observation experimental station in Dali, Yunnan of China was selected as a case study. The dairy herd consisted of 6 calves, 18 heifers and 42 milking cows. Heifers were kept on the grazed pasture for 245 days during spring, summer and part of autumn. Milking cows and calves were kept in an open barn throughout the whole year. Dung and urine excreted on the pastures were kept on the pasture without collection. Excreta produced in the barn were collected separately with solid manure being sold out and liquid manure being stored in an underground tank for a period before being applied to fields. The total cultivated farm area was 9.5 ha, with 3.5 ha for fava-bean and rice rotation, 1.5 ha for annual ryegrass, 1.5 ha for barley and maize rotation, and 3 ha for alfalfa and rape rotation. The harvested rice and rapeseed were sold and other crop products were used as animal feed or bedding. Besides, large amounts of bedding materials and feeds were imported into the farm. Table 7.1 presents the baseline farm performance with a high livestock density, high farm N balance, a low soil OM balance and a low N cycling rate.

Table 7.1. Overview of the baseline performance of the case study farm.

Indicators		Value
Livestock density (LU/ha)		6.5
Feed self-reliance (%)		26.3
Nitrogen losses (kg/ha/year)	N volatilization from manure management	210
	Soil N losses	61
	N balance	272
Nitrogen cycling rate (%)		37
Soil organic matter balance (kg/ha)		-974

Note: The nitrogen cycling rate is defined as the fraction of excreted manure recycled into soil.

Since only a small area could be used for grazed pasture and there was a high potentiality of nutrient losses via leaching and runoff on the farm area, we first revised the baseline farm configuration with no grazing by heifers and keeping all of the animals in the open barn throughout the whole year. Second, we improved the herd structure by replacing dairy cows with heifers and calves at a replacement rate of 25%. Correspondingly, we revised the amount of feed intake to meet animal requirements (Table 7.2). To increase nutrient use efficiency and circularity within the farm, we assumed that all produced manure is applied within the farm. The modified farm was taken as the original farm and as the starting point for farm optimization.

Table 7.2. Overview of decision variables on different farm scenarios of the case study farm.

Description		Baseline	Modified (Original)	Scenario A		Scenario B	
				Minimum	Maximum	Minimum	Maximum
Number of animals kept on the farm	Dairy cows	42	42	12	42	—	—
	Replacement Rate	—	0.25	0.15	0.35	—	—
	Heifers	18	10	—	—	—	—
	Calves	6	10	—	—	—	—
Areas of crop in rotation, ha	Bean-Rice	3.5	3.5	3	9.5	3	45
	Ryegrass	1.5	1.5	0.5	1.5	0.5	1.5
	Barley-Maize	1.5	1.5	0	9.5	0	3
	Alfalfa-Rape	3	3	0	9.5	0	10
	Alfalfa-Rice	0	0	0	9.5	0	45
	Bean-Maize	0	0	0	9.5	0	45
Amount of external feeds, kg DM per year	Alfalfa silage	134558	134558	0	134558	0	134558
	Bean Straw	10000	10000	0	10000	0	10000
	Concentrate	80942	80942	0	80942	0	80942
	Concentrate2	20000	10000	0	10000	0	10000
	Maize straw silage	0	0	0	15000	0	15000
Amount of bedding supplied to animals, kg per day							
Calves		2.5	2.5	2	4	2	4
Heifers		2.5	2.5	2	5	2	5
Dairy Cows		4.5	4.5	2.5	6	2.5	6
Amount of bedding supplied to animals, kg DM per year							
External	Rice straw	64000	60000	0	90000	0	90000
Values of crop products used as feed or bedding*							
Bean-Rice rotation	Rice straw (as feed)	0	0	0	1000	0	1000
Alfalfa-Rice rotation	Rice straw (as bedding)	0	0	—	—	0	1000
Bean-Maize rotation	Bean straw (as bedding)	0	0	—	—	0	1000
Milk proudction and use	Milk Production, kg per cow per day	24	24	24	30	24	30
	Amount fed to animals, kg per year	3000	5000	1000	7500	—	—

*Crop straw could be either used as feed for animals or bedding. Assuming that produced crop straw was only used on the farm, the proportions of crop straw to different destinations will be calculated based on the values input to these destinations. For example, if the input values of rice straw used as feed and bedding are 1000 and 1, the proportion of rice straw used as feed will be 1000/(1000+1).

7.2.3 Manure management scenarios

Four manure management scenarios with different manure treatment technologies were developed and modelled in FarmM3 to investigate the impacts of changes in treatment technologies and mitigation measures of manure management on nutrient losses and nutrient use efficiency (Table 7.3).

Table 7.3. Description of modelled manure management chains.

Manure management chains	Description
S1	All of the manure excreted in the barn was collected and stored in a tank **without cover**. After being stored for two months, the slurry was applied to fields without incorporation.
S2	All of the manure excreted in the barn was collected and stored in an underground tank **with a concrete cover**. After being stored for two months, the slurry was applied to fields without incorporation.
S3	All of the manure excreted in the barn was collected and stored in the underground tank before being **separated into solid and liquid fractions**. The separated solid fraction was stored and **covered during storage**. The liquid fraction was stored in an underground tank with a concrete cover. After being stored for two months, the liquid and solid fractions were applied to fields without incorporation.
S4	All of the manure excreted in the barn were collected and stored in the underground tank before being **separated into solid and liquid fractions**. The separated solid fraction was stored and **covered during storage**. The liquid fraction was stored in an underground tank with a concrete cover. After storage, the solid manure was applied to fields by broadcast application with incorporation and the liquid was applied to fields by trailing hose.

7.2.4 Farm scenarios and objectives

To address the challenges of high nutrient losses and low farm nutrient use efficiency, two scenarios were developed. The first scenario (Scenario A) aimed to optimize the number of livestock units based on the amount of feed produced within the farm. The objectives of this scenario were to maximize whole farm N use efficiency (%) and soil organic matter (OM) balance (kg/ha), and to minimize N volatilization (kg/ha) and soil N losses (kg/ha). In Scenario B, the focus was on increasing nutrient circularity, improving feed self-sufficiency for the baseline herd, and expanding food production by integrating more crop area. This scenario aimed to maximize whole farm N use efficiency (%), self-supply of total feed DM (%), dietary energy production (persons fed/ha), and soil OM balance (kg/ha), while minimizing N volatilization (kg/ha) and soil N losses (kg/ha).

For each farm scenario, we explored alternative farm configurations to meet the target objectives and investigated the impacts of MMCs with different manure treatment technologies on alternative farm configurations.

7.2.5 Decision variables and constraints

For each scenario, the decision variables included management variables of the animal herd, allocation of crop areas, the destination of crop products, and the amount of external feeds and bedding materials supplied for animals (Table 7.2). Constraints were set for the total crop areas that should not be more than 9.5 ha in Scenario A and should be less than 45 ha in Scenario B. The extended farm areas in Scenario B were determined based on the availability of surrounding crop areas. The number of livestock units should be less than the current livestock units (56.2 LU) in Scenario A and be kept the same with the original farm in Scenario B. The feed availability of energy, protein, dry matter intake capacity and saturation should match animal requirements. The soil N losses should not be less than 20 kg N/ha/year, to make sure that enough N is in the system to support crop and grassland production while acknowledging unavoidable losses, and the soil P and K losses should not be less than 0 kg/ha/year to avoid mining. The supplied bedding material should be sufficient given requirements per animal (i.e., the bedding balance) with an allowed deviation of less than 5% for animal welfare (Table 7.4).

Table 7.4. Overview of constraints on different farm scenarios of the case study farm.

Description	Original	Scenario A		Scenario B	
		Minimum	Maximum	Minimum	Maximum
Deviation in feed balance intake (%)	-6.9	-∞	0	-∞	0
Deviation in feed balance energy (%)	4.6	-5	5	-5	5
Deviation in feed balance protein (%)	5.3	0	30	0	30
Deviation in feed balance structure (%)	159.1	0	∞	0	∞
Rotation Area (ha)	9.5	9	9.5	9	45
Livestock Units (LU)	56.2	15	57	—	—
Deviation bedding balance (%)	1.3	-5	5	-5	5
Soil nitrogen (N) losses (kg/ha)	187 – 340 [a]	20	187 – 340 [a]	20	187 – 340 [a]
Phosphorus (P) balance (kg/ha)	39	0	∞	0	∞
Potassium (K) balance (kg/ha)	658	0	∞	0	∞

[a] The soil N losses varied with manure management chains.

7.2.6 Model exploration

For model exploration, we ran the Pareto-based multi-objective optimization for 3000 iterations to get 500 alternative farm configurations for each scenario. The complete mathematical explanation of the algorithm with the corresponding formulae is described by Groot et al. (2012). Here we briefly summarize the optimization process. The DE algorithm generates two populations of solutions which represent the decision variables. The opportunity space created by these populations is diverse; the variety in the decision variables (genotypes) creates diversity in farm performance that is measured by the indicators (phenotypes). The first population of 'parents' serves as the result-set that is iteratively improved, while the second population consists of 'competitors' that are generated by uniform cross-over of three selected 'parent' solutions in each iteration.

The solutions in both populations are ranked using the principle of Pareto-optimality (Groot et al., 2012) and the Euclidean distance between the solutions in the opportunity space is calculated from the normalized indicator values, which serves to quantify a crowding metric. After ranking a selection process is conducted by pairwise comparison: solutions in the 'parent' population are replaced by individuals from the 'competitor' population if the latter has a better Pareto rank or if the ranks are equal it is positioned in a less crowded part of the opportunity space. The rank-based selection results in movement of the 'parent' population in the direction of the trade-off frontier (or surface), while the crowd-based selection ensures spread along the frontier (or surface).

We repeated each optimization for 3 times to get stable outcomes. The parameter settings of uniform cross-over in the Differential Evolution algorithm of the optimization was 0.85 for mutation probability and 0.15 for amplitude of mutations (Groot et al., 2007).

7.3 Results

7.3.1 Impacts of manure management on N losses

Improved manure management could reduce N losses from MMCs and improve N use efficiency in manure management which is defined as the fraction of excreted manure applied to fields. Although there were no obvious differences in N losses and N use efficiencies between S1 and S2, MMC configurations S3 and S4 could reduce N losses by 46 to 58% and increase manure N use efficiencies by more than 30%, compared to S1 (Fig. 7.1). This implied that applying a single emission mitigation measure (i.e., slurry cover) had limited influence on N losses from MMCs and manure N use efficiencies, while combinations of slurry solid-liquid separation, covered solid and liquid fractions storage and improved manure application could remarkably reduce N losses and improve N use efficiencies of manure management.

Differences among the MMC scenarios S1-S4 combined with the baseline farm configuration would not result in reduction of total N losses from the whole farm or improvement of farm N use efficiency. Due to the high livestock density and heavy reliance on imported feeds, the N excreted by animals exceeded the N requirements of crop production, resulting in a high N surplus. Without changing the number of cows and crop areas, the conserved N from improved manure management would be lost after being applied to soil. Therefore, improvements in manure management did not affect farm N use efficiency which is calculated as the ratio of N output (i.e., crop and animal products) from the farm to total N inputs (i.e., imported feeds, fixation and deposition) to the farm.

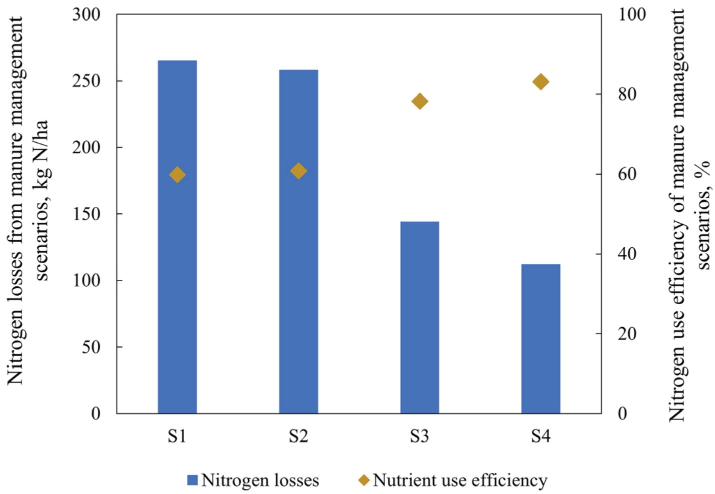

Figure 7.1. Nitrogen losses and N use efficiency under different manure management scenarios. The blue bars represent N losses, and the diamond dots indicate N use efficiency of various manure management scenarios.

7.3.2 Effects of livestock density on N losses and soil OM balance

Exploration results showed that a greater whole farm N use efficiency could be achieved by reducing the number of cows on the farm, i.e., by reducing livestock density (Scenario A). With a decline of livestock density from 5.9 LU/ha to 3.3 LU/ha, the whole farm N use efficiency could be increased by 44% to 67%. The reduced livestock density led to lower manure load per unit of area, further resulting in less N volatilization, soil N losses and lower soil OM balance. This indicated distinct synergies (1) between improving whole farm N use efficiency and reducing N volatilization (Fig. 7.2A), (2) between improving whole farm

N use efficiency and decreasing soil N losses (Fig. 7.2B), (3) between reductions of soil N losses and N volatilization (Fig. 7.2C). On the other hand, trade-offs were found (4) between improvements of whole farm N use efficiency and soil OM balance (Fig. 7.2D), (5) between reducing N volatilization and increasing the soil OM balance (Fig. 7.2E), and (6) between reducing soil N losses and improving the soil OM balance (Fig. 7.2F).

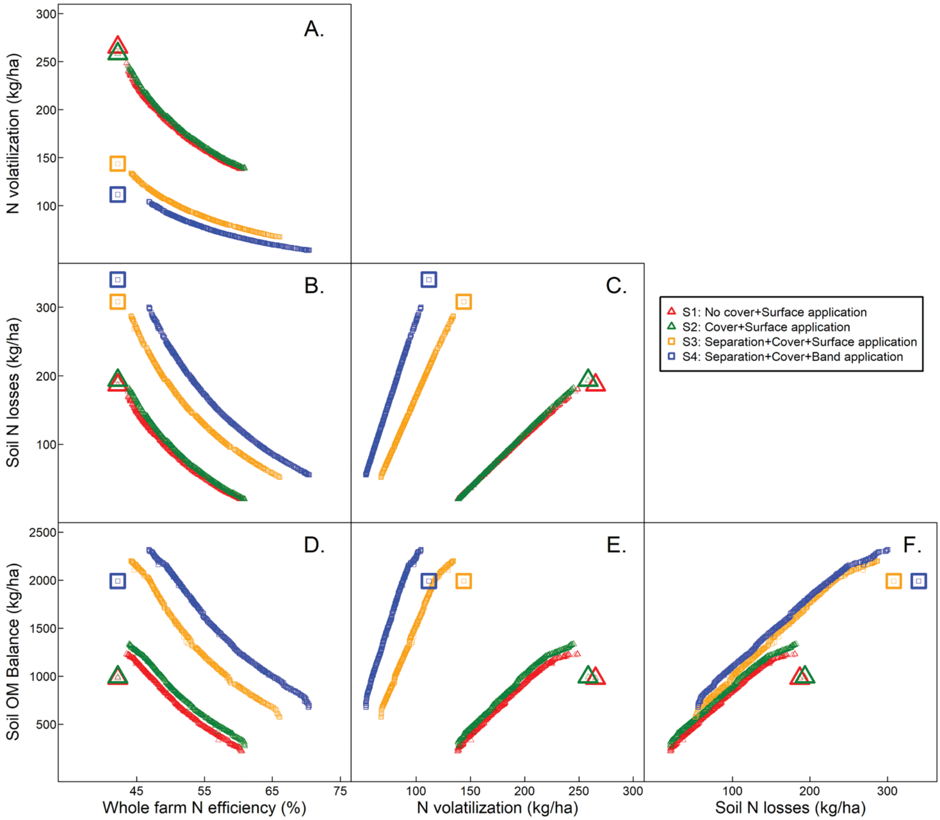

Figure 7.2. Relationships between the objectives whole farm N use efficiency, N volatilization, soil N loss and soil OM balance in the farm scenario A with different manure management chains (i.e., S1, S2, S3 and S4). The larger symbols mark the performance of the original farm configuration.

Although the trade-off or synergy relationships among objectives were present under different MMCs, considerable differences in positions and slopes of solution frontiers in terms of N volatilization, soil N losses and soil OM balance under different MMCs were observed. These differences could primarily be explained by the impacts of manure management technologies on N volatilization, soil N losses and soil OM balance. As shown in Fig. 7.2, the apparent distance between the solution frontiers of MMCs S1, S2 and S3, S4 in terms of N volatilization and whole farm N use efficiency (Fig. 7.2A) showed lower N volatilization in MMCs S3 and S4 due to multiple N mitigation measures (i.e., solid-liquid separation, cover and improved application methods). Similar results were also found in solution frontiers of N volatilization and soil N losses (Fig. 7.2C), and N volatilization and soil OM balance (Fig. 7.2E). But higher soil N losses and soil OM balance were observed in MMCs S3 and S4 than in MMCs S1 and S2 with the same whole farm N use efficiency (Figs. 7.2B and 7.2D). Higher soil N losses in MMCs S3 and S4 demonstrated the compensatory N losses from manure management and soil, with less N volatilization from MMCs S3 and S4 (Fig. 7.2A) leading to higher N losses from soil (Fig. 7.2B). Higher soil OM balance in MMCs S3 and S4 could be as a result of the more contribution of solid manure to soil OM than the contribution of slurry manure in MMCs S1 and S2.

The higher slope of the synergy frontier between N volatilization and whole farm N use efficiency in MMCs S1 and S2 than in MMCs S3 and S4 indicated a stronger reduction in N volatilization in MMCs S1 and S2 needed to reach the same increase in whole farm N use efficiency (Fig. 7.2A). On the contrary, the higher slope of the synergy frontiers between N volatilization and soil N losses in MMCs S3 and S4 showed that with the same decrease in N volatilization, more reduction in soil N losses could be obtained in MMCs S3 and S4 than S1 and S2 (Fig. 7.2C). Similar slopes but larger range of the frontiers were observed between whole farm N use efficiency and soil N losses, between whole farm N use efficiency and soil OM balance, and between soil N losses and soil OM balance in MMCs S3 and S4 than S1 and S2 (Figs. 7.2B and 7.2C). Compared to the baseline farm N use efficiency, a larger improvement in whole farm N use efficiency could be achieved in MMCs S3 and S4 than in MMCs S1 and S2.

7.3.3 Exploration of alternative farm configurations of integrated crop-dairy production

Since there was no clear difference in optimization results of objectives between MMCs S1 and S2, and between MMCs S3 and S4, we only performed farm explorations under two contrasting MMCs (S1 and S4) in farm Scenario B. Compared to exploration results of farm Scenario A, similar but less linear trade-offs and synergies among objectives of whole farm N use efficiency, N volatilization, soil N losses and soil OM balance were observed in farm Scenario B due to more complicated interactions with added objectives of feed self-sufficiency and dietary energy supply. Exploration results of integration of crop and dairy production with multi-objective optimization indicated that changing farm configurations could also substantially improve whole farm N use efficiency, reduce N volatilization and soil N losses (Fig. 7.3). We did not observe clear trade-offs or synergies between whole farm N use efficiency and self-supply rate of feed DM (Fig. 7.3A), and between whole farm N use efficiency and dietary energy supply (Fig. 7.3K). But a clear trade-off between increasing self-supply rate of feed DM and improving dietary energy supply was observed in Fig. 7.3L, indicating food-feed competition in integrated crop-dairy systems. Fig. 7.4 shows how the shift in cropping patterns and changes in the amount of external feed inputs define the relationship between these two objectives. In farm Scenario B, to approach the objective of increasing feed self-sufficiency, the model shifted the feed production from external importation to on-farm production, subsequently causing the increase of areas to feed crops, such as alfalfa (Figs. 7.4A and 7.4B). Conversely, in the alternative configurations with greater dietary energy supply, the model allocated larger areas to food crops (i.e., Chinese cabbage and maize) and smaller areas to alfalfa. The reduced on-farm alfalfa silage production was compensated by an increased external alfalfa silage input (Figs. 7.4C and 7.4D).

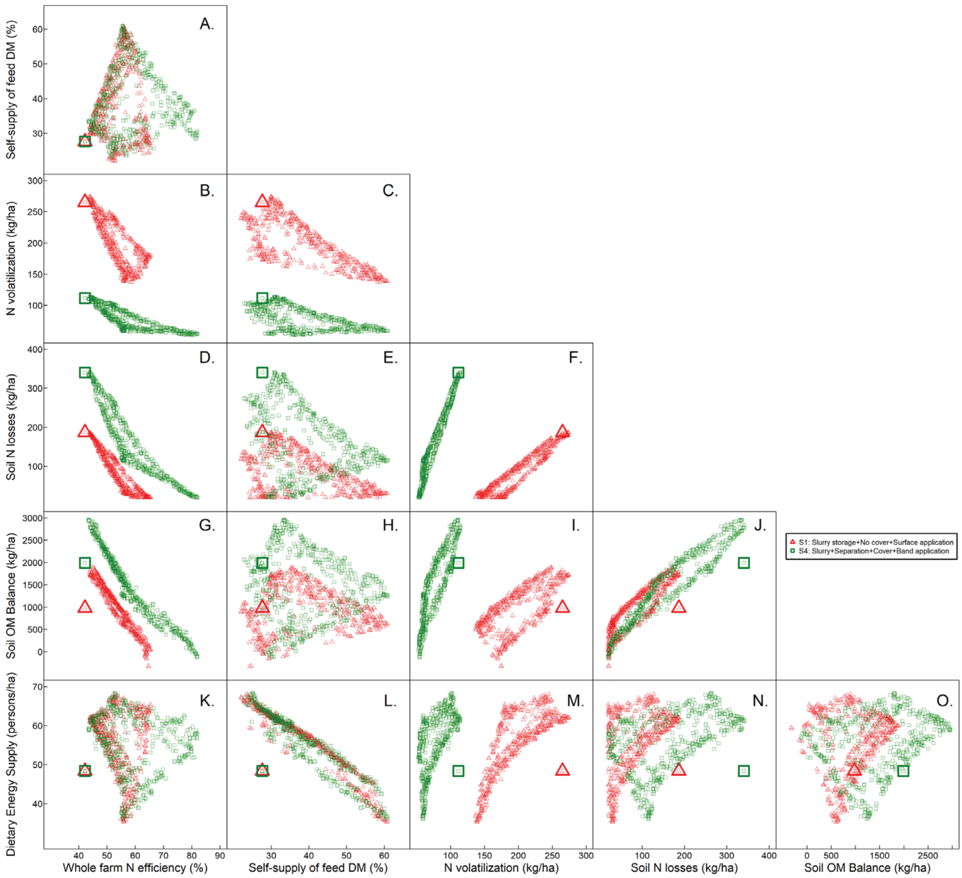

Figure 7.3. Relationships between the objectives whole farm N use efficiency, self-supply of feed DM, N volatilization, soil N loss, soil OM balance and dietary energy supply in the farm scenario B with two different manure management chains. The larger symbols mark the performance of the original farm configuration.

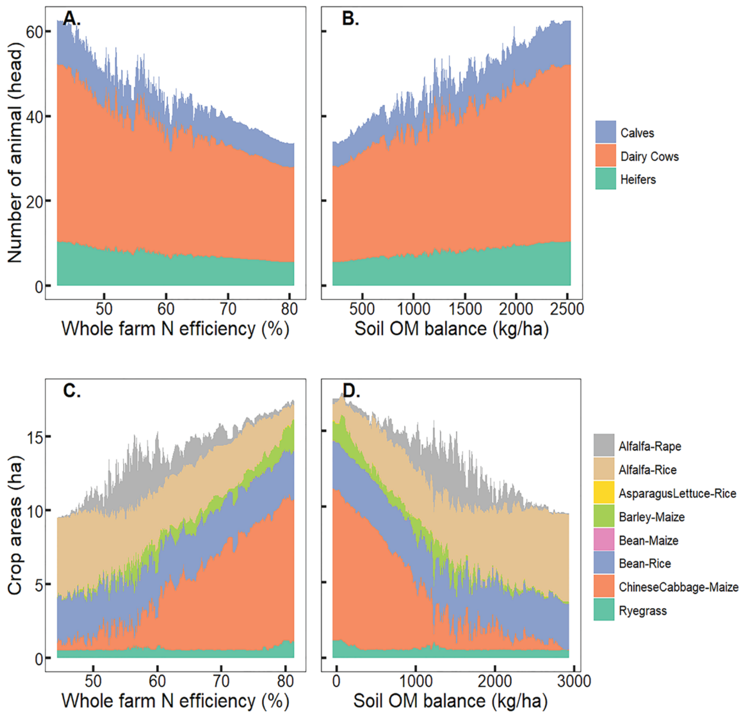

Figure 7.4. Modeled allocation of crop areas and amount of imported feeds in each alternative farm configuration generated to meet the objectives of maximizing self-supply rate of feed DM and dietary energy yield in farm Scenario B. The x values were ordered from minimum to maximum for each objective.

7.4 Discussion

7.4.1 Effects of improved manure management on N losses

Comparisons of N losses under various manure management scenarios demonstrated that an individual emission mitigation measure was insufficient to reduce N losses at manure management level. The reduced N losses at earlier stages of manure management chains could result in increased losses at later stages, i.e., compensatory N losses between manure management stages (Shah et al., 2013). To avoid compensatory losses among manure management technologies, an integrated approach with combined manure management technologies was more efficient to reduce nutrient losses from the whole manure management chains (De Vries et al., 2015a; 2015b). Our study confirmed that, compared to manure management scenario with slurry storage and surface application

(S1), a combination of solid-liquid separation, covered solid or liquid manure storage and improved manure application could substantially reduce N losses from the whole manure management chains.

It is important to note that the lack of observed impacts of improved manure management practices on total farm N losses and N use efficiency in this study may not necessarily apply to all farms or situations. The high livestock density and imported feed N on the case study farm may have contributed to the limited response to improved manure management practices. In other farm systems with lower livestock density and less imported feed N, improved manure management practices may have a greater impact on reducing N losses and increasing N use efficiency (Shah et al., 2013). For instance, Rotz et al. (2006) showed that the total farm N use efficiency could be increased by 5% to 7% by implementing nutrient conservation technologies, including a barn floor for feces and urine separation, covered six-month manure storage, manure injection, etc. Tan et al. (2022) also found that the farm N use efficiency could be increased from 53 to 65% by applying tightly covered manure storage and solid-liquid separation.

In addition, the limited response of whole farm N use efficiency to improved manure management practices could also be attributed to the static response of crop yields to nutrient supply in the FarmDESIGN model. In this model, increased productivity is only obtained when variants with higher target crop yields as reflected in the input-output relations are included in the cropping mix. As an extension to the model, using response curves with a yield plateau at saturating N availability (Lassaletta et al., 2014) or a dynamic crop simulation approach (Holzworth et al., 2014) could enable to quantify crop yield response, resulting in reduced nutrient losses and higher N use efficiency through improved manure management.

7.4.2 Effects of integrating crop and dairy production on farm performance

Integrated crop-dairy systems offered further possibilities for minimizing environmental impacts. Larger reductions in N volatilization and soil N losses, as well as greater whole farm N use efficiencies were achieved by reducing livestock density and through integrating crop and dairy production. This study showed that reducing livestock density from 5.9 to 3.3 LU/ha resulted in a more than 40% increase in whole farm N use efficiency. The negative relationship between whole farm N use efficiency and livestock density was also reported in the study of Powell et al. (2010) who highlighted that optimal livestock density can reduce total N losses and enhance farm N use efficiency.

Reduced livestock density also resulted in a decline in soil OM balance due to a decrease in manure load per unit of area. Improved manure management could enhance soil OM balance by reducing OM degradation and conserving more OM in manure. For instance, when solids and liquids from animal manure are separated, more than twice the amount of OM can be retained compared to not separating them. The buildup of soil OM depends on the amount and type of manure applied (Rayne and Aula, 2020). For example, applying dairy manure compost at rates of 35, 70, and 105 Mg/ha could increase soil OM by 33%, 88%, and 88%, respectively, compared to not applying dairy manure compost (Butler et al., 2008). Solid manure has been found to contribute more to the accumulation of soil OM than liquid manure (Rodrigues et al., 2021). According to Peyraud et al. (2014), SOC storage increased by 20% to 60% from the addition of solid manure and by 10% to 30% from the addition of liquid manure. These findings emphasize the importance of considering livestock density and manure management in optimizing soil OM accumulation.

Despite the positive impacts of reducing N losses and improving N use efficiency by integrating crop and dairy production, we identified the obvious trade-off between feed self-sufficiency and dietary energy supply. The issue of crop-dairy integration has raised the question about resource allocation for food and feed production (Muscat et al., 2020; Puech and Stark, 2023). The food-feed competition in our study was mainly generated from the use of cropland, with more cropland used to produce livestock feeds (e.g., alfalfa and whole maize silage) leading to a smaller area for food production. A survey on intensive dairy farms in Henan province of China presented that food–feed competition was prevalent in many intensive dairy farms, where dairy cows consumed more human-edible protein than they produced in milk and meat (Wang et al., 2022). Other ways to alleviate the food-feed competition on integrated crop-dairy farm might include increasing nutrient use efficiency in cropping systems through optimizing crop rotations and increasing crop yields per area (Barbieri et al.,2021). Increasing animal feed use efficiency at the animal and herd level is also an important lever to save feed resources (Barbieri et al.,2022). Additionally, collaboration between local crop and dairy farms for direct exchange of manure and crop by-products could further close nutrient loops at larger scales, promoting resource utilization efficiency and contributing to a circular food system (Martin et al., 2016; De Boer and Van Ittersum, 2018).

7.4.3 Limitations

Some limitations of this study were identified based on current results. First, we explored alternative farm configurations with a focus on optimizing environmental and nutritional indicators and without considering economic indicators. Although some studies have proven that crop-livestock integration could limit the negative environmental impacts without compromising farm economics (Dumont et al. 2013; Guillou et al. 2013; Martin et al. 2016), in practice, the cost for manure management varies with types of treatment technologies. A more comprehensive evaluation that considers environmental, economic and nutritional indicators would contribute to a better understanding of the role of manure management in farm management, and would help farmers to adopt cost-effective manure management technologies. Second, our study might underestimate the contribution of manure management on farm nutrient management as the model calculated nutrient flows based on mass balance without considering the nutrient availability of different manure types as fertilizer which highly depends on methods of manure handling, storage and treatment (Rufino et al., 2006; Johansen and Jensen,2009; Risbery et al., 2017). In addition, the impacts of manure management technologies on soil biodiversity are worthwhile to investigate, which can further facilitate sustainable agriculture development.

7.4.4 Future research

The improved FarmDESIGN model offers a useful tool to explore how improved manure management influence nutrient flows and use efficiency at the whole farm, and to help farmers to reintegrate crop and dairy production and to reallocate farm resources to achieve their objectives. As integration of crop and dairy production within farms requires greater workload and increased skills and knowledge in animal, manure, soil and crop management, future research on integrating crop and dairy production beyond the farm scale by exchanging manure and feedstuff between dairy and crop farms are necessary, especially for intensive and specialized farms. Nutrient “sharing” between crop and dairy farms within a region can provide complementary interactions and benefits, reducing externalities of specialized farms and contributing to close nutrient cycles at a larger scale.

7.5 Conclusions

Manure management plays an important role in farm nutrient management of intensive mixed crop-dairy farms. Due to substantial nutrient losses from a large amount of produced manure, as well as complicated interactions of manure management and soil and crops, the effects of improved manure management on farm nutrient management should also be considered when seeking to optimize farm configurations. Our study integrated an external manure management model (FarmM3) to a whole farm model (FarmDESIGN), which enables (i) to evaluate the effects of diverse improved manure management technologies on nutrient losses from manure management and from the whole farm system; (ii) to identify potential influence of improved manure management on multi-objective optimization of farm configurations.

To reduce nutrient losses from the whole manure management chain, a single manure management technology was insufficient, highlighting the importance of integrated approaches to reduce N losses from manure management. At a high livestock density, total N losses from the whole farm were not influenced by improved manure management, since conserved N from manure management could be lost after being applied to cropland. A greater reduction of N losses from both manure management and soil could be achieved by reducing livestock density, resulting in an improved whole N use efficiency. Although trade-offs and synergies existed among objectives, improved manure management did not change relationships among objectives but did affect the positions and the slopes of the solution frontiers between objectives of N volatilization, soil N losses and soil OM balance. To move towards sustainable intensification of dairy production, increasing nutrient circularity by improving manure management with multiple mitigation measures and integrating crop and dairy production within farm or between farms are necessary. Given the trade-off of food-feed competition in integrated crop-dairy systems, food production should also be considered when optimizing farm configurations towards more sustainable agricultural production.

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Chapter 8

General Discussion

8.1 Overview of the main findings

The main objectives of this thesis were to increase the understanding of losses of different manure constituents along complex manure management chains, and to explore alternative options to increase nutrient circularity of intensive confinement dairy farming systems. The following section provides a brief overview of the main results. Some of these results give new insights, whereas the others may be seen as confirmation of the earlier findings.
Chapters 2, 3 and 4 systematically reviewed the pathways and magnitude of losses of nutrients from manure in different manure management facilities, including dairy barns, slurry storage and solid manure storage. Large variation was found in reported nutrient losses across publications, especially for ammonia (NH_3) and methane (CH_4) emissions. Temperature had important effects on NH_3 and CH_4 emissions, with higher temperatures leading to increased emissions. In addition, manure characteristics and management practices also affected the pathways and magnitude of nutrient losses. These findings highlight the complexity of manure management and suggest that a tailored approach is necessary for effective mitigation of nutrient losses.

Chapters 5 and 6 proposed a flexible modular approach and developed a modular manure management (FarmM3) model that can be used to quantify degradation and losses of different manure constituents (OM, C, N, P and K) from various manure management chains (MMCs). The impact of manure management facilities on the fate of manure constituents varied among MMCs. The MMCs with deep litter stables and anaerobic digesters yielded higher OM degradation, C losses and GHG emissions due to breakdown of the added straw. Application of mechanical solid-liquid separation could reduce GHG emissions, but its effect on NH_3 emissions varied depending on the characteristics of the separated slurry. The most important parameters determining degradation and losses of manure constituents varied among MMCs. In MMCs with deep litter, the immobilization and mineralization rates between inorganic and organic N were more influential than the loss coefficients of NH_3-N. In MMCs with solid-liquid separation and anaerobic digesters, the loss coefficients of NH_3-N from liquid manure storage and application were more influential than from solid fractions. The separation efficiencies of organic N and inorganic N did not influence total N losses from MMCs with solid-liquid separation. In contrast, the separation efficiencies of P and K from solid-liquid separation influenced total losses of P and K.

Chapter 7 zoomed out from manure management chains and explored alternative options to increase nutrient use efficiency and circularity at farm level. These alternative options included improved manure management and integration of crop-dairy production. The study found that individual manure management technologies were insufficient in reducing N losses from MMCs due to compensatory losses. However, combinations of slurry solid-liquid separation, covered solid and liquid fractions storage, and improved manure application could remarkably reduce N losses at manure management. To develop more environmentally friendly intensive dairy farming systems, manure management chains could be designed effectively to reduce N volatilization and soil N losses, and to increase soil OM balance. Integrated crop-dairy production could improve whole farm N use efficiency and increase nutrient circularity. But trade-offs between increased whole farm N use efficiency and soil OM balance, and between improved feed self-sufficiency and dietary energy supply were found. Both improved manure management with multiple mitigation measures and integrating crop and dairy production are promising options to increase nutrient use efficiency of intensive dairy farming systems.

8.2 Limitations

By integrating contrasting manure management facilities, the model described in Chapters 5 and 6 can cover various manure management chains in on-farm settings. With a focus on effects of manure management facilities on degradation and losses of manure constituents along the whole MMCs, we might neglect the accuracy of loss coefficients of manure constituents by using the empirical values from publications, while effects of environmental factors, especially temperature on loss coefficients were not considered. Future studies should avoid this limitation by integrating established empirical equations or developing new ones that relate loss coefficients to environmental factors, and then incorporating these equations into the FarmM3 model. For example, emission factors quantifying losses during manure field application could first be estimated using the ALFAM2 model, which takes into account the slurry dry matter (DM), application rate, temperature and rain at, or immediately after the time of application (Sommer et al., 2019).

Another limitation was the lack of on-farm measurements on nutrient losses from the whole MMCs to validate the estimation accuracy of FarmM3 model. Due to limited time and budget, I was not able to set up a measurement campaign on a candidate dairy farm to measure and collect data on nutrient losses from manure management during my PhD period. Although the comparisons between the estimates of our approaches and calculations using other modelling approaches verified the feasibility of our developed model, validation using on-farm measured data can further increase the reliability of our modelling estimates.

8.3 Nutrient losses from manure management

8.3.1 Nutrient loss coefficients

Within the continuum of manure management from housing facilities, storage and treatment, and field application, manure nutrients could be lost from the point of excretion by the animal until they are incorporated into soil. A considerable number of studies have been conducted to measure nutrient losses from different manure management facilities (Sneath et al., 2006; Leytem et al., 2012; Shah et al., 2012; Ngwabie et al., 2014; Balde´ et al., 2018). The specific units used in quantifying these loss coefficients depend on the pollutant being measured and the context of the study or analysis (Vigan et al., 2019). The large diversity in the units (Table 8.1) used to report nutrient losses from manure management complicates the comparison and summary of these data. Standardization of the emission data is essential for compiling and comparing emission values from different studies. For compatibility with emission inventories of livestock manure management systems based on mass flow analysis, it is necessary to express the loss coefficients as a percentage of mass.

Most of the reported NH_3 and CH_4 emission factors in dairy barns were in units of grams or kilograms of gas per cow per day or per year. It was difficult to convert these data to flow-based emission factors due to the lack of information on manure excretion. In this study, we converted the various reported units of gas emissions into a uniform unit of g/AU/d (1 AU equals 500 kg of live weight) to make emission rates comparable for cows at different growth stages (Chapter 2), with a range from 3.6 to 109.4 g/AU/d for NH_3 emissions and a range from 102.1 to 462.2 g/AU/d for CH_4 emissions.

Gas emission factors from slurry storage were expressed in various units, including units of mass of gas per mass of slurry, per square meter of slurry storage, or per cubic meter of slurry. Sommer et al. (2019), based on reported NH_3 emissions from slurry storage, developed NH_3 emission factors in percentage of total ammoniacal nitrogen (TAN) for compatibility with the EMEP/EEA air pollutant emission inventory guidebook (European Environment Agency (EEA), 2016). Kupper et al. (2020) gave a more detailed review on gas emissions from slurry storage, and presented emission values for untreated cattle slurry, with a range from 14 to 19% of TAN for NH_3, 0.08 to 0.18% of total nitrogen (TN) for N_2O, and 2.3 to 3.7% of volatile solids for CH_4. Chapter 3 of this thesis presented larger ranges of flow-based emission factors of NH_3 and N_2O in % of TN and emission factors of CH_4 in % of total carbon (TC) from dairy slurry storage, with a range from 0.09 to 47.7% of TN for NH_3 emissions, 0 to 0.39% of TN for N_2O emissions, 0.01 to 17.2% of TC for CH_4 emissions. In our review, not only gas emissions from raw slurry were included but also emissions from liquid slurry after separation, anaerobically digested slurry and anaerobically digested slurry after separation. The difference in slurry characteristics resulted in a larger variation in flow-based emission factors, which also implied the importance of manure treatment technologies in determining nutrient losses.

Table 8.1. Overview of units used to report emission values in literature.

Gas	Barn	Slurry storage	Solid manure storage
NH_3	g/h; g/d; kg/d; kg/month	µg/m^2/s; g/m^2/d; g NH_3-N/m^2/d	g/m^2; g NH_3-N/m^2
	mg/animal/h; g/animal/h; g/animal/d; g/animal/year; mg/animal/d; kg/animal/year; g NH_3-N/animal/d	g/min; g/d; kg/d	g/m^2/d; g NH_3-N/m^3/d; µg NH_3-N/m^2/s
	mg/LU/h; g/LU/h; g/LU/d; kg/LU/year; kg/LU; g NH_3-N/LU/h; g NH_3-N/LU/d	mg/kg/d	mg/kg/d
	mg/g N intake; g/kg DM intake; g/kg milk; mg NH_3-N/g N intake; g NH_3-N/kg N intake; mg NH_3-N/g urine-N; mg NH_3-N/g TAN	g/t; g/m^3	kg/animal/d
	g/HPU/d; mg/HPU/h	g NH_3-N	kg NH_3-N/t; g/t; g NH_3-N/kg, g/kg; g NH_3-N/kg DM; g NH_3/kg DM
	etc.	% NH_3-N of initial N	g NH_3-N; g NH_3
		etc.	% NH_3-N of initial N
			etc.
N_2O	µg/animal/s; g/animal/h; mg/animal/d; g/animal/d	g/m^2/d; µg N_2O-N/m^2/s	mg N_2O-N/m^3/d; µg N_2O-N/m^2/s
	µg/LU/s; mg/LU/h; mg/LU/d; g/LU/h; g/LU/d; kg/LU/year	kg/d; g/min; g/d	mg/kg/d, µg/kg/h
	µg/m^2/s; mg/m^2/h; mg/m^2/d	mg/kg/d	g N_2O-N/animal/week; g/animal/d
	mg/g N intake; mg/kg DM intake; g/kg milk	g/m^3; g/m^2; g N_2O-N/m^2	g N_2O-N/kg, g/kg; g N_2O-N/kg DM; g N_2O/kg DM
	g/HPU/d	g/animal/week; g/animal/d	g N_2O-N; g N_2O
	etc.	% N_2O-N of initial N	% N_2O-N of initial N
		etc.	etc.
CH_4	g/h; g/d; kg/d	g/m^3/d; kg/m^2/year; L/m^3/d; µg/m^2/s	g/m^3/d; g CH_4-C/m^3/d; µg/m^2/s
	mg/animal/s; g/animal/d; mg/animal/h; g/animal/h; kg/animal/d	g/t; g/m^3; kg/m^2	g/animal/d; g/animal/week; kg/animal/d; kg/animal/year
	g/LU/d; g/LU/h; mg/LU/s	g/animal/week; g/animal/d; kg/animal/d	mg/kg/d
	mg/m^2/s; g/m^2/d; mg/m^2/h	mg/kg/d	g/kg DM; g CH_4-C/kg; g/kg
	g/kg DM intake; g/kg FPC milk	kg/d	g CH_4-C; g CH_4
	g/HPU/d	% CH_4-C of initial VS; % CH_4-C of initial C; g/kg VS; L/kg VS	% CH_4-C of initial C
	etc.	etc.	etc.

Gas	Barn	Slurry storage	Solid manure storage
CO_2	g/h; g/d; kg/d	kg/d	mg/kg/h
	mg/animal/s; g/animal/d; mg/animal/h; g/animal/h; kg/animal/d	% CO_2-C of initial VS; mg/L	g/kg DM; g CO_2-C/kg; g/kg
	g/LU/d; g/LU/h; mg/LU/s; kg/LU/year	etc.	g CO_2-C; g CO_2
	mg/m^2/s; g/m^2/d; mg/m^2/h		etc.
	g/kg DM intake; g/kg OM intake; g/kg NDF intake; g/kg FPC milk		
	etc.		

For solid manure storage or composting, the emission factors of gas emissions were mostly reported in units of grams or kilograms of gas per unit of mass of manure or compost or per unit of time. In Chapter 4, we developed emission factors for NH_3, N_2O, CH_4 and CO_2 in units of grams or kilograms of gas per unit of dry matter or TN based on reported data in various units from different publications. These flow-based loss coefficients could benefit emission inventories of dairy manure management systems with composting and contribute to assessments of nutrient flows and losses from complete manure management chains.

8.3.2 Factors influencing nutrient losses from manure management facilities

A number of literature reviews have been conducted to identify the influence of interactions between animal diet, manure characteristics, manure management and environmental factors on nutrient losses from dairy barns (Bougouin et al., 2016; Edouard et al., 2019; Sanchis et al., 2019; Poteko et al., 2019; Chapter 2), liquid manure storage (Kupper et al., 2020, Chapter 3) and solid manure storage (Chapter 4). Although the objectives of different studies varied, environmental factors, especially temperature, were proven as the main influencing factors of NH_3 emissions from dairy barns, implying the importance of defining emission factors of NH_3 for particular climate zones (Sommer et al., 2019). The effect of dietary crude protein on NH_3 emissions was highlighted by the studies of Bougouin et al. (2016) and Edouard et al. (2019). Lowering diet crude protein leads to less N excretion in urine, which is the primary factor influencing NH_3 emissions. In this thesis, we only found 13 studies reporting feed information, with 11 studies reporting crude protein or N intake and four studies reporting crude fiber or carbon intake. The insufficient data prevented us from investigating the influence of feed type on gas emissions. A recent study however showed that a low-protein feed had limited effects on improving the whole farm N use efficiency (Tan et al., 2022).

Floor type (solid or slatted) and manure handling methods (scraped or flushed) did not affect NH_3 and CH_4 emissions from dairy barns (Chapter 2), which agreed with the meta-studies of Bougouin et al. (2016) and Poteko et al. (2019). Differences in floor properties of design (sloped or levelled) and surface (smooth or rough) played a larger role in modulating NH_3 emission rates than floor type (solid or slatted) (Braam et al., 1997; Zhang et al., 2005; Pereira et al., 2011). Compared to other reviews on gas emissions from dairy barns, Chapter 2 highlighted that the measurement technique has a crucial impact on the estimated emission values, especially for emission of CH_4. Standardization of measurement methods and reported results of continuous measurement are crucial and needed to reduce the large variability and uncertainty of estimating gaseous emissions from dairy buildings (Janke et al., 2020).

Liquid and solid manure storage are other major sources of nutrient losses. In addition to temperature which significantly affected the magnitude of losses, especially for NH_3 and CH_4, manure characteristics and management practices also played important roles (Chapter 3 and 4). Application of emerging manure management practices could induce changes in physical, chemical and/or biological properties of manure and hence influence the fate of manure constituents (Hou, 2014; Khalil et al., 2016; Aguirre-Villegas et al., 2019). Lowering slurry pH by acidification could simultaneously reduce NH_3, CH_4 and N_2O emissions from storage (Sajeev et al., 2018). Solid-liquid separation could reduce total solids content of separated liquid manure storage, further resulting in decreased NH_3 and N_2O emissions due to the absence of natural crusts (Aguirre-Villegas et al., 2014; Holly et al., 2017). A gas emission trade-off between reduced CH_4 emissions and increased NH_3 emissions during digested slurry storage could be explained by reduced volatile solids and increased TAN content of digested slurry (Holly et al., 2017; Aguirre-Villegas et al., 2019).

8.3.3 Nutrient losses from the whole manure management chains

The importance of integrated modelling approaches in estimating gaseous emissions and nutrient flows from a whole chain perspective has been pointed out, given the possible interactive effects of manure management facilities on emissions (Hou et al., 2014; Sajeev et al., 2018; Wei et al., 2021). Some modeling approaches to simulate or quantify flows and losses of manure constituents in livestock manure management systems have been developed, using either relatively simple emission factors, or empirical equations, or very detailed process-level simulations. These approaches are varied in their complexity and accuracy. For example, Webb and Misselbrook (2004) and Dämmgen and Hutchings (2008) developed mass-flow methods to estimate gaseous N emissions along MMCs using emission factors. The mechanistic model Manure-DNDC developed by Li et al. (2012) provided the most detailed representation of the biogeochemical processes producing emissions from manure handling. All these modeling approaches have appropriate applications. Emission-factor based models provide useful tools for decision support in the strategic design of manure management systems. Weaknesses of this type of model are that they may not appropriately represent actual degradation and loss processes over the full range of possible conditions, and they have less accuracy compared to process simulation models that consider biochemical and biophysical processes that govern the transport and transformation of nutrients in the manure life cycle (Li et al., 2012; Rotz et al., 2012). However, mechanistic models are not as flexible as emission-factor based models. The addition of new processes requires detailed process models, which often require much time and effort to develop or adapt to the existing model structure (Rotz et al., 2017).

In this thesis, we proposed a modular approach (Chapter 5) based on loss coefficients and developed a modular manure management (FarmM3) model (Chapter 6) to extend the flexibility of model application in complex manure management systems. This model can be used as a helpful tool for quantifying degradation and losses of different manure constituents (OM, C, N, P and K) in contrasting MMCs. Based on flow-based emission factors, this parsimonious simulation method avoids estimation of too many detailed parameters and has enough accuracy, thus ''*getting the right answers for the right reasons*'' (Keating 2020). The results can contribute to understanding the effects of various manure management facilities on the flows and losses of manure OM, C, N, P and K through the whole MMC. Being linked with the FarmDESIGN model (Groot et al., 2012) (Chapter 7), this integrated model can be used as a decision-assistance tool to support farm managers in designing and optimizing manure management strategies and provide case-by-case recommendations that include the complexity of dairy farm systems.

8.4 Contribution of improved manure management to farm sustainability

Manure management as an important component in intensive dairy farming system, plays a vital role in farm nutrient management. Improved manure management can contribute to farm sustainability by addressing environmental challenges associated with manure production and management (Malomo et al., 2018).

8.4.1 Reduced nutrient losses

Integrated manure management approaches have been proven more efficient in reducing N losses from manure management chains than single mitigation measures due to the potential pollution swapping. A combination of solid-liquid separation, covered storage and banding application of manure could substantially reduce N losses by 46 to 58% compared to manure management with storage and broad application (Chapter 6). Rotz et al. (2006) demonstrated that implementation of combination of a low-emission barn floor, an enclosed covered manure storage and the deep injection of manure could reduce N losses by 24 to 29%. By conserving more manure nutrients in manure products, improved manure management could also increase farm nutrient use efficiency. Tan et al. (2022) found that the farm N use efficiency increased from 53 to 65% by applying tightly covered manure storage and solid-liquid separation.

8.4.2 Reduced GHG emissions

Application of manure management technologies, such as solid-liquid separation, anaerobic digestion, and composting, could reduce GHG emissions and contribute to global efforts to mitigate climate change. Mitigation of GHG emissions can be achieved by reducing CH_4 and N_2O emissions from slurry storage, which can be accomplished by decreasing slurry dry matter and easily degradable organic matter content (Amon et al., 2006). Compared to untreated slurry, GHG emissions could be reduced by 13 to 25% for anaerobic digestion, 31 to 38% for solid-liquid separation, and 40 to 41% for combined anaerobic digestion and solid-liquid separation (Holly et al., 2017; Aguirre-Villegas et al., 2019). Solid-liquid separation had a greater potential for GHG mitigation than anaerobic digestion, but the variability depended on the performance of the digester in which the level of degradation and carbon capture would greatly impact the downstream mitigation potential (Holly et al., 2017). Composting is an effective method for treating manure by transforming degradable organic matter into stable humus for use as organic fertilizer (Chadwick et al., 2011). Although unfavorable by-products, including CH_4 and N_2O emissions, might be generated during the composting process (Yin et al., 2021), well-managed composting could mitigate overall GHG emissions compared to raw manure storage. A combined solid-liquid separation and rotating composter could reduce overall GHG emissions by 36 to 74% (Fillingham et al., 2017). Our study also showed that, compared to other composting methods (i.e., static, turning, windrow), silo composting had the best performance in reducing overall GHG emissions, since the aerobic conditions in the composter prohibited the production of CH_4 and N_2O.

8

8.4.3 Improved soil quality

Manure application can improve soil physical, chemical and biological properties. A recent meta-study presented that the application of manure on agricultural soils increased soil organic carbon stocks by 35.4% on average (Gross et al., 2021), varying with soil properties, tillage intensity, climate conditions and manure types and amounts (Maillard et al., 2014). Compared to storage of raw manure, improved manure management with solid-liquid separation, cover and banding application can reduce C losses by 31% from manure management, and conserve more C in manure products, contributing to increased SOC stocks (Chapter 7). Applying manure is also promising to increase soil buffer and cation exchange capacities, to alleviate soil acidity (Cai et al., 2015), and to enhance microbial activity, the abundance and biomass of soil fauna (Watts et al., 2010). Due to the presence of heavy metals, antibiotics, pathogens in animal manure, Köninge et al. (2021) highlighted the importance of manure quality and its management on soil biodiversity. Studies showed that aerobic composting could reduce the amounts of antibiotic resistant genes in manure (Tien et al., 2017), and increase the abundance of earthworms in soil (Rollett et al., 2020), thus benefiting soil biodiversity.

8.4.4 Other benefits

Manure management technologies can produce more alternative manure products, creating room for better management of nutrient balances among regions (Peyraud et al., 2014). Solid-liquid separation of slurry can make it easier to apply liquids nutrients in a targeted manner, and to store and transport the separated solid fractions off the farm, avoiding over-application and reducing the risk of nitrogen losses to the atmosphere. Composting can also reduce the volume and density of manure, facilitating transportation of the final product over longer distances (Jørgensen and Jensen, 2009).

8.5 (Re)Integration of crop and dairy production

Mixed-farming systems that reconnect livestock and crop production are proposed as a viable strategy to move towards to sustainable and resilient agricultural production (Lemaire et al., 2014; Moraine et al., 2014). For intensive dairy farming systems with heavy reliance on off-farm feeds and with substantial manure nutrient surpluses, closing the loop in nutrient and energy cycles by recoupling dairy and crop systems at farm and territorial scales can help reduce the environmental externalities of intensive farms and increase their resilience (Garrett et al., 2020).

8.5.1 (Re)Integration levels

The integration of crop and dairy production can occur at various hierarchical levels, ranging from an individual farm to a larger agricultural region or even the national level. The success of integration efforts within or between farms depends on the motivation, resources, and knowledge of farmers involved. The enabling environment, such as infrastructure, policy support, and markets, could also facilitate the integration. In Chapter 7, a case study was conducted on a dairy farm with high livestock intensity (6.5 LU/ha) and a nutrient surplus, where more than 70% of the feed was purchased off-farm. To increase nutrient cycling and feed self-sufficiency on the farm, alternative configurations were explored and evaluated based on available resources, such as the number of cows and the areas of croplands. It was found that reducing livestock intensity from 6.5 to 3.0 LU/ha by either reducing the number of cows or increasing the area of croplands could achieve synergy between nutrient cycling and feed self-sufficiency. However, this also resulted in a decrease in food energy production (Chapter 7).

One of the key attributes of integrated crop-dairy systems at the farm level is the increased nutrient cycling and the reduced financial risks (Russelle et al., 2007; Ryschawy et al., 2017). Compared to specialized farming systems, integrated mixed crop-livestock systems were less sensitive to market price fluctuations in inputs and sales prices (Ryschawy et al., 2012). In addition, on farm autonomy allowed farmers to retain more control of the whole integrated production system (Entz et al., 2005). Potential limitations of within farm integration are that farmers need to absorb a greater workload through the year and require additional skills and knowledge of animal, manure, soil and crop management, which makes it difficult to recouple crop and dairy farms at the farm level (Moraine et al. 2014; Martin et al., 2016; Ryschawy et al., 2013).

As an alternative, some researchers have proposed to recouple dairy and crop production between farms by direct exchange of manure and straw among specialized farms (Martin et al., 2016). This form of integration allows some of the synergies normally provided by within-farm integration to be obtained, but with much smaller increases in farm workload, skills and infrastructure on the individual farms involved. But exchanges between crop and livestock farmers have transaction costs and require coordination and management of trade-offs between individual and collective objectives and performances. For instance, manure moves from livestock farms to crop farms, but nutrients do not necessarily return to livestock farms through feedstuff (Moraine et al. 2014; Peyraud et al. 2014). Therefore, strategic planning among crop and livestock farms is vital to match crop rotations and related plant products to animal feed requirements (Peyraud et al. 2014). Economically efficient transport of exchanged material also determines farmers' economic acceptance. Dagnall et al. (2000) reported that manure with high dry matter content (~70 %) can be profitably transported up to 40 km from animal houses, whereas manure with low dry matter content (<10 %) can only be transported 10 km. Improved manure management with solid-liquid separation or composting in livestock farms could facilitate this cross-farm cooperation. The integration of crop and livestock production on a larger level, both nationally and globally, often requires coordination through national or global markets rather than direct communication between farmers. This integration can be influenced by a range of socio-economic and political factors (Martin et al., 2013). Optimizing international trade based on coupling crop and livestock production could reduce 30% of manure N losses and transportation cost (Zhang et al., 2022).

8.5.2 Research methodology

A variety of quantitative and qualitative design approaches have been developed to analyze and design alternative farming systems, including optimization approaches, and participatory and simulation-based approaches (Martin et al., 2013). Optimization approaches aim to develop problem-solving algorithms with the emphasis on the computational exploration of the solution space using computer models (Martin et al., 2013; Ryschawy et al., 2014). The design process consists of three main activities: conceptualizing and analyzing the current farm performance, optimizing and generating alternative farm configurations, and evaluating generated alternatives (Martin et al., 2013). The whole farm model FarmDESIGN developed by Groot et al. (2012) follows the Describe – Explain – Explore – Design cycle (DEED) (Giller et al., 2008), and supports the design of mixed farming systems in an iterative learning cycle. Chapter 7 used this model to design strategic plans for reintegrating crop and dairy production within the case study farm based on Pareto-based multi-objective optimization algorithm. Alternative farm configurations were generated to match supply and demand for manure and feedstuff to achieve synergy between environment benefits and agricultural production. Instead of generating a single optimal solution, the FarmDESIGN model aims to generate a set of alternative solutions that reveal synergies or trade-offs among the objectives (Groot et al., 2012). These alternative solutions can support decision-makers to have their autonomous choice from a broad portfolio of alternatives and can serve as entry points for future participatory processes with multiple stakeholders (Groot and Rossing, 2011).

In contrast to optimization approaches, participatory and simulation-based approaches involve a broader range of stakeholders, including researchers, farmers, advisors and consumers (Martin et al., 2013). These approaches have been used to design and assess scenarios for integrating crop-livestock at the territory level (Moraine et al., 2016; Moraine et al., 2017; Ryschawy et al., 2017). Through collective organization, the expectations of both crop farmers and livestock farmers can be specified in the design process (Ryschawy et al., 2017). The study by Moraine et al. (2017) showed that the potential complementarities between crop farmers' supply and livestock farmers' demand could enhance self-sufficiency at the collective level. However, collective organization to manage exchanges of crop products and livestock manure between farms revealed specific logistical and social barriers, such as the distance and scale for the exchanges (Asai et al., 2014), trust development among famers (Ryschawy et al., 2017), and trade-offs between individual and collective environmental and socio-economic objectives (Asai et al., 2018).

8.6 Outlook of future research

8.6.1 More measurements on pilot or farm scales

Although laboratory measurements of gaseous emissions from manure management facilities provide valuable information about the relative effects of different treatments on emissions (Perazzolo et al., 2015), they may not be representative of values found in real-world settings, and thus could not be included in calculating emission factors. To obtain more accurate estimations of nutrient losses from manure management facilities, measurements on pilot or farm scales are necessary and crucial. While about 46% and 31% of measurements on emissions from slurry storage were conducted at farm and pilot scales, respectively (Kupper et al., 2020), more on-farm measurements with detailed information, such as manure characteristics, management practices, and climate conditions, are needed to ensure efficient measurements and better comparisons between studies.

Measurements of NH_3, CH_4 and N_2O emissions from manure management facilities have received considerable attention, but there is still limited knowledge about the mechanisms of manure OM degradation, the conversion of manure organic and inorganic N, and the potential influencing factors. More research is needed to fill these knowledge gaps. Understanding these mechanisms is important for increasing the accuracy of estimates of emission factors, as well as for improving the efficiency and sustainability of manure management practices.

8.6.2 Enabling environment for manure management

Improved manure management has the potential to bring significant environmental benefits. However, the implementation of manure management technologies in practice remains low (Chadwick et al., 2020; Tan et al., 2022) despite the urgency of sustainable animal manure management. Training and communication of improved manure treatment technologies and management practices through Agricultural Extension Services can be helpful. Moreover, policy support, such as reallocated subsidies, can reduce the financial constraints associated with improved manure management practices, e.g., purchasing manure containment facilities, establishing contractor businesses for transporting and spreading manures, and the development of small-scale manure spreading equipment. Future research is needed to understand the barriers preventing farmers from adopting existing manure management technologies and to identify enabling conditions, such as policies, market opportunities, knowledge, and assistance that could further facilitate adoption.

8.6.3 Recoupling crop and dairy production beyond farm scale

The case study in Chapter 7 showed that on farm crop–dairy integration can improve nutrient cycling and limit the negative environmental impacts. However, for intensive dairy farms with high stocking densities, limited land availability can be a significant constraint. In such cases, integration at larger levels beyond the farm is necessary. The integration of specialized crop and dairy farms at regional level can contribute to creating a more sustainable and resilient food production system that maximizes resource efficiency and reduces waste. This approach also contributes to nested circularity at larger geographical scales, such as regional or national levels, ultimately leading to reduced stress on planetary boundaries. To design effective scenarios for crop-livestock integration at the local or regional level, future research should focus on case studies of intensive agricultural regions. And participatory approaches would be needed to conduct in-depth case studies by taking into account the local agricultural context, the stakeholders involved, and cultural differences that may exist.

8.7 Concluding remarks

In this thesis I explored alternatives to increase nutrient circularity of intensive dairy farming systems through improved manure management and re(integration) of crop and dairy production. The following conclusions can be drawn:

- A tailored approach is necessary for effective mitigation of nutrient losses and improving manure management.
- The developed FarmM3 model can be used to estimate nutrient flows and losses from contrasting manure management chains and identify the entry points to improve manure management chains.
- A combined manure management practices is more effective to reduce nutrient losses from manure management chains than individual practices.
- For highly intensive confinement dairy systems, improved manure management has limited effects on reducing nutrient losses at farm level.
- (Re)integrated crop-dairy production can be a valuable approach to increase nutrient cycling, but it is important to consider the potential trade-offs between increased feed self-sufficiency and food production.

The findings of this thesis may provide insights for both farmers and policy makers to reduce nutrient losses and improve nutrient circularity in intensive dairy farming systems.

- For farmers, it is essential to recognize the complexity of manure management and the need for tailored approaches to achieve effective nutrient management. The FarmM3 model developed in this study can help farmers estimate nutrient losses from manure management and identify areas for improvement.

- For policy makers, it is important to tailor policies that address the specific needs of regions with different levels of dairy farming intensity.
    - o In regions with less intensive dairy farming, policy makers should prioritize promoting integrated dairy and crop production systems, while avoiding policies that encourage intensification and decoupling between livestock and crops. In addition, it is important to provide incentives for the adoption of improved manure management practices and increasing access to manure management facilities. This could include offering subsidies for the adoption of improved manure storage and application technologies, as well as for the installation and use of solid-liquid separators, composting facilities, and anaerobic digesters. Collaboration between governments and private sector partners could also improve the availability and affordability of these facilities, particularly for small and medium-sized dairy farms.
    - o In regions with concentrated and intensive dairy farming, it is important to recognize that improving manure management alone may not be sufficient to reduce nutrient losses and increase nutrient circularity. Instead, a combination of improved manure management practices and (re)integrated crop-dairy production are necessary to achieve sustainable nutrient management. To support this integration, policy makers could provide financial incentives, such as tax breaks or subsidies, for specialized crop and dairy farms that work together to create integrated crop-dairy production systems. Additionally, supporting the development of infrastructure such as equipment for transporting exchanged materials and for spreading manure, promoting the development of local markets for dairy and crop products, such as farmers' markets or community-supported agriculture programs, can help to make integrated crop-dairy production systems more economically viable.

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Summary

As global demand for dairy products grows, the dairy industry has intensified and specialized towards larger, industrial farms. This contributes to increased production efficiency and economy of scale, but at a cost to planetary boundaries. Intensive dairy production has led to deforestation and associated biodiversity loss, impaired nutrient cycling, and increased greenhouse gas (GHG) emissions. To address these issues, improving manure management and integrating crop-dairy production have been proposed as crucial approaches to improve circularity and sustainability of intensive dairy farming systems.

In the **General introduction**, I present the challenges and opportunities of intensive dairy farming systems. With focus on the crucial role of manure management and its impact on the overall performance of intensive dairy farming systems, the main objective of this thesis was to increase the understanding of conversion and losses of different manure constituents along complex manure management chains, and to explore alternative options to increase nutrient use efficiency and circularity of intensive dairy farming systems with diverse manure management chains.

Chapter 2 presents a meta-analysis of measured NH_3 and CH_4 emissions from confinement dairy barns. The analysis aimed to explore the relationships between gas emission rates and housing system, environmental factors and measurement methods. A large variation in measured gas emission rates from dairy buildings was observed, indicating that a single emission factor for all dairy farms barns is not realistic. Ammonia emissions in commercial barns were more affected by environmental factors than by housing factors and measurement methods. Thus, NH_3 emission factors should be defined for particular climate zones. A positive relationship between temperature and CH_4 emission rates was proven within a practical temperature range. Both NH_3 and CH_4 emissions presented no significant difference between solid floor and slatted floor, or between flushed and scraped systems. Methane emissions showed more variability between different measurement methods than NH_3 emissions. The results highlight the need for standardization of measurement methods and reported results of continuous measurement to reduce the large variability and uncertainty of estimating gaseous emissions from dairy buildings.

Chapter 3 provides a comprehensive analysis of published data on cumulative NH_3, N_2O and CH_4 emissions from dairy slurry storage. The aim was to evaluate the integrated effects of slurry pH, total solids (TS), ambient temperature and length of storage on emissions using linear mixed-effects models. The results revealed a large range of flow-based gas emission factors of slurry storage, with variation among laboratory, pilot and farm scale studies. Slurry composition and storage conditions play a crucial role in defining carbon and nitrogen transformations, and thus the resulting losses of NH_3–N, N_2O–N and CH_4–C. The NH_3–N losses were highly related to slurry pH, lowering slurry pH leading to a significant decrease

in emissions. Temperature also affected NH_3–N losses, with higher emissions under warm conditions. No significant relationship was observed between NH_3–N losses and slurry TS contents within a range from 21 to 169 g/kg. The losses of N_2O–N from dairy slurry storage were not significantly affected by slurry pH, TS contents and temperature. Emissions of CH_4–C showed a significant positive relationship with temperature, a negative relationship with slurry TS contents, and no significant relationship with slurry pH ranging from 6.6–8.6. Length of storage (more than 30 days) had no significant influence on cumulative gas emissions from slurry storage. This study provides new emission factors of NH_3, N_2O and CH_4 as a percentage of TN or TC from dairy slurry storage, highlighting the potential interactive effects of slurry characteristics and storage conditions on gaseous emissions from slurry storage. More measurements on flow-based gas emission factors at farm scale are needed to better estimate carbon, nitrogen flows and cycles and improve nutrient use efficiency in dairy farming systems.

Chapter 4 presents a meta-analysis of NH_3 and GHG emissions from different dairy manure composting methods, including static, turning, windrow, and silo composting. The study showed a wide range of gaseous emissions from dairy manure composting, which were influenced by the physical characteristics of the composted material and the management practices of composting. Principal component analysis indicated that the initial TC and TN content of the composted material were crucial in mediating gaseous emissions. Low TC and TN content can simultaneously reduce emissions of CH_4, CO_2 and N_2O. Among different composting methods, turning composting resulted in the highest gas emissions, whereas silo composting had the best performance in reducing GHG emissions, although it significantly promoted N losses through NH_3 emissions. The study also highlighted effective mitigation measures for gas emissions during composting, such as applying compost biofilters, which have proven to be the most effective way to reduce NH_3 emissions. Adding sawdust or straw to the composting process can significantly reduce CH_4 and N_2O emissions.

Based on emission factors from the systematic reviews on manure treatment facilities, in **Chapter 5**, I proposed a modular approach to estimate flows of TAN and organic N and to quantify different N species losses (e.g., NH_3, N_2O, NO and N_2 emissions, N leaching and runoff) from MMCs with different complexity in dairy farms. The simulated N losses from various MMCs from nine published case studies ranged from 20% to 50% of excreted N. These estimates were within the range of estimates obtained from other Tier2/Tier 3 modelling methodologies, which confirmed the reliability of our estimates.

In **Chapter 6**, the approach was extended to include degradation of OM and C, losses of P and K by integrating loss coefficients from the literature. A modular manure management (FarmM3) model was developed. The model was used to simulate flows and losses of manure OM, C, N, P and K from MMCs with different manure management facilities, including deep litter, anaerobic lagoon, solid-liquid separation (SLS), anaerobic digestion (AD), and combinations of SLS and AD. The results showed that MMCs with deep litter and AD led to higher OM degradation, C losses, and greenhouse gas (GHG) emissions due to the substantial amounts of straw added to bedding and the digester. A trade-off between GHG and ammonia emissions was identified in the MMCs with deep litter. Application of SLS could reduce GHG emissions by 40% to 60% due to reduced CH_4 and N_2O emissions from separated liquid fraction storage. A stronger reduction of ammonia emission was observed when applying SLS to digested slurry than to raw slurry. The sensitivity analysis of degradation and losses of manure constituents showed that the N loss was most sensitive to N transformation in the MMC with deep litter, and was most affected by the loss coefficients of NH_3 during liquid manure storage and application in MMCs with SLS and AD. Losses of P and K from MMCs with SLS were influenced by separation efficiencies of SLS and loss coefficients of solid fraction storage. This study demonstrated that manure management facilities have a strong influence on the fate of manure constituents. The FarmM3 model can be used to quantify the degradation and losses of different manure constituents in complex MMCs and to identify the most important parameters determining these losses.

In **Chapter 7**, I used a whole farm model (FarmDESIGN) that was extended with FarmM3 to investigate how different (combinations of) manure management technologies influence nutrient losses at manure management and farm levels. In addition, I assessed how manure management affects farm multi-objective optimization results, with aims to improve farm environmental performance, increase feed self-sufficiency, and enhance food production. The results showed that individual manure management technologies were insufficient to reduce N losses from manure management chains due to compensatory losses. However, combining slurry solid-liquid separation, covered storage of solid and liquid fractions, and improved manure application can remarkably reduce N losses from manure management. Furthermore, multi-objective optimization results indicated that improved manure management did not dissolve trade-offs or synergies among objectives but did affect the positions and the slopes of the solution frontiers between objectives. MMCs could be designed effectively to optimize N volatilization, soil N losses, and soil OM balance. The clear trade-off between feed self-sufficiency and dietary energy production indicated the food-feed competition of integrated crop-dairy systems. Our study confirmed the value of improved manure management in reducing N losses and improving the soil OM balance, and highlighted the potential to increase nutrient cycling without compromising food production by integrating crop and dairy production.

In the **General discussion**, I summarize and synthesize the main findings from this thesis, identify the limitations of our study, put the findings into a broader context and reflects on the methodology. For manure management in intensive dairy farms, it is essential to recognize the complexity of manure management and the need for tailored approaches to achieve effective nutrient management. Policy support, such as incentives for the adoption of improved manure management practices and increased access to manure management facilities are necessary for intensive dairy farms. Enabling environments for (re)integration of crop and dairy production within farm and beyond farms are necessary to achieve more circular and sustainable dairy production.

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About the Author

Qingbo Qu was born on February 13th, 1992 in Shanxi province, located in Northern China. In 2010, she started her bachelor in Environmental Engineering at Hubei University in Wuhan, the capital city of Hubei province. After completing her four-year undergraduate program, she decided to further her study and enrolled in the MSc programme in Environmental Engineering at the Agro-Environmental Protection Institute (AEPI), Chinese Academy of Agricultural Science (CAAS). In September 2017, she joined "Wageningen University-CAAS joint PhD programme" and started her PhD journey, under the supervision of Dr. Jeroen Groot from WUR and Prof. Keqiang Zhang from CAAS. During her PhD, she worked on nutrient flows and losses from manure management chains in dairy farms.

List of publications

Qu, Q., Groot, J.C.J., Zhang, K., 2022. A modular approach for quantification of nitrogen flows and losses along dairy manure management chains of different complexity. Nutrient Cycling in Agroecosystems 122, 89-103.

Qu, Q., Groot, J.C.J., Zhang, K., Schulte, R.P.O., 2021. Effects of housing system, measurement methods and environmental factors on estimating ammonia and methane emission rates in dairy barns: A meta-analysis. Biosystems Engineering 205, 64-75.

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Ba, S., **Qu, Q.,** Zhang, K., Groot, J.C.J., 2020. Meta-analysis of greenhouse gas and ammonia emissions from dairy manure composting. Biosystems Engineering 193, 126-137.

Qu, Q., Zhang, K., Groot, J.C.J., 2023. A model to identify entry points to curb emissions from complex manure management chains. *Submitted to Journal of Cleaner Production, under review after major revision.*

PE&RC Training and Education Statement

With the training and education activities listed below the PhD candidate has complied with the requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

The C.T. De Wit Graduate School PE&RC
PRODUCTION ECOLOGY & RESOURCE CONSERVATION

Review/project proposal (6 ECTS)

- Exploration of scenarios to increase circularity and sustainability of intensive dairy farming systems in China

Post-graduate courses (5.3 ECTS)

- Grasping sustainability; PE&RC/SENSE (2018)
- The protein transition: divers perspectives; PE&RC (2023)
- Introduction to R for statistical analysis; PE&RC/SENSE (2018)
- Meta-analysis; PE&RC/SENSE (2018)
- Mixed linear models; Chinese Academy of Science (2019)

Deficiency, refresh, brush-up courses (2 ECTS)

- Analysis and design of organic farming systems; FSE (2018)

Competence strengthening/skills courses (5.55 ECTS)

- Effective academic development; WGS (2017)
- Information literacy including endnote; WUR Library (2018)
- Research data management; WUR Library (2018)
- Scientific publishing; WGS (2018)
- Reviewing a scientific manuscript; WGS (2022)
- Adobe InDesign; WGS (2023)

Scientific integrity/ethics in science activities (0.3 ECTS)

- Ethics in plant and environment sciences; WGS (2018)

PE&RC Annual meetings, seminars and the PE&RC retreat (2 ECTS)

- PE&RC First years retreat(2018)
- PE&RC Last years retreat (2022)
- Symposium resilience of agroecosystems (2022)

Discussion groups/local seminars or scientific meetings (4.8 ECTS)

- International symposium on animal environment and welfare (2017)
- WGS PhD Workshop carousel (2018)
- Emissions & nutrient management (2021-2023)
- WACASA meetings (2021-2023)
- Sustainable intensification of agricultural systems (2022-2023)
- Modelling & simulation discussion group (2022-2023)

International symposia, workshops and conferences (4.4 ECTS)

- International symposium on agricultural and rural environment protection and sustainable development; Tianjin, China (2019)
- The XXI International N workshop; Madrid, Spain (2022)

BSc/MSc thesis supervision (4 ECTS)

- Greenhouse gas and ammonia emissions from dairy manure composting
- Developing scenarios of sustainable production of feed and use of manure: a case study in China

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