Towards Improving the Manure Management Chain

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This dissertation is dedicated to

My beloved wife

Contents

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CHAPTER 1

General introduction

1.1 The need for mitigating emissions from animal manure

The expansion of global livestock production during recent decades is driven by an increasing demand for livestock products, which follows from human population growth, income growth, urbanization and shifts in dietary preferences (Steinfeld *et al.*, 2006; Thornton, 2010). The livestock sector becomes one of the largest current users of national resources. Livestock production systems occupy about 30% of the planet's ice-free surface, use one-third of the freshwater, and one-third of global cropland as feed (McMichael *et al.*, 2007; Herrero *et al.*, 2013). The growth of livestock production also leads to an increase in the production of animal manures. Animal manures contain all 14 nutrient elements that are essential for plant growth and development. Return of manure nutrients to crop land is crucial for closing the nutrient cycle. The plant nutrients in manure, if used appropriately, can replace significant amount of mineral fertilizers, and the organic matter in manure can improve soil quality and can also be used for energy production. Globally, livestock excretes 80-140 Tg nitrogen (N) and 20-30 Tg phosphorus (P) per year, which are 1-2 times the current amounts of N and P in mineral fertilizers consumed (Sheldrick *et al.*, 2003; Potter *et al.*, 2010).

However, improper management and utilization of manure results in nutrient losses, which may lead to negative impacts on the environment locally and globally. It has been estimated that globally only 20-40% of the N in animal excreta is returned to agricultural land, while the remainder is dissipated into the environment (Oenema & Tamminga, 2005). The livestock sector is responsible for 40-65% of global anthropogenic ammonia (NH_3) emissions, 40-60% of nitrous oxide (N_2O) emissions, and 30-40% of methane (CH_4) emissions (Bouwman *et al.*, 1997; Galloway *et al.*, 2004; Oenema *et al.*, 2005). The livestock sector contributes nearly 80% to the total greenhouse gas (GHG) emissions from the global agricultural sector (Steinfeld *et al.*, 2006).

In Europe, approximately 70% of the total land area utilized for agriculture was used in 2005 for animal feed and forage production (Lesschen *et al.*, 2011). About 60-65% of the plant biomass consumption in Europe is associated with the livestock sector (Krausmann *et al.*, 2008). The intensification of animal production during the 2nd half of the 20th century has led to high animal density in several regions of the European Union (EU) and has contributed to high nutrient surpluses regionally (Figure 1-1). These surpluses increase the risk of polluting the natural environment with nutrients (Velthof *et al.*, 2014). The total amount of N in animal excreta is as large as the amount of mineral N fertilizer currently used in Europe. Roughly 50-

60% of the N excreted in animal housing systems is effectively returned to cropland or grassland, while the remainder is lost during storage and following the application of manure to land (Oenema *et al.*, 2007). Animal manures are currently responsible for about 80% of the NH_3 emissions, 50% of the N_2O emissions, 15-25% of the CH_4 emissions from agriculture in EU (Oenema *et al.*, 2007; Leip *et al.*, 2015).

Clearly, animal manures are large sources of nutrients and organic carbon. Because of improper storage and management, manures are also a large source of NH_3 and N_2O emissions into the air, and of N and P losses to water bodies. There is an urgent need to decrease nutrient losses and emissions of NH_3 and GHG from animal manure management, and to increase the utilization of the nutrients in manure for feed and food production.

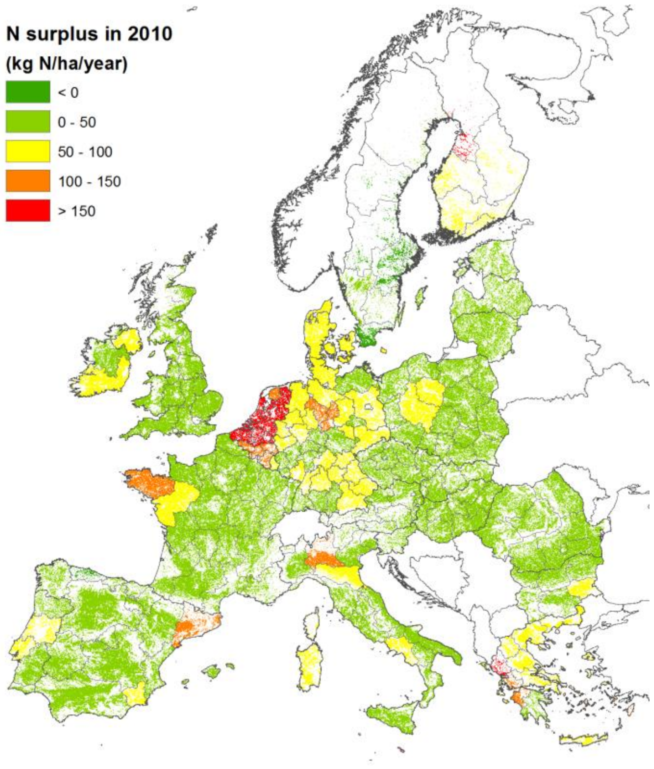

Figure 1-1. Soil nitrogen (N) surplus in the EU-27 in 2010 (kg N per ha of utilized agricultural area per year), based on the calculations with the MITERRA-Europe model (unpublished, Alterra).

1.2 Towards improved manure management

Manure management consists of a chain of activities and technologies (Figure 1-2). The manure management chain is diverse, depending on the combination of for example animal categories, feeding strategies, grazing activities, housing structures, manure storage and treatment facilities and manure application methods.

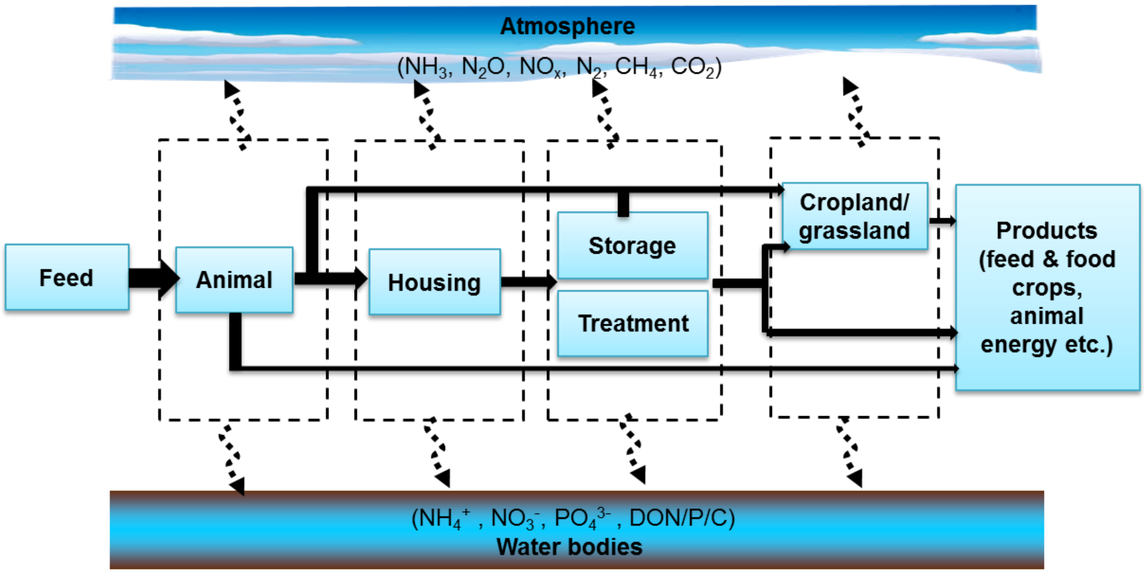

Figure 1-2. Schematic representation of the manure management chain. Solid arrows show the main flows of nutrients; dashed arrows show possible losses of nutrients to the atmosphere and to groundwater and surface waters.

A fraction of nutrients in feed are utilized by animals to produce marketed products (e.g. meat, eggs and milk); the remainder is excreted in urine and faeces. Animal excreta are generally handled in the housings as slurries (mixture of dung and urine) or solid manure (mixture of bedding material, dung and urine) with or without separate collection of liquids (mainly urine). The slurries, solid manure and liquids are often removed periodically to outdoor storages (tanks, lagoons or heaps with or without covers and leak-tight floors). After storage, the slurries, solid manure and liquids are applied to land. Urine and faeces from grazing animals are directly dropped in pastures, and are generally not collected. Nutrients in manure are partially used to fertilize cropland for feed production, thereby closing the nutrient cycle between crop and animal production systems (Oenema *et al.*, 2007). In the whole chain from animal feeding to manure application to land, a range of measures can be taken to mitigate nutrient losses and to increase the utilization of the nutrients and organic matter contained in manures.

Nutrient losses and emissions of NH_3, CH_4 and N_2O may occur from animal manures at various management stages (Figure 1-2). Evidently, accurate estimates of the amounts of nutrients in animal manures is the starting point for estimating nutrient flows and losses in the manure management chain, and are also required for nutrient management planning (Sutton *et al.*, 2011). The fraction of the nutrients in animal feed that ends up in manure depends on animal category, feed composition and animal productivity. International and national statistics (e.g., Eurostat and FAO statistics) provide national data on animal production and animal number per animal category annually. However, there is lack of animal category-specific data on feed use and composition at national levels. Such data are needed for improving the accuracy of manure production estimates.

1.2.1 Ammonia emission mitigation measures

Emissions of NH_3 are often the largest N loss pathway. These emissions can contribute to the acidification and eutrophication of nitrogen-limited ecosystems (Sutton et al., 2008). Emissions of NH_3 also decrease the N fertilizer equivalent value of animal manure. In response to the negative effects of NH_3 emissions on the environment, series of policy measures have been proposed through the 1999 Gothenburg Protocol under the UN-ECE Convention on Long-Range Transboundary Air Pollution (LRTAP). This Protocol aims to abate acidification, eutrophication and ground-level ozone, and addresses also sulphur (SO_2), nitrogen oxides (NO_x), and volatile organic compounds (VOCs) emissions. The NH_3 emission mitigation measures of the Gothenburg Protocol have been implemented in the EU member states through the 2001/81/EC National Emission Ceiling Directive. This directive requires that Member States develop national programmes that aim at meeting agreed ceilings of national NH_3 emissions by 2010 and 2020. The suggested NH_3 emission mitigation measures have been described in detail in the Guidance Document for preventing and abating NH_3 emissions (Bittman *et al.*, 2014). Different NH_3 emission abatement measures may target at different stages of the manure management chain, such as low-emission animal housing, covered storages and low-emission manure application (Bittman *et al.*, 2014).

These measures typically consider NH_3 emissions only, although these measures may have possible side-effects. Introducing a measure to decrease emissions at a particular stage may affect emissions downstream in the chain (e.g. pollution swapping), or emissions of other pollutants (Petersen *et al.*, 2007; Velthof *et al.*, 2009; Reis *et al.*, 2015). For example, several studies have indicated that some NH_3 mitigation measures may increase N_2O emissions from slurry storages or during field application, or enhance CH_4 emissions from solid manures storages (e.g. Berg *et al.*, 2006; Szanto *et al.*, 2007; Hansen *et al.*, 2009). These unwanted side-effects decrease the effectiveness and efficiency of mitigation measures.

A systematic quantitative analysis of the effects of NH_3 mitigation measures on emissions of NH_3, N_2O and CH_4 is still lacking, especially in consideration of the whole manure management chain. The assessment of a measure or technology must include the effects of the performance of other management measures in the whole chain from animal feeding to animal excretion, in-house and outdoor manure storage, manure treatment and up to application/deposition of manure to crop land and grassland (Figure 1-2).

1.2.2 Manure treatment

In the EU, manure storage conditions and the maximum amount of manure to be applied per unit of agricultural area are regulated, through the 1991 EU Nitrates Directive. This Directive and the governmental incentives to replace fossil energy sources by renewable energy sources (e.g., Renewable Energy Directive-2009/28/EC) stimulate the development of manure treatment technologies, to increase the utilization of manure in terms of energy and nutrient sources, and to decrease manure surpluses in areas with high animal density (Menzi *et al.*, 2010; Sommer *et al.*, 2013).

A wide range of manure treatment technologies have been developed and, in part, applied in practice in EU, such as anaerobic digestion, liquid-solid separation, slurry acidification, composting, drying, and incineration (Foged *et al.*, 2011). Different treatment technologies have different objectives, including reducing manure volumes to facilitate transport, bioenergy production, emission mitigation, improving fertilizer value of manure, increasing stability of organic matter in manure, and sanitation and odor control (see Box-1.1). Treatment technologies alter the physical, chemical and/or biological characteristics of manure, and thus may affect emissions of NH_3, N_2O and CH_4 from the manure management chain. A large number of laboratory and pilot experiments have been carried out to analyze NH_3 and/or GHG emissions from processed manure. Most of these studies typically focus on a specific gas and/or emission source (i.e. manure storages or application of manure). Few studies have focused on whole-farm or life cycle effects of for example solid-liquid separation (ten Hoeve *et al.*, 2014), anaerobic digestion (Sandars *et al.*, 2003; Hamelin *et al.*, 2011; De Vries *et al.*, 2012; Mezzullo *et al.*, 2012), and pyrolysis and incineration processes

(Fernandez-Lopez *et al.*, 2015). Country-scale analyses of the effects of manure treatment technologies have not been conducted yet.

Clearly, there is a lack of quantitative insights about the current and future possible effects of manure treatment technologies on nutrient and GHG emissions, and on the amount of remaining manure nutrients at various stages of the ‘manure management chain’, at regional and national levels. There is also lack of knowledge about optimal combinations of manure treatment technologies. Optimizing the implementation of treatment technologies requires a ‘chain’ approach, because of the possible interactions between measures and technologies.

1.2.3 Stakeholder perceptions

Implementation of manure treatment technologies in practice is still limited in most EU countries. The diffusion and exploitation of cleaner technologies rely on a combination of factors including governmental policies, financial incentives, technical and service supports and social acceptance (Montalvo, 2008). The development of manure treatment involves stakeholders across government, industry, academia, extension service and agricultural production sectors. However, there is lack of understanding about the key factors influencing the adoption of treatment technologies in practice. Identifying stimulants for and obstacles to adoption of technology is an essential step to develop effective governmental supporting programs and marketing strategies to promote the development of manure treatment.

Box 1.1 - Brief description of manure treatment technologies that are currently implemented in Europe (Foged *et al.*, 2011)

- *Solid-liquid separation* has the objective of separating manure slurry into a dry matter and nutrient -rich solid fraction and a diluted liquid fraction.
- *Anaerobic digestion* includes a series of biological processes in which microorganisms break down organic molecules in absence of oxygen, resulting in the production of a mixture of gases (mainly methane and carbon dioxide) that can be used as bioenergy.
- *Acidification* involves the application of an acidic reagent (e.g., sulfuric acid) to manure slurry to lower its pH to 5.5, and thereby to reduce ammonia emissions and deactivate pathogens.
- *Biological nitrogen removal* (i.e. nitrification-denitrification) has the objective of biological conversion of ammonium to inert nitrogen gas (N_2) using classical nitrogen removal treatment, combining autotrophic nitrification and heterotrophic denitrification processes. The feedstock is generally the liquid fractions separated from slurry.
- *Membrane (ultra) filtration* targets the removal of solid particles from the separated liquid fractions in the range of 5 to 200 nm. *Reverse osmosis* aims to further separate dissolved components in permeates produced by ultrafiltration (or other treatments separating small particles), by using pressure to force a solvent through a semipermeable membrane that retains the solute on one side and allows the pure solvent to pass to the other side.
- *Composting* includes aerobic biological decomposition and stabilization processes under conditions that allow for the development of thermophilic micro-organisms that convert the solid manure into compost. Compost is sufficiently stable for storage and subsequent soil application.
- *Drying* aims at reducing the moisture content and offensive odor emissions of solid manures/poultry litters by external heating, which is usually followed by pelletizing, a process of compressing a material into a smaller, denser form.
- *Combustion/incineration* aims to complete the thermo-chemical oxidation of organic matter in manures in order to obtain recoverable heat, ashes and gasses. For feedstocks with high moisture contents, a pre-drying process is generally required to lower moisture contents to less than 50-60% (by weight).

1.3 Objectives

The first objective of the research described in this thesis is to increase the understanding of the agronomic and environmental impacts of the manure management chains in EU-27[1]. The second objective is to explore options towards improving the performance of the management chain, especially related to the mitigation of NH_3 and GHG (N_2O and CH_4) emissions, and to the utilization of manure N and P, through application of mitigation measures and manure treatment technologies. The utilization of manure N and P depends on the fraction of the excreta N and P retained in the manure throughout the manure management chain, on its fertilizer effectiveness values, and on the judicious application of the manure to cropland or grassland. The term 'recovery of manure N and P' has also been used to refer to the fraction of N and P excreted that ultimately returns to cropland or grassland.

This PhD thesis research was part of the Marie Curie Training Program ReUseWaste (Recovery and Use of Nutrients, Energy and Organic Matter from Animal Waste). In total eleven PhD students and two post docs have been working on various specific mitigation measures and treatment technologies (http://www.reusewaste.eu/). My research focused on the integral analysis of the whole manure management chain, and on the up-scaling and synthesis of research results, including those from literature.

The specific objectives of this thesis are to:

- Analyze the feed resource use and nutrient (especially N) excretion by animals in the EU-27 at country levels **(Chapters 2 and 3)**
- Assess the effects of NH_3 mitigation technologies on emissions of NH_3, CH_4 and N_2O from individual sources, and their whole-chain effects **(Chapter 4)**
- Analyze the contribution of manure treatment activities to emissions of NH_3, N_2O and CH_4 from manure management chains in EU-27 at country levels in 2010; and explore the potential whole-chain effects of these treatment technologies on emissions and nutrient recovery in the EU-27 **(Chapter 5)**
- Provide insights into stakeholders perceptions about factors that facilitate or hinder the adoption of treatment technology in practice **(Chapter 6)**

[1] In Chapters 3 and 5, the reference year of the analyses is 2010. Croatia is a EU member country since 2013, therefore the analyses focus on the EU-27.

1.4 Outline of this thesis

This thesis contains a general introduction (Chapter 1), five research chapters (Chapters 2-6) and a general discussion (Chapter 7).

Chapter 2 reviews N excretion factors of animal categories, and the guidelines and methodologies for the estimation of N excretion factors, as used by member states of the EU.

Chapter 3 presents a transparent and uniform methodology for estimating annual feed use and N excretion per animal category for all member states of the EU in 2010, based on the energy and protein requirements of the animals and statistics of feed use and composition, animal number and productivity.

Chapter 4 presents a meta-analysis of the effects of a set of NH_3 mitigation measures on NH_3, CH_4 and N_2O emissions from individual sources of the manure management chain, and an integrated assessment of whole-chain effects of these measures on emissions through scenario analysis.

Chapter 5 presents scenario analyses of effects of manure treatment technologies on the recovery of N and P, and emissions of NH_3, N_2O and CH_4 in manure management chains in EU at country level.

Chapter 6 reports on a survey study about stakeholder perceptions of manure treatment technologies in Denmark, Italy, the Netherlands and Spain; which factors facilitate or hinder the implementation in practice of manure treatment technologies?.

Chapter 7 provides a general discussion of the main findings of the study, and identifies remaining future research needs.

Chapters 2, 3, 4, 6 have been published in peer-reviewed journals. Chapter 5 is accepted subject to revision.

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CHAPTER 2

Nitrogen excretion factors of livestock in the European Union: a review

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Abstract

Livestock manures are major sources of nutrients, used for the fertilization of cropland and grassland. Accurate estimates of the amounts of nutrients in livestock manures are required for nutrient management planning, but also for estimating nitrogen (N) budgets and emissions to the environment. Here we report on N excretion factors for a range of animal categories in policy reports by member states of the European Union (EU). Nitrogen excretion is defined in this paper as the total amount of N excreted by livestock per year as urine and faeces. We discuss the guidelines and methodologies for the estimation of N excretion factors by the EU Nitrates Directive, the OECD/Eurostat gross N balance guidebook, the EMEP/EEA Guidebook and the IPCC Guidelines. Our results show that N excretion factors for dairy cattle, other cattle, pigs, laying hens, broilers, sheep, and goats differ significantly between policy reports and between countries. Part of these differences may be related to differences in animal production (e.g. production of meat, milk and eggs), size/weight of the animals, and feed composition, but partly also to differences in the aggregation of livestock categories and estimation procedures. The methodologies and data used by member states are often not well described. There is a need for a common, harmonized methodology and procedure for the estimation of N excretion factors, to arrive at a common basis for the estimation of the production of manure N and N balances, and emissions of ammonia (NH_3) and nitrous oxide (N_2O) across the EU.

2.1 Introduction

Livestock manures are major sources of nutrients, to be used for the fertilization of cropland and grassland. Accurate estimates of the amounts of nutrients in livestock manures are required for nutrient management planning, but also for estimating nitrogen (N) budgets and emissions of N to the environment (Sutton *et al.*, 2011). Nitrogen excretion is defined in this paper as the total amount of N excreted by livestock per year as urine and faeces. The total amount of N excreted as manure by livestock in the member states of the European Union (EU) was 9.2 Tg in 2011, which is only 15% less than the amount of 10.6 Tg artificial fertilizer N consumed (Source: Eurostat statistics). The N in manure is vulnerable to losses via emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrogen oxides (NO_x) to the atmosphere and via leaching of nitrate (NO_3^-), ammonium (NH_4^+) and organic N to groundwater and surface waters. Emissions of NH_3 occur from manure N in animal houses, manure storages, and following the application of manure to soils, including the droppings from grazing animals in pastures (Webb *et al.*, 2013). Deposition of atmospheric NH_3 to land and surface waters results in eutrophication of ecosystems, acidification and loss of biodiversity. Moreover, NH_3 may form fine particles (PM) in the atmosphere, which negatively affect human health (Moldanová *et al.*, 2011). The nitrification and denitrification of N from manure may release nitrous oxide (N_2O), nitrogen oxide (NO_x) and dinitrogen (N_2) into the atmosphere. Gaseous N emissions occur from manure applied to land, deposition of excreta by grazing animals, and from animal houses and manure storages. Nitrous oxide is a potent greenhouse gas and is an important cause for depletion of stratospheric ozone (Ravishankara *et al.*, 2009; IPCC, 2014). Nitrogen oxides can react with carbon monoxide and volatile organic compounds in air, resulting in the formation of tropospheric or ground level ozone (O_3). Ground-level O_3 pollution has several negative effects on health (Moldanová *et al.*, 2011). Dinitrogen is harmless for the environment, but N_2 losses like other N loss pathways decrease the N use efficiency of agricultural systems and, by that, increases the need for N fertilizers. Application of manure and droppings of grazing animals may also result in N leaching to groundwater and surface waters, which subsequently may cause eutrophication of surface water and pollution of groundwater as drinking water (Camargo and Alonso, 2006; WHO, 2011).

Several policies have been implemented by the European Union (EU) and United Nations (UN) bodies to improve the utilization of manure nutrients in agriculture and to decrease N emissions to the environment (Oenema *et al.*, 2011). These policies include the EU National Emission Ceiling Directive (European Commission, 2001) and the Gothenborg protocol of the UN-ECE Convention on Long-range Transboundary Air Pollution (UNECE, 1999) to decrease NH_3 and NO_x emissions, the UN-FCCC Kyoto protocol to decrease N_2O emission (UNFCCC, 1997), and the EU Nitrates Directive to decrease N leaching to groundwater and surface waters (European Commission, 1991). In addition, OECD and Eurostat developed agri-environmental indicators for monitoring of the pressures by agriculture on the environment and of the state of the environment (European Commission, 2006; OECD, 2013).

Evidently, there is a need for accurate estimates of N excretion by livestock, to be able to calculate and monitor manure production, N balances of agriculture and emissions of N to the environment. Currently, there are no uniform, standard and accepted methodologies and terminologies for estimating the amounts of N and P in animal excrements. Guidelines and recommended methodologies for the estimation of N excretion and associated emissions of NH_3 and NO_x (EEA, 2013) and N_2O (IPCC, 2006) are given by the Gothenborg protocol and Kyoto protocol, respectively. The EU Nitrates Directive includes a maximum manure application standard of 170 kg N per ha, to be applied in so-called nitrate vulnerable zones. Guidelines to calculate manure N application rates are available (Ketelaars and Van der Meer, 1999). The OECD and Eurostat give also recommendations for the calculation of N excretion, as part of the gross N balance indicator (OECD/Eurostat, 2007).

Further, EU policies are evaluated or underpinned using integrated assessment models such as GAINS (http://gains.iiasa.ac.at/models/) and CAPRI (www.capri-model.org). GAINS is used for emission projections of greenhouse gases and atmospheric pollutants, including NH_3. CAPRI is an economic model used for exploring reforms of for example the Common Agricultural Policies (CAP) and biofuel policies. CAPRI includes a module for the calculation of N balances and greenhouse gas emissions. GAINS (Klimont and Brink, 2004) and CAPRI (Leip *et al.*, 2010) both use methodologies for the estimation N excretion by livestock.

EU member States tend to use methods to estimate excretion which they have developed and improved over time, and sometimes use different methodologies for different reporting requirements (Oenema *et al.*, 2011). This make comparisons between countries and estimates at EU level complicated.

Here, we present a review of the methodologies for the calculation of N excretion factors in EU, and provide a comparison of the N excretion factors used for different policy streams and models by the EU member states. First, we present a review of the used methodologies. Thereafter, the reported N excretion factors for different livestock categories and EU member states are summarized and discussed. Finally, we present recommendations for a common, harmonized methodology for estimating N excretion factors.

2.2 Review of methods to estimate nitrogen excretion factors

2.2.1 Type of methods

The most common method for deriving N excretion factors is the input-output balance of the animal, which assumes that the amount of N excreted in faeces and urine is equal to the total amount of feed N consumed minus the N retained in marketed products (milk, meat, eggs, live weight gain). Hence, N excretion = N intake – N retention.

Two approaches are used for the estimation of N excretion factors, i.e. the gross and net N excretion. The gross N excretion is the total N excretion in urine and faeces from an animal without accounting for any N loss after excretion. The net N excretion is defined as the total amount of manure N (mixture of collected urine and faeces) that is applied to land, i.e. the net N excretion is defined as the gross N excretion corrected for N losses from housing systems and manure storages. The amount of N consumed by the animal depends on the feed intake by the animal, and the N content of the feed. Total feed intake depends on the maintenance cost and production level of the animal (e.g. growth rate, milk and egg production), and the feeding value and digestibility of the feed. Data on the annual N retention in meat, egg, milk, or wool produced is usually derived from production statistics and scientific reports about the N contents in animal products. The IPCC guidelines present default values of N retention (IPCC, 2006). More details about the gross N excretion methods are provided in the following sections.

The net excretion can be calculated from i) the gross N excretion corrected for gaseous N emissions (as NH_3, N_2O, N_2, and NOx) from housing systems and manure storages, or ii) from the volume or mass and the N content of the manure in the manure storage just before application to land. Estimates for gaseous N losses from housing systems and manure storages can be based on the default emission factors in the guidelines for NH_3 (EEA, 2013) and N_2O

emissions (IPCC, 2006). Some countries have derived country specific emission factors, taking county specific housing systems and conditions into account, e.g. Denmark (Hutchings *et al.*, 2001), France (Gac *et al.*, 2007), Ireland (Hyde *et al.*, 2003), Germany (Dämmgen *et al.*, 2006), The Netherlands (Velthof *et al.*, 2012), and UK (Webb and Misselbrook, 2004).

For the second approach, the volume or mass and the N content of produced manure have to be estimated. Estimating the mass or volume and N content of manures in storages is not without discussion. One option is to use default factors for manure excretion per animal and to correct for changes in volume through for example water entering manure storages (rain, spilling and cleaning water), additions of bedding materials like straw and dust, evaporation of water and compaction processes. The volume or mass can also be estimated by measurement of the dimension of the manure storage or tanker or by weighing. For translation of the volume in mass and vice versa, manure density factors are needed. The N content of manures are either measured or based on defaults. Manures are often very heterogeneous, and the variation in N contents is very large. As a consequence, a large number of samples is needed to obtain an accurate average N content, using well-designed sampling protocols. In the Netherlands, all transports of manure have to be sampled and analyzed on N and P contents because of legislation. The results show large variations in N contents between livestock categories and within a livestock category (Table 2-1). The variation within a livestock category is largest for solid manure. For most manures the variation is larger for P than for N. The large variation of N and P contents in solid manure is due to the heterogeneous distribution of nutrients in these manures, and also due to the diversity in solid management systems. Nicholson *et al.* (1996) also found large variations in the composition of solid poultry manures in England and Wales.

The calculation of gross and net N excretion factors is reported on different spatial scales. Generally, NH_3 and N_2O emissions are reported on a national level for the Gothenborg and Kyoto protocols, nutrient balances at regional to national level for the OECD/Eurostat reports, and the manure production at farm level for the Nitrates Directive.

Table 2-1. Average and standard deviation (SD) of N and P (expressed in P_2O_5) contents of livestock manure in the Netherlands (Van Bruggen, 2014), based on data of 2011-2013 of the Dutch Ministry of Economic Affairs.

Animal and manure type		No of samples	N (g kg^{-1} manure)		P_2O_5 (g kg^{-1} manure)	
			Average	SD	Average	SD
Cattle	Solid cow manure	10436	7.7	5.7	4.3	5.2
	Liquid fraction separated cow slurry	1136	4.5	2.1	1.8	0.8
	Cow slurry	201087	4.0	0.7	1.5	0.4
	Calve (white meat) slurry	46526	3.1	1.0	1.4	1.1
	Calve (red meat) slurry	24216	5.6	1.7	2.3	0.8
Poultry	Chicken slurry	602	10.0	3.2	6.0	2.5
	Chicken manure (manure belt)	9477	28.4	9.9	23.0	10.5
	Chicken dried manure (manure belt)	9149	32.7	9.4	25.9	8.9
	Chicken litter	29107	29.0	8.6	25.6	8.0
	Broiler	27266	34.1	6.7	16.6	5.8
	Turkey	2053	32.7	8.7	22.5	5.9
	Duck	1268	10.7	6.6	9.7	6.5
Pigs	Solid pig manure	3895	13.9	14	13.6	13.5
	Liquid fraction separated pig slurry	17672	1.9	1.8	0.8	0.8
	Sow slurry	70598	3.8	1.7	2.5	1.8
	Fattening pig slurry	118616	7.0	1.9	3.9	1.5
Sheep		256	8.7	2.7	5.1	2.4
Goat		11135	9.7	10.1	5.4	7.2
Horses		2585	5.6	2.4	3.0	1.5
Rabbits		964	13.6	5.4	12.6	5.1

2.2.2 Methodologies in the IPCC guidelines

Countries have to report annually greenhouse gas emissions to the UNFCCC; the National Greenhouse Gas Inventories (NIR). The method used to calculate greenhouse gas emissions is based on the IPCC guidelines. The IPCC Guidelines include recommendations for NIR. These guidelines also include recommendations for the calculation of N excretion factors, to be able to calculate direct and indirect N_2O emissions, and thereby NH_3 emissions and NO_3 (nitrate) leaching. Losses of NH_3 and NO_3 are indirect sources of N_2O emissions. The IPCC Guidelines contain recommendations at different levels of detail, i.e. the Tier levels. The Tier 1 approach is the most simple method and includes default estimates of N excretion. The Tier 2 and 3 approaches are more detailed and include country specific estimates and/or models. Nitrogen excretion factors have be determined for each livestock category.

The Tier 1 excretion factors are expressed in kg N 1000 kg^{-1} animal day^{-1}. These rates can be applied to livestock sub-categories of varying ages and growth stages using a typical average animal mass (TAM) for that population sub-category. The IPCC Guidelines include default TAM coefficients, but it is recommended to use country-specific TAM factors.

For Tier 2, the annual N excretion is calculated from the N intake and N retention data as Nex(T) = Nintake(T) * [1 - Nretention(T)], where Nex(T) is the annual N excretion rate of animal of species/category T (kg N $animal^{-1}$ $year^{-1}$), Nintake(T) is the annual N intake per head of animal of species/category T (kg N per animal per year), and Nretention(T) is the dimensionless fraction of annual N intake that is retained by animal of species/category T. Default N retention fractions are provided in the IPCC Guidelines: 0.20 for dairy cows, 0.07 for other cows, 0.30 for pigs and poultry, 0.10 for sheep and goats, and 0.07 for horses. The IPCC guidelines recommend the use of country specific N intake and retention data for each livestock category, which should be based on national data statistics and information from animal nutrition specialists.

For Tier 3, country-specific methods and models are being used. These methods and models should be well documented and reviewed, and should clearly describe the estimation procedures.

Our analyses of the national inventory reports for 2011 show that most Member States use a country specific approach for N excretion estimates (Table 2-2). Some member states use a country specific approach for the main livestock categories and the IPCC default values for the other categories.

2.2.3 Methodologies in the EMEP/EEA inventory guidebook

The EMEP/EEA air pollutant emission inventory guidebook provides guidance on estimating emissions from both anthropogenic and natural sources of gaseous pollutants, including NH_3 (EEA, 2013) The Tier 1 approach for NH_3 emissions in this guidebook is not based on N excretion factor. It includes default NH_3 emission factors in kg NH_3 animal^{-1}.

The Tier 2 and Tier 3 are based on emission factors, which relate excreted total N or excreted total ammoniacal nitrogen (TAN) to NH_3 emissions. The Tier 2 approach is based on default N excretion factors and comparable to the IPCC Tier 1. For Tier 3 country-specific models and country specific data are required. There is no restriction on the form of Tier 3, provided it can be demonstrated that the estimated emissions with Tier 3 are more accurate than those with Tier 2. If data are available, emission calculations may be made for a larger number of livestock categories than listed under Tier 2. Generally, standard values for TAN, per animal category, are used to calculate the TAN excretion (Reidy *et al.*, 2008). Default values for the fraction of TAN in total N range from 0.5 – 0.7 (EEA, 2013). In the Netherlands, TAN values are calculated annually on the basis of data statistics about feed composition (Velthof *et al.*, 2012).

Our analyses of national inventory reports for the Gothenborg protocol shows that 11 countries use a Tier 1 methodology (not based on N excretion factors), and the other use (combinations of) Tier 2 excretion factors and country specific excretion factors (Table 2-2).

Table 2-2. Sources/methods of nitrogen excretion used in the National Inventory Reports of UNFCCC (NIR), reports for Gothenborg protocol and action programmes of the Nitrates Directive.

Country	NIR	Gothenborg protocol	Nitrates Directive
Austria	N balance[a]	As NIR	Country specific net excretion
Belgium	Country specific	As NIR	As NIR for Flanders. Gross excretion for Walloon
Bulgaria	IPPC default	Tier 1 - NH_3 based[b]	N content and volume of manure[c]
Cyprus	IPPC default	Tier 1 - NH_3 based	N content and volume of manure
Czech Rep.	IPPC default	Not clear	N content and volume of manure
Denmark	N balance	N balance	As NIR; corrected for gaseous N loss
Estonia	IPPC default, except dairy cow	Tier 1 - NH_3 based	N content and volume of manure
Finland	N balance	As NIR	N balance
France	IPPC default	Tier 1 - NH_3 based	N balance; corrected for gaseous N loss
Germany	Region specific N balance	As NIR	Country specific gross excretion. Method not indicated
Greece	IPPC default	Not indicated	N content and volume of manure
Hungary	IPPC default	Tier 1 - NH_3 based	Country specific net excretion, based on literature
Ireland	N balance	Tier 1 - NH_3 based	As NIR
Italy	N balance	As NIR	As NIR
Latvia	N balance	Tier 1 - NH_3 based	N content and volume of manure
Lithuania	N balance for cattle and pigs. Other default IPCC	Tier 1 - NH_3 based	Net excretion based on N balance and gaseous N loss
Luxembourg	Country specific; method not indicated	Tier 1 - NH_3 based	Not indicated
Malta	Country specific; method not indicated	Not indicated	Not indicated
Netherlands	N balance	As NIR	As NIR
Poland	Country specific for dairy cattle, pigs, and horses	Country specific	N content and volume of manure
Portugal	Country specific	As NIR	N content and volume of manure
Romania	IPPC default	Tier 1 - NH_3 based	As NIR
Slovakia	IPPC default	Tier 1 - NH_3 based	N content and volume of manure
Slovenia	Dairy cattle based on milk production	As NIR	Country specific net excretion. Method not indicated.
Spain	N balance	As NIR	Country specific gross excretion. Method not indicated.
Sweden	Country specific model	Country specific model	Country specific model
UK	N balance	As NIR	As NIR

[a] Input – output balance of the animal
[b] Emission of NH_3 is expressed in kg NH_3 per animal; N excretion is not used in Tier 1 approach
[c] The manure production/ N excretion calculated from volume of manure and the N content of the manure

2.2.4 Methodologies for the EU Nitrates Directive

The EU Nitrates Directive has the objective to decrease nitrate leaching to groundwater and surface waters. One of the measures in the Nitrates Directive is a maximum application standard of manure of 170 kg N per ha per year. Farmers have to calculate the amount of manure N that is produced on their farm and that can be applied to land. Member States of the EU have to report N excretion values in their Nitrates Directive Action Programmes for so-called Nitrate Vulnerable Zones. Our analyses indicate that the approaches for the calculation of the N excretion per livestock category differ between the Member States (Table 2-2). Many countries use a feed balance method to calculate N excretion for a large number of animal categories, but some countries use default estimates of N excretion, and others use a method based on estimating the volume or mass of manure and the N content of the manure.

The 1999 study "Establishment of criteria for the assessment of the nitrogen content of animal manures" presents guidelines for the calculation of manure N production, to be used within the framework of the EU Nitrates Directive (Ketelaars and Van der Meer, 1999). The manure N production is calculated as: Nmanure = Ndiet – Nanimal products – Nlosses from buildings and manure storage, where Nmanure is the manure production in kg N animal^{-1} year^{-1}, Ndiet is the feed N consumed in kg N animal^{-1} year^{-1}, Nanimal products is N retention (N in animal products) in kg N animal^{-1} year^{-1}, and Nlosses from buildings and manure storage are the gaseous N losses in animal housings and manure storages, in kg N animal^{-1} year^{-1}. Details of feed composition, daily feed intake, N contents of products, production rounds per year, and feed conversion ratios are included in the guidelines (Ketelaars and Van der Meer, 1999). North Italy (Xiccato *et al.*, 2005) and UK (ADAS, 2007) use these guidelines for the calculation of manure N production.

2.2.5 Methodologies used for the OECD/EUROSTAT Gross N balance

The methodology for deriving the Eurostat/OECD nutrient balances are described in the OECD/Eurostat (2007) Gross Nitrogen Balances Handbook. The following countries submitted country specific N excretion factors to Eurostat for 2011: Austria, Belgium, Czech republic, Denmark, Finland, Germany, Hungary, Ireland, Italy, Malta, Netherlands, Poland, Portugal, Slovakia, Slovenia, and Sweden. Often similar methodologies were used as for the UNFCCC reporting. Some countries have not supplied data to Eurostat/OECD or the data were not compliant with the guidelines in the OECD/Eurostat Handbook. For these countries

Eurostat/OECD used the N excretion factors reported in the 2010 submission of greenhouse gases to the UNFCCC.

2.2.6 Methodologies used in the GAINS model

The GAINS model estimates emissions of NH_3, N_2O and CH_4 from animal manures per livestock category and Member State. GAINS relies on data submitted by national experts, based on a questionnaires (Klimont and Brink, 2004; Asman et al., 2011). The questionnaires included questions for dairy cattle and other cattle, fattening pigs, sows, horses, sheep and goats, laying hens, broilers, other poultry (geese, ducks, turkey), and fur animals. If no national information was provided, assumptions were made. If no country-specific data for dairy cattle was provided, a relationship between milk yield and N-excretion was used: Nx = 0.0178 x M + 0.2271, where Nx = N excretion rate (kg N animal^{-1} year^{-1}), and M = milk yield (kg animal^{-1} year^{-1}). The available data do not allow conclusions for milk yields below 3500 kg milk year^{-1}; then a mean N excretion value of 50 kg N animal^{-1} year^{-1} is assumed, irrespective of milk yield.

2.2.7 Methodologies used in the CAPRI model

The CAPRI (Common Agricultural Policy Regional Impact) model is an agricultural sector model covering the whole of EU, Norway and Western Balkans at regional level (250 regions), and links global agricultural markets at country or country block level. Besides economic evaluations, CAPRI is also used for the calculation of balances for N, P, K and greenhouse gas emissions (Leip *et al.*, 2010, 2011). The excretion of N per animal category is calculated as the difference between N intake and N retention of animals. CAPRI includes a feed module in which available feed in a region/country is distributed over the animals in a region, based on energy and nutrient requirements of the animals. Feed statistics from FAO and Eurostat are used. The yields and N content of grassland are based on estimates made in the MITERRA-model (Velthof *et al.*, 2009).

2.3 Nitrogen excretion factors

Table 2-3 shows the calculated N excretion factors for EU countries using the default IPCC Tier 1 factors, the EMEP/EEA Tier 2 approach, and the gross N excretions in the Nitrates Directive report. For the Nitrates Directive study, ranges are presented, which reflect differences in breed, N content of the diet, protein conversion ratio, and live weight. There is a wide range in the N excretion factors of dairy cattle (75 to 184 kg N animal^{-1} year^{-1}) and other cattle (20 – 90 kg N animal^{-1} year^{-1}). The IPCC uses defaults for Western and Eastern Europe and EMEP/EEA uses only one default. In the EMEP/EEA Guidelines it is indicated that the default N excretion data were taken from IPCC. However, it is not indicated if Western or Eastern European excretion coefficients were used and which animal category was selected. For most animal categories, the EMEP/EEA defaults are different from the IPCC defaults, when using the TAM values in the IPCC report. It is not clear how the EMEP/EEA Tier 2 defaults have been obtained for the IPCC Tier 1 defaults. If the ranges provided by the Nitrates Directive report reflect the actual variations in practice, it will be difficult to accept that the default values presented by the IPCC Tier 1 and the EMEP/EEA Tier 2 approaches have high accuracy, even though the factors do not differ much between the two latter approaches.

Table 2-3. N excretion factors in kg N animal^{-1} yr^{-1} according to IPCC Tier 1 guidelines, EMEP/EEA Tier 2 guidebook, and recommendations for Nitrates Directive.

Animal	IPCC Tier 1[a]		EMEP/EEA Tier 2	Nitrates Directive[e]	Explanation of range in Nitrates Directive factors
	Western Europe	Eastern Europe			
Dairy cattle	105	70	105	75 - 184	Small breed low N diet to large breed high N diet
Other cattle	51	50	41	20 - 90	Growing cattle low N diet to suckler cow high N diet
Fattening pig[b]	9.3	10	12	12 - 15	High to low protein conversion
Sow[c]	30	30	35	32 - 38	High to low protein conversion (incl. piglets < 25 kg)
Layers[d]	0.63	0.54	0.77	0.64 - 0.96	Good to poor feed conversion
Broilers	0.36	0.36	0.36	0.41 - 0.61	Good to poor feed conversion
Turkeys	1.84	1.84	1.64	1.6 - 2.5	Good to poor feed conversion
Ducks	0.82	0.82	1.26	1.0 - 1.4	Good to poor feed conversion
Sheep	15	16	16	15 - 30	Low to high N diet (incl. lambs < 40 kg)
Goats	18	18	16	16 - 26	Low to high N diet (incl. lids < 7 kg)
Horses	36	41	48	39 - 67	400 to 800 kg
Rabbits	8.1	8.1	-	7 - 12	Low to high N diet (incl. kittens)

[a]Calculated from IPCC Tier 1 tables with excretion per mass per day and IPCC default mass values (TAM)
[b]Market swine in IPCC
[c]Breeding swine in IPCC
[d]Hens ≥ 1 yr in IPCC
[e] Ketelaars and Van der Meer (1999)

Tables 2-4 to 2-7 show the N excretion factors for dairy cattle and other cattle, pips, poultry, and sheep and goats, respectively, reported in i) the National Inventory Reports (NIR) for 2011, ii) the reports to UNECE Gothenborg protocol for the years 2007 – 2010 (years differ between countries), (iii) the Action Programmes for the Nitrates Directive available for 2011, (iv) the OECD/Nutrient balance for 2011, (v) the GAINS model (Asman *et al.*, 2011) and (vi) the CAPRI model (Leip *et al.*, 2010). Member states using a Tier 1 approach for the Gothenborg Protocol do not report N excretion factors. For the Nitrates Directives, only data are presented for countries that present gross N excretion factors or net N excretion factors and gaseous N emission factors from which gross N excretion can be calculated.

The N excretion of dairy cattle varied from less than 80 kg N cow^{-1} year^{-1} to more than 140 kg N cow^{-1} year^{-1} (Table 2-4). This large range is partly due to differences in breed, N content of the diet and level of milk production (Ketelaars and Van der Meer, 1999). However, differences in methods and in data combination are also causes of the variation shown in Table 2-4. For nearly all countries, there are differences in N excretion of dairy cattle between the policy reports. The excretion factors used in National Inventory Reports for UNFCCC and for the Gothenborg protocol are for most member states similar or only lightly different (except Germany; 132 kg year^{-1} for NIR 2011 and 114 kg year^{-1} for the Gothenborg protocol). The CAPRI estimates were very high for some countries, e.g. 194 kg N cow^{-1} year^{-1} in Denmark and 180 kg N cow^{-1} year^{-1} in Sweden. One of the uncertainties in calculation of N excretion of dairy cows, is the estimate of grassland yields and the N content of the grassland (Velthof *et al.*, 2009). Grassland yields are not yet recorded in statistics, as harvested and grazed grass is generally used on the own farm. Lesschen *et al.* (2011) estimated that the EU livestock sector uses around 500 million tonnes of animal feed per year, 40% of which is grass (expressed in dry matter). Grasslands are highly diverse in terms of management, yield and biodiversity value, which has a large effect on the N and P content of the herbage (Whitehead, 2000). Calculations show that the N excretion of dairy cows decreases on average with 0.17 kg N kg^{-1} decrease in fertilizer N input, in the range of 200 to 400 kg N ha^{-1} year^{-1} (Velthof *et al.*, 2014).

The category ‘other cattle’ consists of cattle with different breeds, weight and age, and thereby with large differences in N excretion factors, from less than 15 kg N head^{-1} for calves to more than 75 kg N for beef cattle and suckling cows (Table 2-4). This hampers the estimation of an average excretion rates for the category “other cattle”. Evidently, excretion factors for the different functional categories within the category ‘other cattle’ have to be estimated first, and then aggregated proportionally to arrive at a mean excretion factor for ‘other cattle’. The dominant functional categories within other cattle should be described as well to allow comparison. In conclusion, it is recommended to estimate excretion factors for functional cattle categories within other cattle instead of using an average excretion factor for “other cattle”. This is also needed because the excretion factors have to be multiplied with the appropriate number of animals, which mostly are registered for detailed categories. This procedure leads to a transparent and accurate estimation of the N excretion factor for other cattle.

There is also large variation in N excretion factors for pigs (Table 2-5). The N excretion factors for pigs in NIR range from 8.4 kg N head^{-1} per year for Denmark to 20 kg N head^{-1} per year^{-1} for Romania and Czech Republic. The pig category also consists of a large number of different types (breeds, sex, age, weight). Further, it may not be excluded that some countries express the N excretion factors on an animal head basis and others on an animal place basis. For sows, it is important to report whether piglets are included or not (and till which weight and age). The CAPRI model calculates livestock feed using a feed distribution tool and data from market balances on a country level, regional fodder availability and animal requirements (Leip *et al.*, 2010). The CAPRI estimates for N excretion factors for pigs are much higher than that for the other methods (Table 2-5).

The same comments as for other cattle and pigs hold for poultry. There is a large diversity in poultry categories (laying hens, broilers, turkey, ducks etc.), which hampers the use of one excretion factor for one poultry category. There are sometimes large differences in N excretion factors between countries, which are difficult to explain (Table 2-6). Differences may be partly due to differences in age/weight, and unit (animal head or animal place).

Large differences in N excretion factors between sheep categories are shown in the UNFCCC report; N excretion factors range from 5.2 kg N head^{-1} for Spain to 20 kg N head^{-1} in Slovenia (Table 2-7). Also the figures used for the Nitrates Directive show large differences for sheep. These differences may partly be due to differences in the way male, female and young animals are considered in the excretion calculation.

Table 2-4. Gross nitrogen excretion factors for dairy cows and other cattle (kg N $head^{-1}$ yr^{-1}) in EU-27 for different sources.

	Dairy cows						Other cattle					
Country	NIR (2011)	UNECE (2007-2010)	Nitrates Directive (2011) [a]	OECD/ Eurostat (2011)	CAPRI (2004)	GAINS (2010)	NIR (2011)[b]	UNECE (2007-2010)[a]	Nitrates Directive (2011)[a]	OECD/ Eurostat (2011)[a]	CAPRI (2004)	GAINS (2010)
Austria	97	97	-	97	90	106	47	26-74	-	15-69	40	46
Belgium	115	-	-	109	95	118	54	-	-	-	47	50
Walloon	-	121	-	111	-	-	-	11-97	-	11-77	-	-
Flanders	-	97	97	106	-	-	-	13-98	23-61	11-77	-	-
Bulgaria	70	-	-	-	116	75	50	45	-	-	49	45
Cyprus	-	-	-	107	134	103	-	-	-	19-48	43	40
Czech Rep.	145	-	-	105	114	131	70	-	-	20-60	43	45
Denmark	138	138	-	129	194	132	48	-	-	11-66	62	37
Estonia	102	-	-	62	122	113	44	-	-	11-45	42	45
Finland	127	122	-	-	92	121	50	38-66	-	16-51	30	53
France	100	-	-	125	105	112	58	-	-	53-67	53	50
Germany	132	114	100-149	119	106	130	41	44-88	60-87	84	40	40
Greece	100	-	-	-	97	111	45	-	-	25	47	45
Hungary	114	-	-	125	149	146	48	-	-	12-65	51	45
Ireland	85	-	85	109	88	105	49	-	24-65	14-55	48	69
Italy	116	116	-	94	97	112	49	50	-	63-74	39	47
Latvia	70	-	-	70	139	88	50	-	-	-	57	51
Lithuania	99	-	120	-	99	95	58	-	33-95	-	38	50
Luxembourg	102	-	-	71	-	114	68	-	-	10	-	42
Malta	-	-	-	103	155	98	-	-		10-28	51	40
Netherlands	127	130	99-131	135	119	147	83	12-83	35-71	16-75	38	40
Poland	87	-	-	70	91	81	58	-	-	36-40	36	35
Portugal	115	112	-	112	121	102	51	25-80	-	25-55	68	50
Romania	70	-	-		96	67	50	-	-	21	39	53
Slovakia	100	-	-	105	119	135	60	-	-	21-60	42	45
Slovenia	111	-	-	113	85	110	42	-	-	27-35	38	40
Spain	68	68	89	103	108	71	53	52	-	21-73	51	45
Sweden	126	125	117-139	117	180	132	42	28-63	22-63	26-57	61	39
UK	110	-	-	117	142	133	55	-	-	31-63	53	49

[a]Range in N excretion factors indicate that more categories of cows are included, e.g. suckler cows, calves, young cattle (<1 yr or >1 yr), or bulls.
[b]Data: from common reporting format (CRF) of greenhouse gas inventory submission of 2011.

Table 2-5. Gross nitrogen excretion factors (kg N head^{-1} yr^{-1}) for pigs.

Country	NIR (2011)[a]	UNECE (2007-2010)[b]	Nitrates Directive (2011)[b]	OECD/ Eurostat (2011)[b]	CAPRI (2004)	GAINS (2010)
Austria	9.6	10-29	-	4-24	17.3	9.0
Belgium	10.1	-	-	-	18.4	11.1
Walloon	-	2-22	-	2-21	-	-
Flanders	-	5-43	13-24	2-21	-	-
Bulgaria	20.0	-	-	20	21.6	12.4
Cyprus	-	-	-	3-25	21.5	12.4
Czech Rep.	20.0	-	-	4-21	19.8	12.4
Denmark	8.4	8	-	2-23	22.8	9.6
Estonia	12.9	-	-	1-30	18.1	12.4
Finland	-	3-29	-	3-26	12.3	10.1
France	16.5	-	-	1-29	16.6	12.2
Germany	12.1	3-28	12-35	-	18.4	14.8
Greece	16.0	-	-	-	16.1	11.5
Hungary	8.1	-	-	3-27	26.9	8.9
Ireland	8.5	-	9-35	3-24	15.2	12.4
Italy	11.8	13-28	-	4-26	20.0	11.5
Latvia	10.0	-	-	20	24.4	10.0
Lithuania	12.3	-	5-43	12	17.5	12.4
Luxembourg	11.9	-	-	10	-	9.9
Malta	-	-	-	3-28	24.1	12.4
Netherlands	8.9	12-30	9-22	13-31	15.8	9.2
Poland	13.6	-	-	3-15	16.6	11.1
Portugal	9.5	7-42	-	7-42	19.9	9.1
Romania	20.0	-	-	20	18.8	12.4
Slovakia	15.8	-	-	3-22	18.0	12.4
Slovenia	11.9	-	-	14-36	15.0	11.9
Spain	9.4	7-22	10-24	2-23	17.5	9.4
Sweden	9.1	11-34	11	2-27	21.3	11.0
UK	10.6	-	-	2-25	17.6	12.4

[a]Data: from common reporting format (CRF) of greenhouse gas inventory submission of 2011.
[b]Range in N excretion factors indicate that more categories of pigs are included.

Table 2-6. Gross nitrogen excretion factors (kg N head^{-1} yr^{-1}) for poultry (NIR 2011, and CAPRI), laying hens (UNECE, Nitrates Directive, OECD/Eurostat, GAINS) and broilers (UNECE, Nitrates Directive, OECD/Eurostat).

	NIR (2011)[a]	UNECE (2007-2010)		Nitrates Directive (2011)		OECD/Eurostat (2011)		CAPRI (2004)	GAINS (2010)
	Poultry	Laying hens	Broilers	Laying hens	Broilers	Laying hens	Broilers	Poultry	Laying hens
Austria	0.55	0.52	-	-	-	0.72	0.28	0.49	0.73
Belgium	0.58	0.74	-	-	-	-	0.55	0.42	0.70
Walloon	-	-	-	-	-	-	0.54	-	-
Flanders	-	-	-	-	-	-	0.60	-	-
Bulgaria	0.60	-	-	-	-	-	-	0.68	0.80
Cyprus	-	-	-	-	-	0.63	0.32	0.58	0.80
Czech Rep.	0.60	-	-	-	-	0.60	0.35	0.56	0.80
Denmark	0.53	0.79	-	-	-	1.11	0.63	0.84	0.71
Estonia	0.60	-	0.40	-	-	0.78	0.23	0.58	0.80
Finland	0.58	0.67	-	-	-	0.64	0.48	0.43	0.80
France	0.60	-	0.55	-	-	-	-	0.61	0.80
Germany	0.78	0.84	-	0.75-0.79	0.31-0.47	0.78	0.37	0.52	0.84
Greece	0.60	-	-	-	-	-	-	0.52	0.80
Hungary	0.60	-	-	-	-	0.74	0.38	0.69	1.50
Ireland	0.31	-	-	0.56	0.24	-	0.60	0.47	0.84
Italy	0.53	-	-	-	-	0.66	0.38	0.47	0.66
Latvia	0.60	-	-	-	-	-	-	0.83	0.90
Lithuania	0.60	-	-	0.8-0.87	0.12	-	-	0.61	0.80
Luxembourg	0.74	-	-	-	-	-	-	-	0.80
Malta	-	-	0.50	-	-	0.57	0.04	0.62	0.80
Netherlands	0.65	-	-	0.37	0.36	-	0.53	0.49	0.67
Poland	0.35	-	0.45	-	-	0.70	0.14	0.58	0.70
Portugal	0.56	0.80	-	-	-	0.80	0.45	0.64	0.60
Romania	0.60	-	-	-	-	-	-	0.58	0.78
Slovakia	0.73	-	-	-	-	0.70	0.30	0.62	0.80
Slovenia	0.60	-	0.43	-	-	0.71	0.40	0.43	0.71
Spain	0.45	0.49	0.29	0.80	0.60	0.49	0.42	0.56	0.80
Sweden	0.40	0.64	-	0.60	0.28	0.73	0.28	0.73	0.64
UK	0.57	-	-	-	-	1.89	0.51	0.58	0.85

[a]Data: from common reporting format (CRF) of greenhouse gas inventory submission of 2011.

Table 2-7. Gross nitrogen excretion factors (kg N head^{-1} yr^{-1}) for sheep and goats in EU-27.

Country	NIR (2011)[a]		UNECE (2007-2010)		Nitrates Directive (2010)		OECD/ Eurostat (2011)		CAPRI (2004)	GAINS (2010)
	Sheep	Goats	Sheep	Goats	Sheep	Goats	Sheep	Goats	Sheep and goats	Sheep and goats
Austria	13.1	12.3	13.1	12.3	-	-	-	13.1	5.2	13.0
Belgium	7.5	8.4	8.8-10.5	8.8-10.5	-	-	8.9	8.3	5.5	7.4
Walloon	-	-	-	-	-	-	10.2	9.2	-	-
Flanders	-	-	-	-	-	-	7.4	7.1	-	-
Bulgaria	14.7	17.0	-	-	-	-	14.7	17.0	9.5	12.0
Cyprus	-	-	-	-	-	-	9.5	11.2	9.2	12.0
Czech Rep.	20.0	25.0	-	-	-	-	9.8	9.8	4.7	12.0
Denmark	15.3	16.4	17.0	16.3	-	-	-	-	8.8	17.0
Estonia	16.0	25.0	-	-	-	-	14.0	14.0	6.5	14.0
Finland	10.0	10.7	10.0	10.7	-	-	10.0	10.7	4.0	16.0
France	18.3	25.0	-	-	-	-	18.3	25.0	7.7	12.0
Germany	7.4	11.0	7.8	11.0	18.1-18.6	14.8	-	14.8	5	7.5
Greece	10.7	12.0	-	-	-	-	-	-	7.9	12.0
Hungary	20.0	18.0	-	-	-	-	-	14.6	7.9	12.0
Ireland	6.3	9.0	-	-	7-13	9	10.6	12.9	5.1	8.0
Italy	16.2	16.2	16.2	16.2	-	-	12.8	13.5	6.2	16.2
Latvia	13.0	13.0	-	-	-	-	6.0	6.0	10.8	7.0
Lithuania	16.0	16.0	-	-	12	10-12	16.0	16.0	6.7	12.0
Luxembourg	17.0	17.0	-	-	-	-	17.0	17.0	-	12.0
Malta	-	-	-	-	-	-	9.1	-	8.3	12.0
Netherlands	6.7	9.9	14.1	17.5	7.4 - 10.2	3.1-5.8	-	-	4.8	11.5
Poland	6.8	6.7	-	-	-	-	8.0	7.0	6.2	13.7
Portugal	7.1	6.0	6.6	-	-	-	-	-	8.4	7.0
Romania	16.0	25.0	-	-	-	-	16.0	25.0	7.8	5.2
Slovakia	16.0	16.0	-	-	-	-	-	10.0	6.9	12.0
Slovenia	20.0	25.0	-	-	-	-	20.0	-	5.0	11.3
Spain	5.2	11.3	5.1	11.3	10.0	8.8	6.6	9.0	6.8	12.0
Sweden	6.1	8.7	13.0	-	14.0	-	13.0	11.3	8.2	6.1
UK	5.2	20.6	-	-	-	-	-	-	-	6.4

[a]Data: from common reporting format (CRF) of greenhouse gas inventory submission of 2011.

2.4 Need for harmonization of methods and data

Our comparison of the excretion factors between policy reports and between countries shows large differences. The differences in N excretion factors between countries result from differences in animal productivity and animal husbandry practices, but also from differences in animal categorization and aggregation, calculation methods, year of reporting or data collection, data and information collection/processing/reporting procedures, and adjustments for the actual length of the production cycle. These adjustments are necessary to allow for non-productive time needed for cleaning and re-stocking the housings. A difference in the calculated N excretion between member states due to methodology is unacceptable for policies and calculation of total excretion and emissions in EU, and therefore there is a need for harmonization of methodologies.

The N excretion factor has a large effect on the calculated NH_3 and N_2O emissions, and N surplus. The emission factors for NH_3 and N_2O are expressed in percent of N excreted or N applied to soil (Hyde *et al.*, 2003; Dämmgen *et al.*, 2006; IPCC 2006; Gac *et al.*, 2007; Hutchings *et al.*, 2011; Oenema *et al.*, 2011; Velthof *et al.*, 2012; EEA, 2013). Evidently, variations in N excretion factors affect the calculated emissions directly. Thus, large variations in N excretion factors between member states have a large effect on the reported emissions. Emission reduction targets for NH_3 and N_2O are set by UNFCCC, UNECE and the European Commission. Manure application standards set by the European Commission have less meaning if N excretion factors have low accuracy. Evidently, there is a clear need for a robust, common, harmonized approach for estimating N excretion factors, so that the emissions reported by member states have a common and transparent basis and can be used for estimates of total emission in the EU.

Any common, universal approach must account for the differences between countries in (i) the importance of livestock production, and hence in the relative magnitude of N and P excretion as a source of N and P, (ii) the type of livestock production systems (animal species, animal housing, animal feeding), and (iii) in the data and information collection infrastructure. This holds especially also for the EU, where livestock density may range from an average of less than 0.5 livestock units (LSU) per ha to more than 3 LSU per ha. Also, some countries have relatively large populations of cattle (dairy and/or beef), while other countries have relatively more pigs or poultry or sheep or goat. Countries with a high livestock density commonly have developed a more detailed infrastructure for data and information collection than countries with a low livestock density.

The first step would be the definition of animal categories for which excretion figures have to be calculated. These are preferably categories for which animals number are gathered, so that total manure production in regions and countries can be made. For the EU, it is recommended to use animal categories in the Farm Structure Survey (European Commission, 2009) as a basis. It is also recommended to harmonize the livestock categories in the different policy reporting streams or develop procedures to aggregate or disaggregate livestock categories for the N excretion calculation. This is especially important for other cattle, pigs, and poultry, as these categories include livestock types with a large difference in N excretion.

The second step would be to recommend a set of methodologies with different levels of detail (i.e. a Tiered approach) to estimate the N excretion for each category. The accuracy of the balance method to calculate excretion is higher than that of estimation on basis of measured manure composition. The variability of N contents in manure is large (Table 2-1) and accuracy of weighing or measurement of volumes of manure on animal or farm scale is often low. The uncertainties in data needed for the balance are generally lower than those of measurement of manure composition and amount of manure. Therefore, the basis for the calculation of N excretion factors should be the balance method, i.e., N excretion = feed N intake – N retention in the animal and animal products. The Tier 1 approach is the most simple approach and would be an approach with default N excretion figures for certain regions or farming systems (depending on intensity). In other Tier levels, harmonized methodologies to calculate N excretion data are needed, which use available information for productivity and inputs on regional to national scales and are updated every 1 -5 years.

The third step is the harmonization of collection and processing of data, such as feed intake per animal category, feed production (e.g. grassland yields) and composition (e.g. protein content of grassland), animal production (e.g. production of meat, milk, and eggs), and the composition of animal products. For accurate estimates of N excretion of dairy cows, there is a need for accurate estimates of grassland yields and N uptake. Estimates of grassland yields can be based on empirical data (field experiments), results of crops models, expert estimates, and feed balances of dairy cattle (i.e. the feed N intake can be estimated from the milk yield).

Common methodologies for N excretion calculation and data collection would allow for a harmonized and transparent estimation of actual N excretion, and hence for a common basis for the estimation and sound comparison of manure N, N balances, and NH_3 and N_2O emissions in different member states and EU policies.

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CHAPTER 3

Feed use and nitrogen excretion of livestock in EU-27

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Abstract

Livestock excreta is a large source of nitrogen (N) in the European Union (EU), used to fertilize crops, and also a main source of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3^-) losses to the environment. The amount of N in excreta mainly depends on the animal category and productivity, and on feed use and management. National inventories of emissions to the environment are often based on different methodologies for the estimation of N excretion. Here, we present a transparent and uniform methodology for estimating annual feed use and N excretion per animal category for all countries of the EU-27, based on the energy and protein requirements of the animals and statistics of feed use and composition, animal number and productivity.

The calculated total feed use in the EU-27 was 506 Tg dry mass in 2010. Dairy cows used 29%, other cattle 34%, pigs 17%, chicken 9%, sheep and goats 8%, and other animal categories 3% of the total feed use. Grass and annual forages were mainly used by dairy cows (30 and 49%, respectively) and other cattle (55 and 44%); pigs used most of the feed cereals (53%); protein-rich feed (e.g., soybean meal) were mostly used by pigs (34%) and chicken (24%). Differences between countries in feed use were large, mainly related to variations in national feed supply and animal productivity. Total N excretion of the animals amounted to 9.7 Tg in 2010, and varied between countries from 14 to 291 kg ha^{-1} of utilized agricultural land. The present study provides a uniform and transparent approach for evaluating feed use and N excretion in all countries of the EU-27. Our results underline the significant differences in N excretions between EU countries as a result of feed use variations, suggesting the need for basing N excretion estimations on feed use data. The dataset present in this study may serve as a basis for such efforts, also to improve national inventories of N emissions.

3.1 Introduction

The livestock sector is a key user of natural resources, including land, water, nutrients and biomass. Recent estimates suggest that 4.7 to 7.0 billion tonnes dry biomass is used by livestock, equivalent to nearly 60% of the global plant biomass use (Krausmann *et al.*, 2008; Wirsenius *et al.*, 2010; Herrero *et al.*, 2013). A similar estimate (60-65%) is reported for Europe (Krausmann *et al.*, 2008). The livestock sector also contributes approximately 40% to the global anthropogenic ammonia (NH_3) and nitrous oxide (N_2O) emissions (Galloway *et al.*, 2004; Oenema *et al.*, 2005). In Europe, livestock contributes as much as 80% to the total NH_3 emissions (EEA, 2014), and about 40% to the total N_2O emissions (Bellarby et al., 2013; Oenema et al., 2014). In addition, over use of livestock manure results in leaching of nitrates to groundwater and surface water in Europe (Velthof *et al.*, 2014).

Animal production is projected to continue growing in the next decades, driven by human population growth, rising incomes and dietary preferences towards 'western' diets (Steinfeld *et al.*, 2010; Thornton, 2010). The expansion of animal production, feed use and associated environmental impacts will increase the pressures on natural ecosystems further, unless large improvements are being made in animal productivity, manure handling and manure nutrient recycling (Wirsenius *et al.*, 2010; Kastner *et al.*, 2012; Tilman & Clark, 2014). Strategies such as improvements in feed quality and management, low-emission animal housing and manure management, and timing and rate of N application can greatly abate the growing pressure on the environment (Thornton & Herrero, 2010; Wirsenius *et al.*, 2010). To that end, knowledge and quantitative information on feed use and nutrient excretion rates of the animals, depending on regional resource availability, is crucial for the development of sustainable agro-ecosystems.

Feed composition and animal productivity have significant influences on nitrogen (N) excretion, and on N emissions downstream in the manure management chain (Olesen *et al.*, 2006; Oenema *et al.*, 2009; Hou *et al.*, 2015). International and national statistics (e.g., Eurostat and FAO statistics) provide national data on animal production and animal number per animal category annually. However, animal category-specific data on feed use and composition and on nutrient excreta are usually not available at regional or national levels, and therefore have to be collected or estimated. Various approaches are being used to estimate feed use and N excretion. The N balance approach covers both aspects of feed intake and animal production, i.e., the N excretion is equal to the total amount of feed N consumed minus the N retained in animal products (e.g., milk, eggs and live-weight gains). This approach has been widely employed in field and farm scale research, and benefits from timely measured feed composition and production performance (Arriaga et al., 2010; Galassi et al., 2010; O'Connell et al., 2006; Philippe et al., 2012, 2009). Further, efforts have been made to scale up this N balance approach to regional and national scales, based on estimates of regional and category specific feed use, to support national inventories of N emissions (Webb, 2001; Velthof *et al.*, 2012; Bai *et al.*, 2014). Most of these national-level studies have focused on a single animal category. However, only national studies that include all animal categories would allow to check the feed balance; do the total supplies of feed resources in a country indeed match with the sum of the estimated national feed use by all animal categories, within acceptable ranges of uncertainty?

The overall objective of our study was to provide a uniform approach for the estimation of feed use and N excretion rates of the animals in EU-27 through linking statistical data on feed quality and quantity with energy and protein requirements per animal category at country levels. Firstly, we developed a uniform method for the estimation of animal category-specific average feed use per country, as function of animal productivity and feed availability. Secondly, we estimated the N excretion rates of individual animal categories using the N balance approach. Thirdly, sensitivity analyses were carried out to get quantitative insight in the effect of changes of several methodological parameters on the feed use and N excretion. We then discussed regional variations in feed use and N excretion, and compared our results with national inventories, and also discussed the implication of our study.

3.2 Materials and methods

3.2.1 Concept

The methodology developed here aims at deriving animal category-specific and country-specific N excretion coefficients through linking statistical data and information on the availability (quantity and quality) of feed with animal numbers, and the energy and protein requirements of the animals. Animal categories included dairy cows, other cattle, sheep and goats, pigs, laying hens, broilers, turkey and other poultry, i.e., similar to the main categories in the Farm Structure Survey (FFS), used by all countries of the EU-27. The methodology developed in this study is described schematically in Figure 3-1. We started by estimating the feed energy required for each animal category per country, considering animal stock and animal production (e.g., milk, eggs and meat). Next, national feed supply and compositions were derived for eight aggregated feed classes: (i) animal and fish derived feed, (ii) protein-rich feed (e.g., soybean meal), (iii) cereal (grain or processed) feed, (iv) brans, (v) oil and sugar crops, (vi) other non-roughage feed (e.g., root crops, and residues of fruits and vegetables), (vii) annual forages (e.g., maize silage, leguminous crops, temporary grass, and crop straw) and (viii) perennial forages (grass harvested by grazing and grass harvested for silage and hay). We then partitioned each feed aggregate over animal categories using a linear optimization approach, by respecting a series of category-specific numerical constraints related to energy and protein requirements, with the objective of minimizing the difference between the total feed biomass supply and the total feed biomass requirements per country. Finally, we quantified N excretion as the difference between the feed N intake (i.e., N in animal diet) and the N retained in animal products for each animal category on the national scale. We based our data on three-year averages of statistical input data (e.g., animal stock, production, and feed biomass supply and production) for the period 2009-2011. The calculation procedures are described in more detail in the following sections.

3.2.2 Calculation of energy requirements

The total annual energy requirement of each ruminant category (dairy cows, other cattle, sheep and goats) was estimated per country according to the Tier 2 approach of IPCC (2006). The method described by Wirsenius (2000) was adapted here to calculate the energy requirements of the mono-gastric animal categories. Energy requirements were calculated for animal maintenance, growth, lactation, pregnancy and activity, as a function of the average live weight of the animals, animal production and management conditions (i.e., raised in houses or pasture) (IPCC, 2006; Wirsenius, 2000). Energy requirements for maintenance were based on the number of animals in the statistics and the calculated requirement per animal on a national level. We assumed that the number of animals remained constant throughout the year. Energy requirements for growth, lactation and pregnancy were related to animal production statistical data. Information about pasturing periods of ruminants was derived from national inventory reports to UNFCCC (the United Nations Framework Convention on Climate Change) and used for calculation of energy requirements for activity (IPCC 2006). More details about the calculations of energy requirements can be found in the supplementary information. Animal numbers and production data were derived from Eurostat (2014). Energy required by ruminants was calculated on a digestible energy (DE) basis. For poultry and pigs, the calculated energy requirements are expressed as metabolizable energy (ME) and digestible energy (DE), respectively.

All calculations were performed with GAMS programming software (http://www.gams.com/). Linear optimization was performed using the 'LP' solver in GAMS.

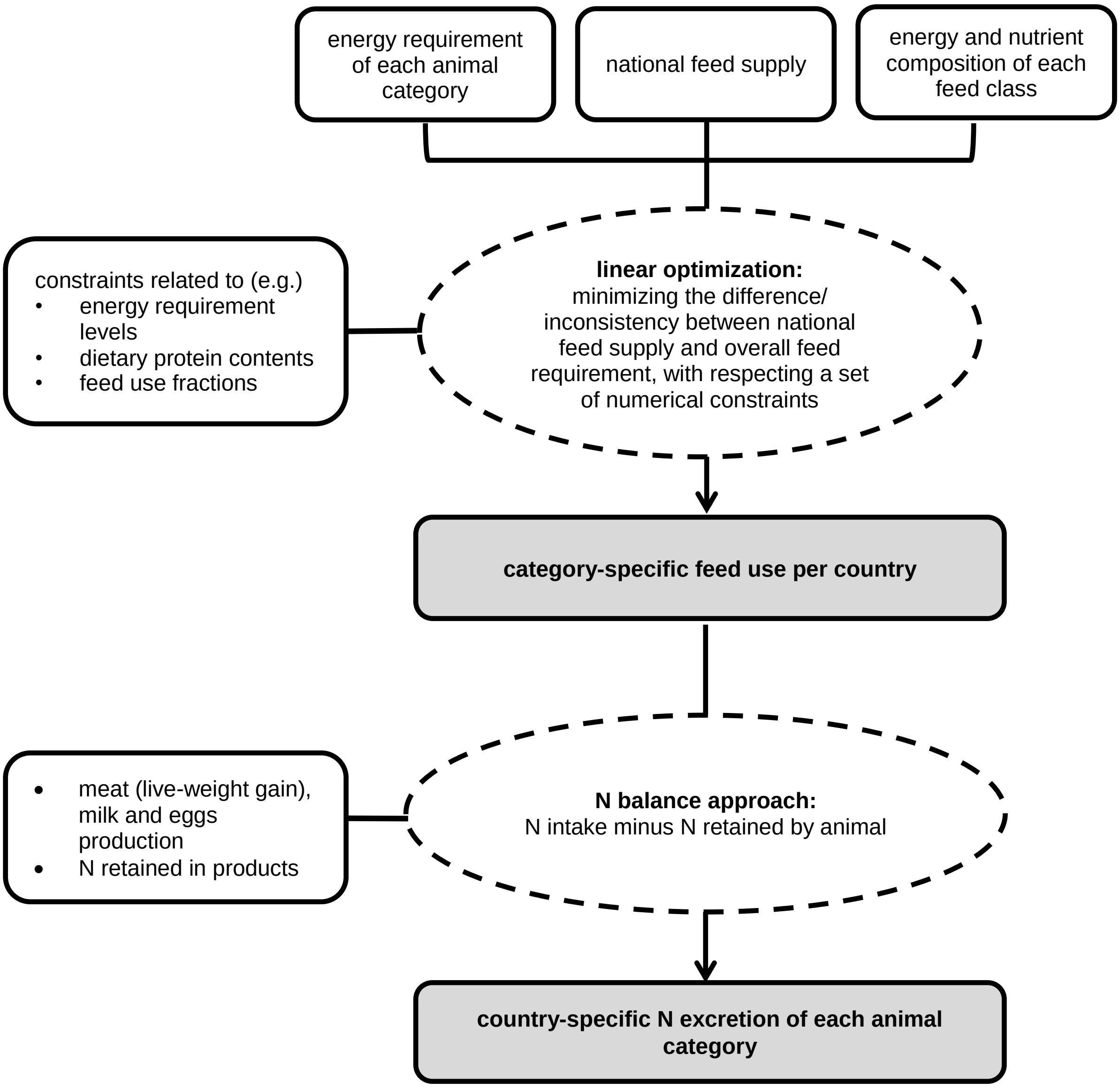

Figure 3-1. A simplified schematic representation of the information flow in calculating country-specific feed use and nitrogen (N) excretion of each animal category. The arrows depict the information flow direction. Information as data and parameter input to the model are sketched in top and left boxes. Results are indicated in the shaded boxes, which are gained through the calculations indicated in dashed circles.

3.2.3 Calculation of feed supply

Data on the national supply of feed resources (except for grass, forages and crop residues) were extracted from FAO commodity balance sheets; data were corrected for export and import (FAOSTAT, 2014). The fresh or air-dry weights in the FAOSTAT database were corrected for moisture content to obtain a uniform dry matter (DM) weight of each feed item. The use of straw as feedstuff was based on domestic cereal production (FAOSTAT), the mean straw/grain ratio (Krausmann *et al.*, 2008) and the proportion of crop straw recovered as feed (Krausmann *et al.*, 2008). The supplies of grass and annual forages were estimated from the land areas of grassland and forages (Eurostat, 2014), multiplied by the regional productivity data of forages (Eurostat, 2014) and grass (Smit *et al.*, 2008).

The specific feed resources present in FAO commodity balance sheets and the grass, forages and straw were aggregated into 20 groups. Parameters related to protein and energy contents were given for each feed group (Table S3 and Table S4). Protein and energy contents were obtained from NRC (2001, 2000, 1998, 1994). The supply of feed protein and energy were estimated for each feed group and each country. To improve the optimization procedure (see Section 3.2.4) and the presentation of results, we further allocated these 20 feed groups into eight main classes (animal and fish derived feed, protein-rich feed, cereal feed, brans, oil-and-sugar crops, other non-roughage feed, annual forages and perennial forages). The classification of feedstuffs is further detailed in the supplementary information.

3.2.4 Feed allocation

The allocation of feed classes over animal categories per country in EU-27 was performed by an optimization procedure and under a set of constraints, dependent on the protein and energy requirements per animal category per country and the availability of national feed resources. The final objective of the optimization is to minimize the difference between the total national feed biomass supply (see Section 3.2.3) and the calculated total feed biomass use by all animal categories in a country. The feed biomass use (feed ration) of the animal was determined endogenously by the feed optimization model, and therefore considered an outcome of the model calculations. The principle behind this minimization objective is that the total national feed supply has to match with the total feed use, within acceptable ranges of uncertainty. We assumed that the feed biomass supply can be equal to or more than the calculated feed biomass use. Four constraints were considered, as follows (see also supplementary information):

1. We estimated the energy requirement per animal category for ‘average’ conditions in a country (e.g., average climate, animal genetics and animal performance; Section 3.2.2). To allow for some variations, we assumed that the ‘actual’ energy requirement may vary from 90 to 110% of the requirement for ‘average’ conditions. Therefore, the feed energy use per animal category (endogenous variable of the model) was considered equal to the calculated energy requirement for ‘average conditions’ within a range of ± 10%.
2. The average crude protein (CP) contents of the animal diets were constrained by a set of category-specific ranges derived from literature (e.g., Bittman et al., 2014) (see Table 3-1). Please note that the ranges of dietary CP contents were the estimates for national average conditions; there may be animal farms in practice where the CP contents in animal diets may be higher (or lower) than the upper (or lower) value of the range, but these farms are not representative of national averages.
3. Roughage (grass, forages and crop residues) were allocated to ruminants, cereals and protein-rich feeds were offered mainly to poultry and pigs. Table 3-1 summarizes these constraints, which were defined according to literature (Bouwman *et al.*, 2005; Lesschen *et al.*, 2011; CBS, 2012) and experts’ opinions.
4. We assumed that all animals in one country use at least 85% of the supply of high-quality feed classes, such as protein-rich feed and feed cereals, and at least 70% of the total supply of animal and fish derived feed, brans, root and sugar crops and grass from managed grassland. The minimum percentages of feed use were set relatively low for annual forages (at least 40% of the supply), other non-roughage feeds (e.g., root crops, residues of fruits and vegetables) in minute supply (20%), natural grass (10%) and crop residues (10%). These assumptions are set up mainly for the sake of proper methodological performances and uncertainty concerns (discussed in detail in Section 3.4.2). For instance, by setting the minimum use of major feed classes high, we guarantee that animals make full use of these high-quality feeds as much as possible. Feedstuffs from crop residues are known to be insignificant in animal diets (in most EU countries), thus low priority for their use were assumed by using a low minimum percentage. In addition, uncertainties in the estimated supply of annual forages, crop residues and natural grasses are relatively large, so that low minima were

assigned. It should be noted that the calculated feed use of these low-quality (and minute-supply) feed classes may be large as percentage of the feed supply, when the high-quality feed classes in a country are hardly sufficient to meet the total feed requirements of the animals.

3.2.5 Calculation of nitrogen excretion

Nitrogen excretion of each animal category at the national level was quantified as the difference between the total feed N intake and the total N retained by animal products (milk, eggs and/or live-weight gains). Feed N intake per animal category was quantified based on individual feed DM intake per animal category and N content of the respective feedstuff. The N retained in animal products per animal category was calculated according to the total production of animal products per animal category and N contents of the products. Nitrogen content is expressed in % N of DM in the case of feed and in % of fresh weight (as-is) in the case of animal products. Nitrogen and crude protein contents were assumed to be related by a constant protein-to-nitrogen coefficient. Country-specific protein contents of dairy milk were derived from Eurostat (three-year averages over 2009-2011), then converted to N contents using a constant milk protein-to-nitrogen conversion coefficient. Protein contents in other animal products are not recorded by Eurostat; therefore, N contents (% N, on as-is basis) were derived from literature and were uniformly applied to all countries: i.e., 0.5% for goat and sheep milk, 1.85% for eggs, 2.5% for live-weight gains of sheep, goat and pigs, 3.0 % for live-weight gains of chicken, 3.1% for turkey and 2.6% for other cattle and other poultry (CBS, 2012).

Table 3-1. Constraints added to the linear equations of feed partitioning, applied uniformly to all EU-27 countries (unless otherwise noted).

Animal categories	Ranges	Constraints of crude protein content in diet (%, in weight)	Constraints related to the fraction of each feed class in diet (on dry matter basis)								
			Animal and fish based feed	Protein-rich feed	Cereals[a]	Brans	Oil and sugar crops	Others	Grass[a]	Annual forages	
										Maize and other forages	crop straw
Dairy cows	Min.	13	0	0.05	0.05	0	0	0	0.20	0.05	0
	Max.	19	0.10	0.10	0.40	0.10	0.25	0.25	0.95	0.60	0.10
Other cattle	Min.	13	0	0	0	0	0	0	0.20	0.05	0
	Max.	19	0.10	0.10	0.40	0.10	0.25	0.25	0.95	0.60	0.10
Sheep	Min.	14	0	0	0	0	0	0	0.20	0.05	0
	Max.	19	0.10	0.05	0.15	0.10	0.25	0.25	0.95	0.60	0.10
Goats	Min.	14	0	0	0	0	0	0	0.20	0.05	0
	Max.	20	0.10	0.05	0.30	0.10	0.25	0.25	0.95	0.60	0.10
Pigs	Min.	14	0	0.05	0	0	0	0	-	-	-
	Max.	18	0.10	0.40	0.85	0.30	0.15	0.15	-	-	-
Laying hens	Min.	15	0	0.15	0	0	0	0	-	-	-
	Max.	18	0.10	0.55	0.80	0.25	0.15	0.15	-	-	-
Broilers	Min.	17	0	0.20	0	0	0	0	-	-	-
	Max.	20	0.10	0.55	0.80	0.25	0.15	0.15	-	-	-
Turkey	Min.	14	0	0.20	0	0	0	0	-	-	-
	Max.	21	0.10	0.55	0.80	0.25	0.15	0.15	-	-	-
Other poultry (duck and geese)	Min.	14	0	0.15	0	0	0	0	-	-	-
	Max.	20	0.10	0.55	0.80	0.25	0.15	0.15	-	-	-

- Not applicable.

[a] The maximum constraints of cereal product fractions for ruminants were adjusted into 0.4 for Cyprus and Malta, and the minimum constraints of grass fractions were 0.1 for Cyprus, Malta and Finland; these adjustment are made considering the relatively small fraction of grass supply in these countries.

3.2.6 Definition of assessment indicators

Feed conversion was defined as the amount of dry weight feed needed to produce 1 kg of animal product. Feed conversion ratio (FCR) was calculated by dividing the overall feed dry mass intake by the total production of animal products per animal category per country. Further, feed energy to edible energy convention ratio (FER), i.e., the feed energy intake needed to yield 1 MJ edible energy contained in animal products, was quantified. Edible energy contents in animal products were derived from USDA (2011). The feed-to-animal protein conversion ratio (FPR), i.e., the feed dry mass needed to yield 1 kg protein in animal product, was also calculated. Nitrogen use efficiency (NUE, %) at animal level was quantified in terms of total N retained in animal production as percentage of total feed N intake. In addition, the mean feed DM (kg DM stock^{-1} yr^{-1}), energy (MJ stock^{-1} yr^{-1}) and N intake (kg N stock^{-1} yr^{-1}) per animal stock, and the N retained and N excretion (N excretion coefficient) per animal stock were calculated. The number of animals (stock) were derived from Eurostat (three-year averages were used, namely 2009-2011). Data were checked for consistency through comparing the number and mean weight of slaughtered animals with the mean stock; we observed for some countries (e.g., Slovenia) unusually high values for the ratio of slaughtered broilers to the mean stock of broilers. Stock data were corrected in such cases by using the median ratio of slaughtered number to stock in the EU-27.

3.2.7 Sensitivity analyses

The impacts of changes of input parameters on total feed use and N excretion per country were assessed, so as to identify the sensitivity of assumptions and input parameters:

- The N concentration of feed may vary due to the variations in, e.g., fertilization, crop varieties and climate conditions. The effect of these possible variations on the total feed use and N excretion were assessed for major feed classes, namely, protein-rich feed (± 5%, relative to default N content), cereals (± 10%) and grass (± 10%). According to the literature (Feedipedia, 2014; NRC, 2001), generally regional variations in protein contents of grass and cereals were larger than that of protein-rich feed (e.g., soybean and soybean meal, and other oil cakes).
- Effects of possible variations in feed energy content on feed use and the amount of N in excreta per country were estimated by assuming a 5% difference in energy content, compared to the default energy content, for cereal feed and grass. This assumption for

the changes (± 5%) in energy content was based on literature information (Feedipedia, 2014; NRC, 2001) and experts' opinions.

- Effects of possible variations in animal production and animal numbers on total national feed use and N excretion were also examined: i) increases in animal production by 5%, with animal numbers being unchanged (i.e., increasing animal productivity); ii) increases in both animal production and animal numbers by 5% (i.e., expanding animal husbandry with the default animal productivity level).
- Effects of changes (±5%) in average live weight of animals, relative to the default value, on total feed use and N excretion per country were analysed.

3.3 Results

3.3.1 Feed supply and use

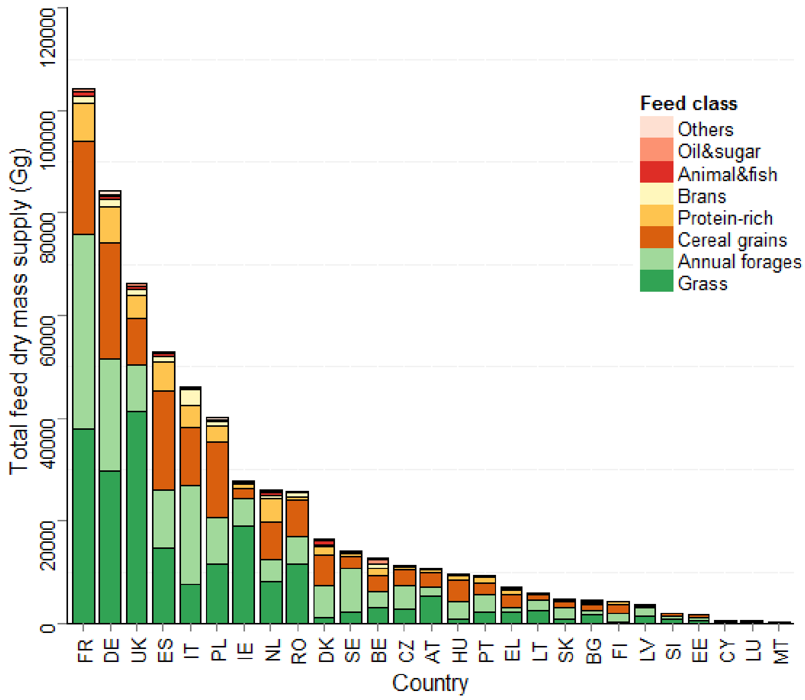

Figure 3-2. The total supply of dry-weight mass per feed class per country of the EU-27, annual averages of 2009–2011 (Gg=10^9 g). Country abbreviations are explicated in Table 3-2.

Figure 3-2 shows the estimated three-year averaged (2009-2011) feed supply per feed class per country in the EU-27. The total feed supply in the EU-27 amounted to 593 Tg (1 Tg = 10^{12} g). Feed supply ranged from < 1 Tg yr^{-1} in Cyprus, Luxembourg and Malta to >50 Tg yr^{-1} in France, Germany, Spain and the UK. Grass, annual forages (e.g., silage maize, leguminous crops) and cereals were the largest feed classes, accounting for 33, 28 and 25% of the total feed use in the EU-27, respectively. However, there were large differences between countries (Figure 3-2). The calculated total feed use was 506 Tg, or 85% of the calculated feed supply in the EU-27; the use of cereal feed and protein-rich feed was on average 90% of their supply totals (see Table S5). Note that effects of feed wastes and conservation losses were not included in the feed use estimations due to the lack of reliable quantitative information.

The annual average feed DM use per animal category in EU-27 was 6252 kg for dairy cows, 2620 kg for other cattle, 571 kg for pigs, 359 kg for sheep and goats, 35 kg for laying hens, and 38 kg for broilers. There were large variations between countries, both in the feed use per animal and the feed composition (Figure 3-3). The average ration of dairy cows comprised 34% perennial forages, 48% annual forages and 17% cereals and protein-rich feed; other cattle used 54, 37 and 8% of the aforementioned feed classes, respectively. Grass was the main ingredient for cattle in countries with large areas of grassland, e.g., Germany, Netherlands, Ireland and the UK (Figure 3-3). Cereals, protein-rich feed and brans were the main feed sources for pigs (on average 78, 17 and 4% of the ration, respectively), laying hens (72, 19 and 6%) and broilers (47, 24 and 18%).

Dairy cows used 29% of the total feed use in the EU-27, other cattle used 34%, pigs 17% and chicken 9% (Table S6). Pigs used 53%, chicken 21%, and dairy cows 14% of the cereal feed in EU. Protein-rich feed was used for 34% by pigs, 24% by chicken, 18% by dairy cows, and for 14% by other cattle (Table S7). The feed use per animal category per country is shown in Figure 3-4. Dairy cows and other cattle had the largest share in the national total feed use, except for Denmark and Spain, where pigs had the largest share.

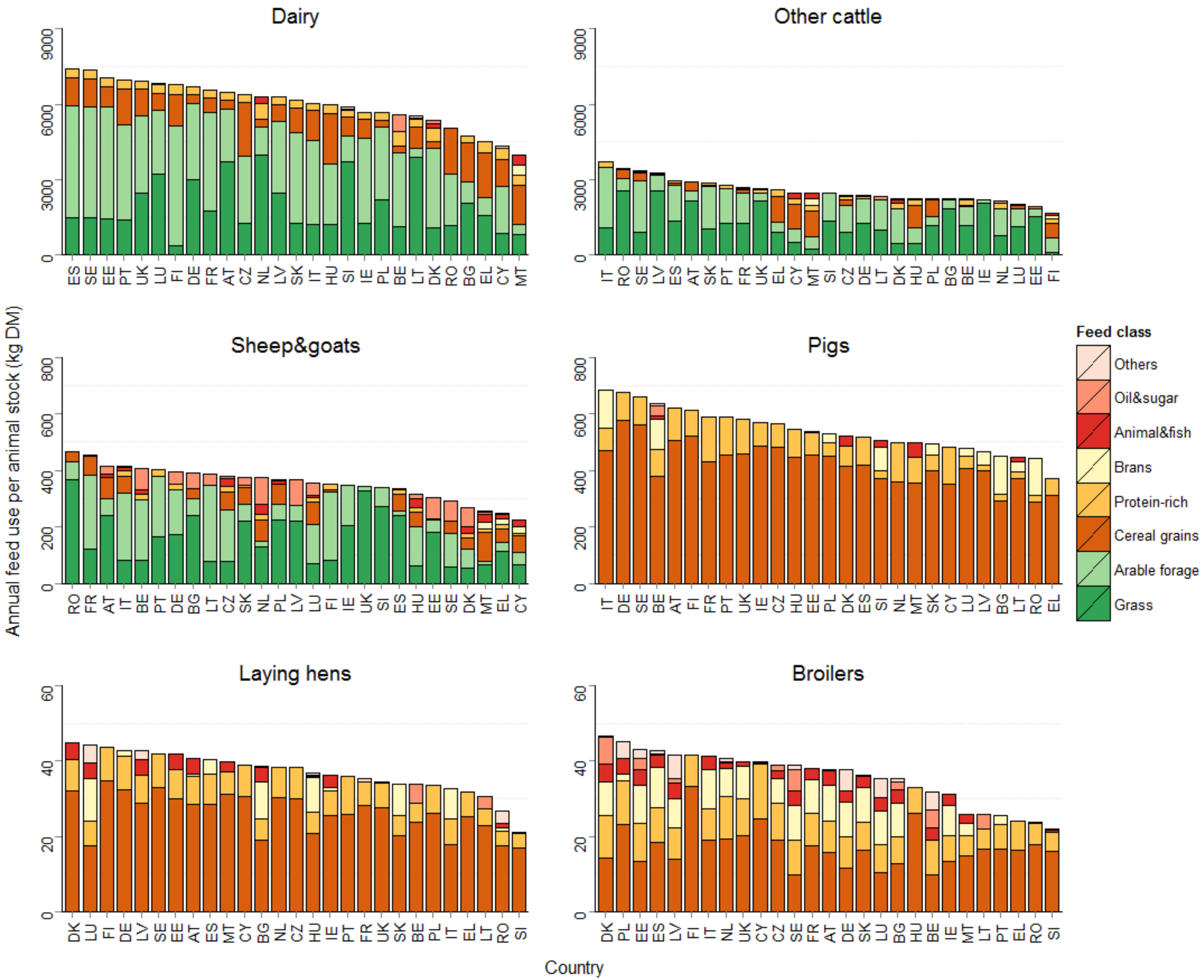

Figure 3-3. Annual average feed dry matter (DM) use per animal stock of the main livestock categories in the EU-27. Notice that member states on X-axis are presented in decreasing order. Country abbreviations are explicated in Table 3-2.

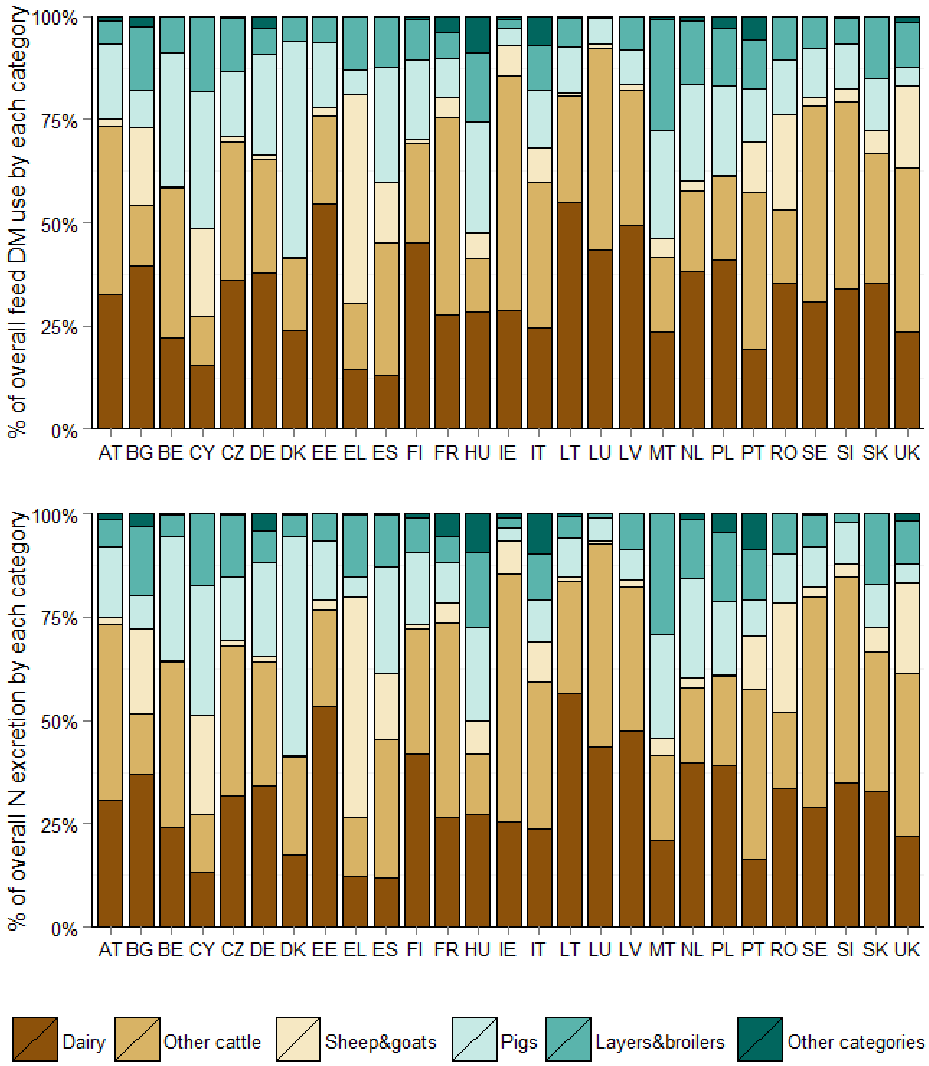

Figure 3-4. Percentage of national total feed dry matter (DM) use by animal category and percentage of total nitrogen (N) excretion by animal category for each country in the EU- 27. Country abbreviations are explicated in Table 3-2.

3.3.2 Feed nitrogen intake

Total feed N intake of all animal categories in EU-27 was estimated at 12.2 Tg for the three-year average of 2009-2011. The calculated feed N intake per head for each animal category and each country is shown in Figure 3-5. The annual average N intake in kg per head was 144 kg for dairy cows (range 94-185 kg), 58 kg for other cattle (42-84 kg), 15 kg for pigs (9-17 kg), 8 kg for sheep and goats (5-15 kg), 1.0 kg for laying hens (0.5-1.3 kg) and 1.2 kg for broilers (0.6-1.5 kg) in the EU-27. For ruminants, protein-N intake mainly originated from perennial (grasses) and annual forages (Figure 3-5). Protein-rich feed and cereals were the main protein-N sources of pigs, laying hens and broilers (Figure 3-5).

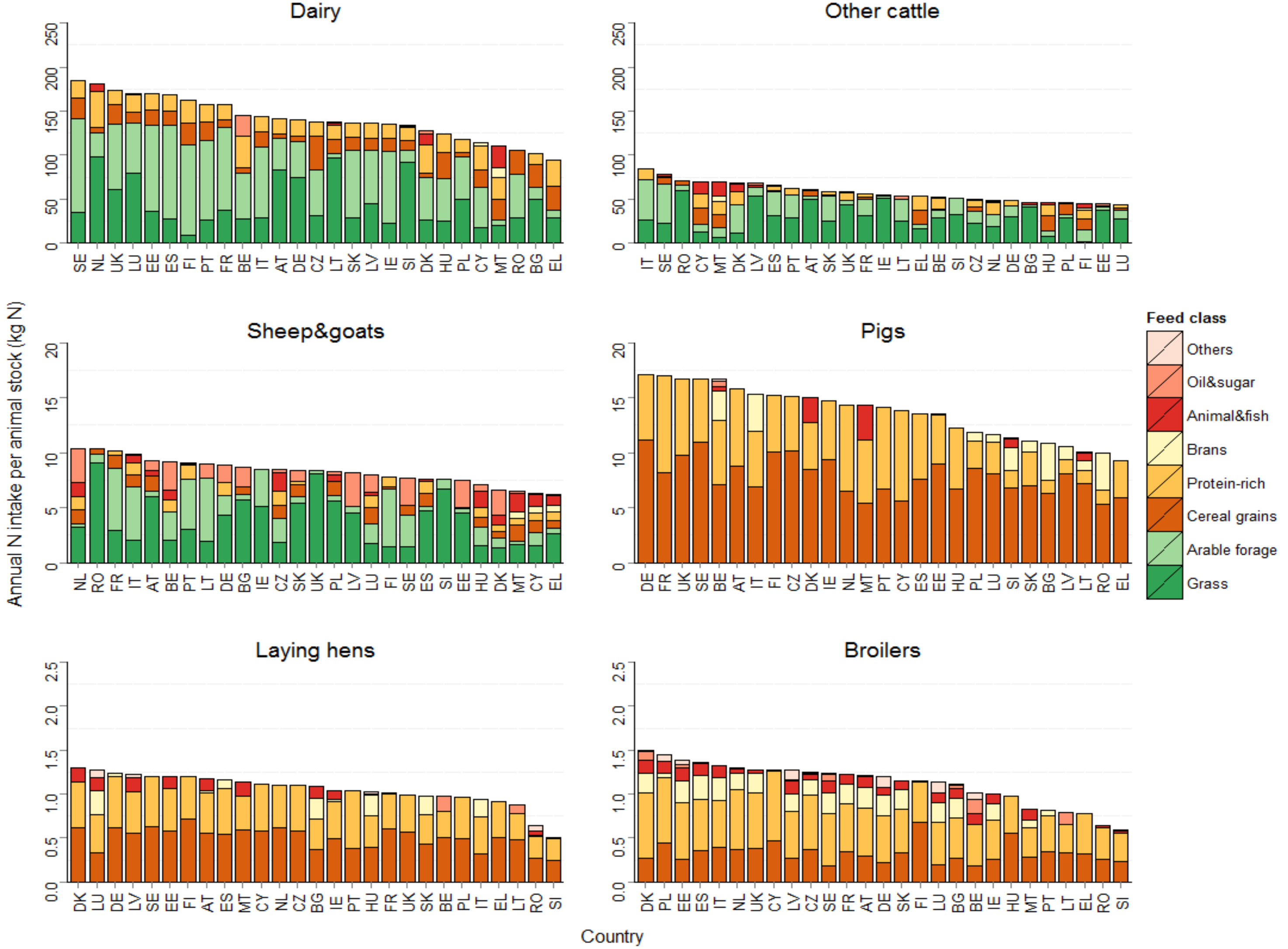

Figure 3-5. Annual average feed nitrogen (N) intake per animal stock per country for the main livestock categories in the EU-27. Notice that member states on X-axis are presented in decreasing order. Country abbreviations are explicated in Table 3-2.

3.3.3 Nitrogen excretion

The total amount of N excreted by animals in the EU-27 was 9.7 Tg yr^{-1} for the period 2009-2011 (Table 3-2). Dairy cows contributed 27%, other cattle 35%, pigs 15%, chicken (i.e., layers and broilers) 10%, sheep and goats 9% (Table 3-2). Countries with the largest total N excretions were France and Germany, followed by UK, Italy, Spain, Poland and The Netherlands. The average annual N excretion per ha of utilized agricultural land (UAA) in the EU-27 was 54 kg ha^{-1}, ranging from < 25 kg ha^{-1} in Bulgaria and Latvia to more than 170 kg ha^{-1} in Belgium and The Netherlands (Table 3-2).

Table 3-2. Total nitrogen (N) excretion per animal category and per country (Gg N yr^{-1}) and N excretion per ha of utilized agricultural land (kg N ha^{-1} UAA) in the EU-27 (1 Gg = 10^9g).

Country names (abbreviations)	N excretion (Gg N yr^{-1})						National sum (Gg N yr^{-1})	National sum as % of EU-27 totals	N excretion per ha (kg N ha^{-1} UAA) [a]
	Dairy cows	Other cattle	Pigs	Chicken	Sheep &Goats	Others			
Austria (AT)	57	80	32	12	3	2	177	2	61
Belgium (BE)	57	95	71	13	1	<1	238	2	176
Bulgaria (BG)	26	10	6	12	14	2	68	1	14
Cyprus (CY)	2	2	4	2	3	-	14	<0.5	121
Czech (CZ)	39	45	19	18	2	<1	117	1	35
Denmark (DK)	46	62	138	15	<1	<1	256	3	98
Estonia (EE)	13	6	3	2	<1	-	26	<0.5	26
Finland (FI)	34	25	14	7	<1	<1	84	1	36
France (FR)	457	819	169	108	84	93	1563	18	56
Germany (DE)	424	367	280	95	19	50	1309	13	74
Greece (EL)	16	20	7	20	73	<1	136	1	35
Hungary (HU)	30	17	25	20	9	11	106	1	20
Ireland (IE)	120	281	15	10	38	5	454	5	106
Italy (IT)	208	309	88	98	84	83	848	9	67
Latvia (LV)	18	14	3	3	<1	-	35	<0.5	21
Lithuania (LT)	41	20	7	4	<1	<1	72	1	26
Luxembourg (LU)	5	6	<1	<1	<1	-	12	<0.5	94
Malta (MT)	<1	<1	<1	<1	<1	<1	3	<0.5	226
Netherlands (NL)	219	98	133	78	14	7	553	6	291
Poland (PL)	237	130	109	100	3	28	646	6	40
Portugal (PT)	29	72	15	22	23	15	152	2	48
Romania (RO)	121	66	43	34	96	-	291	4	26
Slovakia (SK)	17	18	5	9	3	-	45	1	28
Slovenia (SI)	12	17	3	<1	1	<1	32	<0.5	71
Spain (ES)	109	310	239	117	150	3	830	10	39
Sweden (SE)	49	86	17	13	4	<1	141	2	55
United Kingdom (UK)	250	445	50	119	249	18	1012	12	66
Sum of EU-27	2636	3419	1498	932	877	320	9681	100	54

- Not considered or no available data.

[a] The total utilized agricultural area (UAA) were derived from Eurostat (three-year averages of 2009-2011).

Country- and animal- specific N excretion coefficients expressed in kg N per animal per year are presented in Table 3-3. Differences in N excretion coefficients between countries were large. The average N excretion coefficients were 111 kg for dairy cows (range 75-141 kg), 52 kg for other cattle (39-73 kg), 9.9 kg for pigs (6-12 kg), 0.74 kg for laying hens (0.25-0.95 kg), 0.73 kg for broilers (0.15-0.92 kg), 8 kg for sheep (6-10 kg) and about 7 kg for goats (4-12 kg) in the EU-27.

Table 3-3. Country-specific nitrogen (N) excretion coefficients of the main animal categories in the EU-27 (kg N animal $^{-1}$ yr^{-1}).

Country	N excretion coefficient (kg N animal^{-1} yr^{-1})								
	Dairy cows	Other cattle	Pigs	Laying hens	Broilers	Sheep	Goats	Turkey	Other poultry
Austria	107	54	10.1	0.86	0.77	8.1	8.5	3.9	0.8
Belgium	113	46	11.1	0.65	0.26	7.6	10.7	1.5	-
Bulgaria	83	42	7.6	0.86	0.69	8.1	7.2	-	1.1
Cyprus	79	63	9.8	0.83	0.83	6.1	4.9	-	-
Czech	101	47	10.0	0.83	0.78	7.6	9.5	-	1.4
Denmark	82	63	10.8	0.90	0.92	6.0	-	-	1.2
Estonia	133	41	9.3	0.87	0.86	7.0	8.3	-	-
Finland	118	39	10.3	0.86	0.62	7.8	-	2.3	-
France	123	52	11.9	0.71	0.77	8.7	10.7	2.8	0.9
Germany	101	43	10.5	0.87	0.79	8.0	8.5	4.0	1.2
Greece	75	46	6.2	0.74	0.56	5.9	4.1	3.1	0.7
Hungary	96	44	7.8	0.81	0.51	7.1	6.1	2.0	0.7
Ireland	109	50	10.0	0.82	0.67	8.0	-	2.7	1.1
Italy	114	73	9.6	0.73	0.78	9.6	7.4	2.6	-
Latvia	110	64	7.3	0.87	0.77	7.5	8.8	-	-
Lithuania	113	49	7.6	0.66	0.24	7.7	10.3	2.0	0.6
Luxembourg	131	39	7.8	0.95	0.76	6.4	8.9	-	-
Malta	80	62	10.3	0.83	0.58	5.7	4.6	2.5	-
Netherlands	141	41	10.8	0.74	0.82	8.4	12.2	5.1	1.3
Poland	93	41	7.6	0.74	0.86	8.3	6.7	2.7	0.8
Portugal	118	58	7.7	0.66	0.53	9.0	5.9	2.3	-
Romania	88	65	7.4	0.52	0.29	10.2	6.2	-	-
Slovakia	107	57	7.4	0.76	0.71	8.0	8.0	-	-
Slovenia	104	46	8.0	0.24	0.15	7.1	6.3	0.5	-
Spain	130	60	9.3	0.86	0.84	6.9	8.1	3.7	-
Sweden	140	73	11.1	0.86	0.80	7.3	-	3.4	-
UK	135	54	11.2	0.74	0.79	7.9	-	3.9	1.1
Average[a]	111	52	9.9	0.74	0.73	8.0	6.6	2.9	0.9
Median	109	50	9.6	0.82	0.77	7.7	8.1	2.7	1.1
SD[b]	20	10	1.6	0.14	0.22	1.1	2.1	1.1	0.2

[-] Not considered.
[a] The EU-27 weighted average.
[b] SD indicates the difference between countries.

3.3.4 Feed conversion and nitrogen use efficiency

Table 3-4 shows the calculated average feed conversion ratio (FCR), N use efficiency (NUE), feed energy conversion to edible energy ratio (FER) and feed-to-animal protein conversion ratio (FPR) for the main animal categories in the EU-27. Beef cattle had on average the highest FCR, FPR and FER, and the lowest NUE. The FCR was relatively low for dairy cows, compared to other animal categories. Broilers had the highest NUE and the lowest FPR. The average FER of pigs was lower than that of other categories.

Table 3-4. Feed conversion ratio (FCR), nitrogen use efficiency (NUE), feed energy conversion to edible energy ratio (FER) and feed-to-animal protein conversion ratio (FPR), expressed in weighted averages (±SD) for the EU-27[a].

Animal category (product)	FCR (kg kg^{-1})	NUE (%)	FER (MJ MJ^{-1})	FPR (kg kg^{-1} protein)
Dairy cows (cow milk)	1.0± 0.2	23 ± 4	5.1 ± 1.0	30 ± 6
Other cattle (live-weight gain)	12.6 ± 5.6	9 ± 3	22.7 ± 9.6	77 ± 35
Pigs (live-weight gain)	2.9± 0.5	33 ± 5	3.1± 0.5	19 ± 3
Laying hens (eggs)	2.5 ± 0.5	26 ± 7	7.3 ± 1.6	22 ± 5
Broilers (live-weight gain)	2.4 ± 0.5	40± 14	4.3 ± 0.7	13 ± 3

[a] FCR indicates the dry mass of feed use per mass of animal products (milk and eggs) or live-weight gain; NUE indicates the amount of nitrogen retained in animal products as percentage of total feed nitrogen intake; FER indicates the feed energy conversion into per unit human edible energy of animal product; FPR indicates the dry mass of feed use per kg of protein in animal products.

3.3.5 Sensitivity analyses

Increasing animal production by 5% increased total feed use by 2.2% (~11 Tg) in EU-27, and by 4.5% (~23 Tg) when simultaneously increasing animal number by 5% (Figure 3-6). A 5% increase of animal production and animal number resulted in a 4 % increase in N excretion. Altering average live weight of animals by 5% resulted in a ~2.4% change in feed DM use and in a ~3.0% change in total N excretion. Changing energy contents of cereals and grass by 5% changed total feed use with about 7-8 Tg, or 1.3-1.5% of the baseline value. The effect of changes in feed protein content (grass, cereals and protein-rich feed) on total feed use was minute (Figure 3-6). However, changing the N content in grass by 10% changed total N excretion in EU-27 by ~4% relative to the baseline value (Figure 3-6). These effects (sensitivity extents) varied largely between countries (see standard deviations present in Figure 3-6)

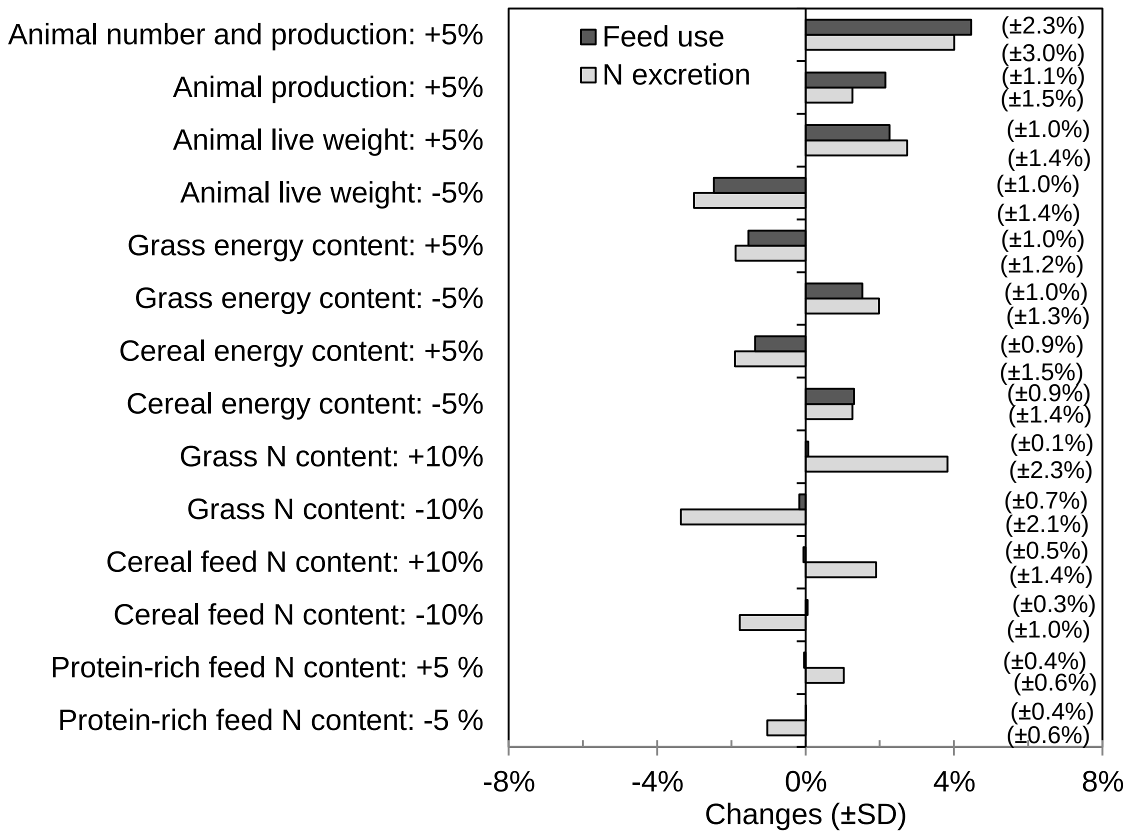

Figure 3-6. Effects of changing parameters on total feed dry mass use and nitrogen (N) excretion of the animals in the EU-27. Changes from the baseline by the respective assumptions were calculated and expressed as a percentage of the baseline value. Standard deviations (SD) are present in the parentheses, indicating the variance between countries. For more details see Section 3.2.7.

3.4 Discussion

We developed a transparent methodology for the estimation of feed use and N excretion by livestock in all countries of the EU-27, based on the mass balance of feed supply and demand, statistical data and an optimization routine. The advantage of this methodology is that feed use and N excretions are estimated in a uniform manner for all Member States. Possible changes in feed supply and animal production statistics will be reflected in feed use and N excretion coefficients.

Currently, there are large differences within and between countries in the methodologies applied for estimating and reporting N excretion coefficients that serve national inventories of NH_3 and greenhouse gas emissions (Velthof *et al.*, 2015). Our methodology could be used as a benchmark for national estimates. The methodology designed here for quantifying N excretions in the EU can easily be applied for the estimation of other nutrient excretions (e.g., phosphorus, potassium), based on the calculated feed use and the nutrient contents of feed and animal products. Our methodology can be used also for global assessments. We used three-years averages (2009-2011) to obtain a robust mean, but the methodology is equally applicable to annual data.

Linking feed use to N excretion coefficients facilitates the analysis of emission mitigation potentials of animal diet-related strategies. For example, decreasing the proportion of protein-rich feed in the ration of animals to an optimal level is an effective measure to reduce N emissions from the whole manure management chain (Hou *et al.*, 2015). Linking feed use to specific animal categories allows also to allocate feed production related resource use (e.g., land, water, fertilizers) and greenhouse gas emissions to individual animal categories and animal food products (e.g., Tilman and Clark, 2014). Ranking various animal products according to the associated environmental (e.g., N footprints) and human health impacts may facilitate the implementation of revised dietary recommendations (Eshel *et al.*, 2014; Galloway *et al.*, 2014). Our method addresses various key livestock categories simultaneously, using a uniform methodology and common national statistics, and provides national averages, which allow direct comparison of feed use and N excretion coefficients among countries.

3.4.1 Differences between the Member States

Dairy cattle is the dominant animal category in the EU-27 in terms of feed use, N excretion (Figure 3-4), and emissions of NH_3 and greenhouse gases (Lesschen *et al.*, 2011). Countries with large dairy herds in 2009-2011 were Germany, France, Poland, UK, Italy, The Netherlands, together using nearly 70% of the total dairy feed use in EU-27 (Table S6). However, the rations of dairy cows differed between these countries (Figure 3-3). Dairy cows in UK and The Netherlands were fed grass-based diets, while dairy cows in Poland and Italy were fed relatively large portions of annual forages (Figure 3-3), which reflects the estimated availability of grass and annual forages in these countries (Figure 3-2). Statistics Netherlands reported that the average ration of dairy cows in The Netherlands during 2006-2008 included 54-64% grass and 12-27% maize silage, depending on region (CBS, 2012), which is comparable with our results (60% grass and 18% annual forages at national level). The feed use per dairy cow was related to milk yield and feed use efficiency (Figure 3-3). Some countries (e.g., Spain, Sweden) had relatively high milk yields (>7500 kg milk yr^{-1}) and high feed intake (nearly 7500 kg DM yr^{-1}) per animal, while other countries (e.g., Bulgaria, Greece, Romania) had relatively low milk yields (<4000 milk yr^{-1}) and low feed dry matter intake

(<5000 kg DM yr^{-1}). Other countries (e.g., The Netherlands, UK) combined a relatively high milk yield (> 7400 kg milk yr^{-1}) with a modest feed dry matter intake per head (6500~7000 kg DM yr^{-1}), which can be explained by relatively high quality feed and good feed management. Results from the present study show that the aforementioned countries with high milk yields appear to have a relatively efficient feed use at animal level (i.e., FCR ranges between 0.8-1.0; and NUE ranges between 22-25%, Tables S10-11), compared to countries with low milk yields (FCR ranges between 1.3-1.4; and NUE ranges between 17-20%). Our NUE estimates for dairy cows are similar to estimates derived from studies at dairy farms in Spain (mean NUE of 22%; Arriaga et al., 2010) and The Netherlands (NUE ranged from 23-26%; Oenema et al., 2012).

Large pork producers include Germany, Spain, France, Poland, Denmark, The Netherlands and Italy; these seven countries accounted for 75% of the total pig feed use in the EU-27. Differences in the rations between countries were relatively small, with cereals as the largest feed source in all countries (Figure 3-3). This study estimated that over 34% of protein-rich feed and nearly 53% of feed cereals in EU-27 were used by pigs. Herrero et al. (2013) also reported that most of feed concentrates (cereals, pulses, etc.) are fed to pigs (~50%) and poultry (~20%) in regions where industrial systems dominate (e.g., Europe, North America). The quantity of feed use per head per year varied greatly between Member States (Figure 3-3), mainly due to differences in the live-weight gain per stock, which is related to the live weigh at slaughtering and pig husbandry characteristics. A relatively high feed intake (~680 kg DM per stock) was estimated for Germany and Italy due to a high live-weight gain per stock (265 and 230 kg, respectively) and a high live weight at slaughtering (123 and 157 kg, respectively). Feed use (Figure 3-3) and live weight gain (~140 kg) per stock were relatively low in the Netherlands, which is in part related to the export of piglets to other countries, including Germany and Italy.

Large differences were observed between countries for the total amount of N in manure and the amount of N excreted per ha of utilized agricultural area (UAA) (Table 3-2). A relatively high N excretion in kg N ha^{-1} UAA was found for the Netherlands, Belgium, Ireland and Denmark (Table 3-2). These countries with high animal N excretion loads have also relatively high N losses (NH_3 and N_2O emissions, N leaching, etc.) per ha of UAA (Oenema *et al.*, 2007; Velthof *et al.*, 2009).

3.4.2 Feed supply – use balance

The total feed dry mass potentially available in the EU-27 in 2010 was estimated at 593 Tg, and the calculated total feed use by the main animal categories amounted to 506 Tg. Lesschen et al. (2011) estimated a similar amount of feed use (509 Tg) for the EU-27 in 2005. The difference (87 Tg) between the amount of feed potentially available and the calculated feed requirement may be explained by various factors. Firstly, minor animal categories (horses, fur animals, rabbits, etc.) and animals at 'hobby farms' were not taken into account. Secondly, feed conservation losses and feed wastes were not considered. Feed conservation losses of grass and forage silages are usually in the range of 5-10% and those for concentrate feeds in the range of 2 to 5% (McGechan, 1989, 1990; McCormick *et al.*, 2011). Feed wastes (i.e., the feed discarded by the animals) are also in the range of 2-5%, depending on feed quality. Thirdly, there are competing uses for some forages, such as silage maize in anaerobic digesters (Pedroli & Langeveld, 2011). This in part explains that forages from arable land (silage maize, leguminous crops, root crops, crop residues etc.) had a relatively large mismatch between supply and use in our calculations. Fourth, there are uncertainties in the feed statistics, particularly in grass and forage statistical data (Smit *et al.*, 2008), and uncertainties in animal numbers. For example, there is transport of live animals across some borders (summarized in Table S8; FAOSTAT, 2014), which is related to the specialization of animal production systems in some regions. Some countries import animals for raising the animals to final weight, while other countries may import animals at slaughter weight to fully utilize the slaughterhouse capacity. We did not correct for net import, because of the uncertainties related to live weight of the imported animals. Thus, the import-export of live animals creates uncertainties in the estimated national feed use and N excretion coefficients.

3.4.3 Comparison of N excretion with other studies

Nitrogen excretion coefficients are critically important parameters for agricultural N balances and national inventories of NH_3 and N_2O emissions and nitrate leaching from agriculture, which have to be reported for the evaluation and underpinning of EU and United Nations (UN) policies (Oenema et al., 2011; Velthof et al., 2015). The impacts of these agri-environmental policies and possible changes in these policies are evaluated by integrated assessment models such as GAINS (Asman *et al.*, 2011) and CAPRI (Britz & Witzke, 2012). Both models estimate N excretions and emissions of NH_3 and N_2O from animal manures per country, but use in part different approaches. Our estimates of the N excretion per animal category per country were compared with those from Eurostat, national inventory reports (NIRs) to UNFCCC, GAINS and CAPRI (Figure 3-7). The total N excretion estimates for GAINS and CAPRI models were updated, using the animal number data from Eurostat for the years 2009-2011, but the animal excretion coefficients from GAINS and CAPRI. There is a good agreement between the estimates for total N excretion per country (Figure 3-7a) and the total N excretion by cattle (Figure 3-7b). Total N excretions by pigs (Figure 3-7c) and chicken (Figure 3-7d) are largely comparable, except for those from CAPRI. CAPRI reported relatively high N excretions for pigs and relatively low values for chicken. The reason for these differences is unclear, but it is likely due to the differences in the methods and data sources used to estimate N excretion coefficients. Differences between our estimates and those from Eurostat and NIRs were also relatively large for some countries, e.g., Romania and France (Figure 3-7). These differences are likely related to the use of IPCC default coefficients in the national reports (Velthof *et al.*, 2015).

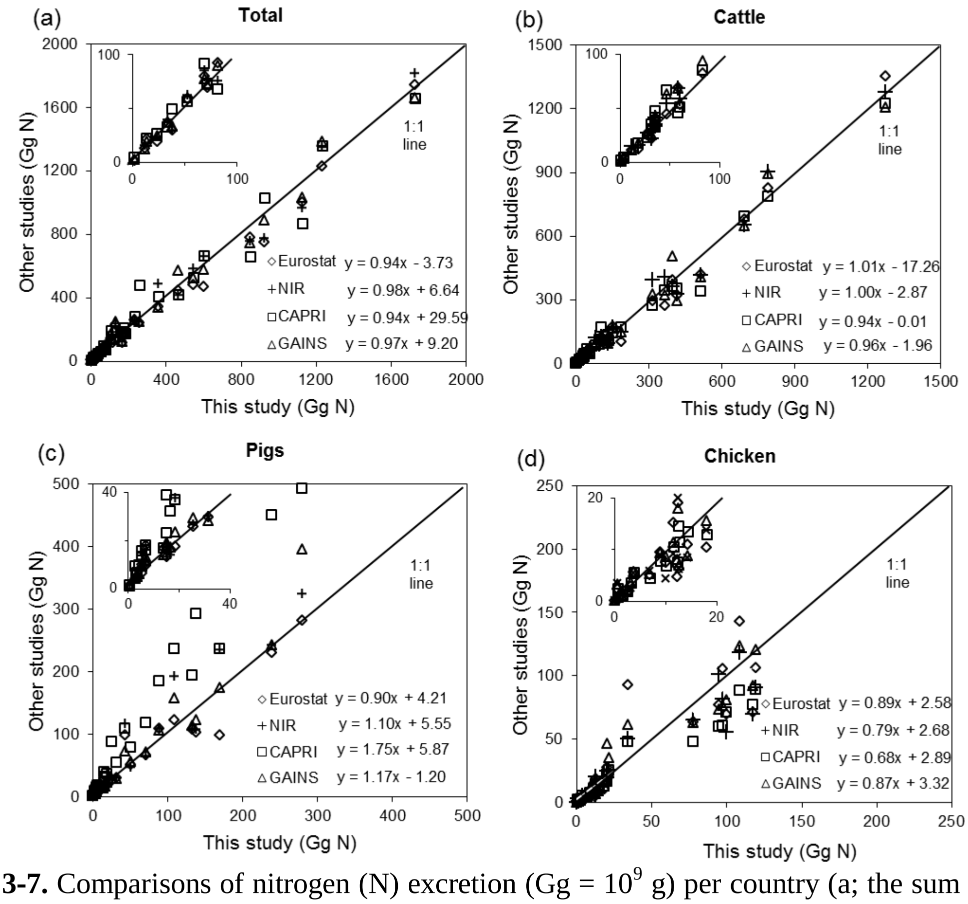

Figure 3-7. Comparisons of nitrogen (N) excretion (Gg = 10^9 g) per country (a; the sum of all animal categories) or per animal category (b–d) with other studies: Eurostat, National inventory reports (NIR) to the UNFCCC, CAPRI model and GAINS model. Linear equations are shown corresponding to each pair of comparison. Magnified subfigures are included.

3.5 Conclusions

Our study is the first assessment of feed use and N excretion per animal category and country in the EU-27, using a uniform methodology. Such a methodology is a necessary first step for environmental impact assessments of livestock production and manure management (i.e., the whole feed production-to-manure use chain) at animal category, at regional and national scales. Currently, various approaches and methods are being used for estimating the total N excretion per animal category at regional and national levels. These different approaches often lead to different estimates, and complicate comparisons between regions and countries. Our methodology could be used as a benchmark for these studies.

The total annual feed dry mass use by livestock in EU-27 during the years 2009-2011 was estimated at ~506 Tg, and total N excretion at ~9.7 Tg. There were large differences between countries in feed use per animal category, which led to differences in feed use efficiency and N excretion per animal category. Hence, feed use and management must be considered in environmental impact studies of livestock production and manure management systems.

We estimated a relatively large difference (87 Tg) between total potential feed supply and total feed requirement in EU-27. We identified several possible reasons for this difference, which must be explored further. This difference may however, also have implications for agri-environmental policies. Evidently, our methodology allows to assess the effects of possible changes in for example agricultural policy (CAP reform) and biofuel policy on feed use, feed use efficiency and N excretion per animal category. Accurate estimations of N excretions are also essential for assessing N emission mitigation measures and policies for animal production systems.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2015.11.025.

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CHAPTER 4

Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: a meta-analysis and integrated assessment

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Abstract:

Livestock manure contributes considerably to global emissions of ammonia (NH_3) and greenhouse gases (GHG), especially methane (CH_4) and nitrous oxide (N_2O). Various measures have been developed to mitigate these emissions, but most of these focus on one specific gas and/or emission source. Here, we present a meta-analysis and integrated assessment of the effects of mitigation measures on NH_3, CH_4 and (direct and indirect) N_2O emissions from the whole manure management chain. We analysed the effects of mitigation technologies on NH_3, CH_4 and N_2O emissions from individual sources statistically using results of 126 published studies. Whole-chain effects on NH_3 and GHG emissions were assessed through scenario analysis. Significant NH_3 reduction efficiencies were observed for i) housing via lowering the dietary crude protein (CP) content (24-65%, compared to the reference situation), for ii) external slurry storages via acidification (83%) and covers of straw (78%) or artificial films (98%), for iii) solid manure storages via compaction and covering (61%, compared to composting), and for iv) manure application through band spreading (55%, compared to surface application), incorporation (70%) and injection (80%). Acidification decreased CH_4 emissions from stored slurry by 87%. Significant increases in N_2O emissions were found for straw-covered slurry storages (by two orders of magnitude) and manure injection (by 26-199%). These side-effects of straw covers and slurry injection on N_2O emission were relatively small when considering the total GHG emissions from the manure chain. Lowering the CP content of feed and acidifying slurry are strategies that consistently reduce NH_3 and GHG emissions in the whole chain. Other strategies may reduce emissions of a specific gas or emissions source, by which there is a risk of unwanted trade-offs in the manure management chain. Proper farm-scale combinations of mitigation measures are important to minimize impacts of livestock production on global emissions of NH_3 and GHG.

4.1 Introduction

Livestock farming systems are main sources of emissions of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4). Emissions of NH_3 are largely responsible for the acidification and eutrophication of nitrogen-limited ecosystems (Sutton *et al.*, 2008). Emissions of N_2O and CH_4 contribute considerably to the radiative forcing of the atmosphere, as the global warming potentials of N_2O and CH_4 are, respectively, 298 and 25 times higher than that of CO_2 per kg (IPCC, 2007). Approximately 40% of the global anthropogenic NH_3 and N_2O emissions are associated with manures from livestock production (Galloway *et al.*, 2004; Oenema *et al.*, 2005). Enteric fermentation and animal manure together contribute some 80% to the global CH_4 emissions from agriculture and about 35–40% to the global anthropogenic CH_4 emissions (Steinfeld *et al.*, 2006).

Emissions of NH_3, CH_4 and N_2O may occur simultaneously from different sources of manure management systems. Animal excreta in housing and manure storage systems, from grazing animals voided on pastures and from land following manure application are main sources of NH_3 and N_2O. Enteric fermentation in ruminants is the dominant source of CH_4 emissions. Manure storages are also a significant source of CH_4 (Sommer *et al.*, 2004). Most agricultural soils are a sink for CH_4 and a source of N_2O, depending on drainage, soil properties, fertilization practices and climatic conditions (Oenema *et al.*, 2001).

Series of measures have been developed to address manure-related emissions, and some have been implemented successfully in practice. However, effects of these measures are typically considered for a specific gas or emission source only (e.g. Petersen *et al.*, 2007), although it is well-known now that measures may have possible environmental side-effects (i.e. pollution swapping). For example, some NH_3 mitigation measures may increase N_2O emissions from slurry storages, or enhance CH_4 emissions from solid manures storages (Berg *et al.*, 2006; Szanto *et al.*, 2007; Hansen *et al.*, 2009; Velthof & Mosquera, 2011). Several recent studies have addressed the possible side-effects of NH_3 mitigation measures on emissions of N_2O and CH_4, but a systematic quantitative assessment of the effects of mitigation options in the manure chain is still lacking (Novak & Fiorelli, 2010; Webb *et al.*, 2010; Chadwick *et al.*, 2011; Montes *et al.*, 2013).

Manure processing creates alternative nutrient management opportunities by producing manure products (e.g. anaerobic digestate, separated liquid and solid fractions, compost) that differ from untreated manure, and it can also induce changes in NH_3 and GHG emissions

(Sommer *et al.*, 2009). Manure processing has become increasingly popular in many countries. For instance, the European Commission recently conducted a survey on manure processing activities in 27 member states of the European Union (EU-27). The results of this survey show that manure processing currently has reached an average level of 7.8% of the livestock manure production in EU-27, although regional variations were large (Foged *et al.*, 2011). Anaerobic digestion and separation technologies were responsible for the processing of 3.5% and 3.1% of the total livestock manure production in the EU, respectively (Foged *et al.*, 2011). The fraction of manure processed is expected to increase in the near future in order to achieve the targets of governmental policies related to further increasing the use efficiencies of manure nutrients, mitigating NH_3 and GHG emissions and to renewable energy. Therefore, there is a need to improve our understanding about emissions of NH_3 and GHG from the use of manure processing products.

The overall objective of the current study is to make a quantitative assessment of the effects of (sets of) mitigation options on the NH_3, N_2O and CH_4 emissions from the whole manure chain, namely livestock housing, manure storage and land application. Firstly, the impacts of a suite of NH_3 mitigation measures on NH_3 emissions at individual stages, and also the associated impacts on N_2O and CH_4 emissions were analysed by means of a meta-analysis of published data. Secondly, we evaluated the overall impacts of combinations of mitigation measures (including manure processing) on NH_3, CH_4, and (direct and indirect) N_2O emissions from the whole manure management chain through scenario analysis.

4.2 Materials and Methods

4.2.1 Manure management chain and emission mitigation measures

An overview of studies examined here is shown in Table 4-1. These studies focused on different stages of the manure management chain (subjected to different management practices), namely housing (different dietary crude protein content in the animal feed, different floor constructions), storage of liquid manure (different covers, and acidification) and solid manure (compaction, covering, stockpiling, composting), and manure application to land (different application techniques, different processed manure products). Within each of these stages, emissions of NH_3, N_2O and CH_4 from manure are compared between given management practices (see Table 4-1). Pig and cattle manures were mainly considered in this meta-analysis. Measures for cattle manure during grazing and feeding strategies affecting CH_4 emissions from enteric fermentation were excluded in the present study.

4.2.2 Literature search and study selection

Studies related to manure management and emissions of NH_3, N_2O and CH_4 were searched using the bibliographic database Scopus, until the beginning of 2014. Specific search terms were combined, depending on animal category (*animal, livestock, pig, swine, cattle or cow*), manure type (*slurry, waste, manure, compost, farmyard manure, digestate, liquid* or *solid*), management measures (feeding: *feed, diet, dietary manipulation* or *dietary crude protein*; animal housing: *housing, barn, slatted floor, deep litter, solid floor* or *straw floor*; manure processing: *acidification, acidified, separation, separated, digestion* or *biogas*; slurry storage: *storage, crust* or *cover*; solid manure storage: *compaction, cover, stockpiling, static piling, turning* or *compost*; field application: *band spreading, trailing hose, trailing shoe, injection, injected, incorporation* or *incorporated*), and emissions (*ammonia, methane, nitrous oxide* or *greenhouse gas emissions*). For example, literature related to NH_3 emissions from acidified pig slurry was searched with the following combination of terms "*pig* OR *swine* AND *slurry* OR *manure* AND *acidification* OR *acidified* AND *ammonia*" in article titles, abstracts and keywords in Scopus.

Only data from studies with reference treatments (i.e. without mitigation/processing measures) were included in our database, so as to allow side-by-side comparisons. To maximize the number of studies, both laboratory and field experiments were taken into account. The selected studies in our database and grouped side-by-side comparisons are listed in Table 4-1. The reported experiments and measurements were predominately conducted in EU, United States of America and Canada. Mean values of replicates for each treatment were included in the database. Manure characteristics (e.g. manure type, dry matter content, total N content, ammoniacal N content and pH), land use parameters (e.g. soil texture, vegetation), environmental conditions (e.g. temperature, seasons and geographic locations) were also included, and used to, if possible, quantify their relationships with emissions and the effectiveness of the measures.

Table 4-1 Matrix showing which studies provided data for each stage of manure management on which effect sizes were estimated

Management stages	Emission sources targeted	Pollutants	Grouped comparisons	References
Feeding	Housing	NH_3	<2 % reduction in dietary CP 2-4 % reduction in dietary CP >4 % reduction in dietary CP	Canh *et al.*, 1998a; James *et al.*, 1999; Külling *et al.*, 2001; Frank & Swensson, 2002; Frank *et al.*, 2002; Portejoie *et al.*, 2004; Hayes *et al.*, 2004; Leek *et al.*, 2005, 2007; Velthof *et al.*, 2005; O'Connell *et al.*, 2006; Panetta *et al.*, 2006; Lynch *et al.*, 2007, 2008; Hansen *et al.*, 2007; Le *et al.*, 2008, 2009; van der Stelt *et al.*, 2008; Li *et al.*, 2009; O'Shea *et al.*, 2009; Agle *et al.*, 2010; Arriaga *et al.*, 2010; Galassi *et al.*, 2010; Hernández *et al.*, 2011; Lee *et al.*, 2012; Koenig *et al.*, 2013; Madrid *et al.*, 2013; Montalvo *et al.*, 2013
Housing	Housing	NH_3	slatted floor/deep litter vs solid floor slatted floor vs deep litter litter removal frequently vs infrequently extra straw addition vs no extra addition	Groenestein & Van Faassen, 1996; Kavolelis, 2006; Gilhespy *et al.*, 2009; Amon *et al.*, 2007; Philippe *et al.*, 2007a, 2007b, 2011, 2013; Ivanova-Peneva *et al.*, 2008; Cabaraux *et al.*, 2009; Dourmad *et al.*, 2009
		N_2O, CH_4	slatted floor vs deep litter litter removal frequently vs infrequently	Groenestein & Van Faassen, 1996; Amon *et al.*, 2007; Philippe *et al.*, 2007a, 2007b, 2011; Cabaraux *et al.*, 2009; Dourmad *et al.*, 2009
Acidification	External storage	NH_3	acidified vs not acidified	Kai *et al.*, 2008; Petersen *et al.*, 2012, 2014; Dai & Blanes-Vidal, 2013; Wang *et al.*, 2014
		CH_4	acidified vs not acidified	Petersen *et al.*, 2012, 2014; Wang *et al.*, 2014
Slurry storage	External storage	NH_3	crusting vs no-crust straw cover vs no-cover wooden lid vs no-cover wooden lid vs crusting granules cover vs no-cover artificial film cover vs no-cover peat/kitchen oil cover vs no-cover	Sommer, 1997; Hörnig *et al.*, 1999; Portejoie *et al.*, 2003; Misselbrook *et al.*, 2005b; Balsari *et al.*, 2006; Berg *et al.*, 2006; Clemens *et al.*, 2006; Amon *et al.*, 2007; Smith *et al.*, 2007; VanderZaag *et al.*, 2009; VanderZaag & Gordon, 2010; Petersen *et al.*, 2013
		N_2O, CH_4	straw cover vs no-cover wooden lid vs no-cover wooden lid vs crusting granules cover vs no-cover granules cover + acids vs no-cover/acids artificial film cover vs no-cover	Berg *et al.*, 2006; Clemens *et al.*, 2006; Amon *et al.*, 2007; Hansen *et al.*, 2009; VanderZaag *et al.*, 2009; VanderZaag & Gordon, 2010; Rodhe *et al.*, 2012; Petersen *et al.*, 2013

Table 4-1 (continued)

Solid manure storage	External storage	NH_3	stockpiling vs turning (composting) compaction/covering vs turning	Sommer & Dahl, 1999; El Kader *et al.*, 2007; Sagoo *et al.*, 2007; Szanto *et al.*, 2007; Jiang *et al.*, 2013
		N_2O, CH_4	stockpiling vs turning (composting) compaction/covering vs turning	Sommer & Dahl, 1999; Hao *et al.*, 2001; El Kader *et al.*, 2007; Szanto *et al.*, 2007; Ahn *et al.*, 2011; Jiang *et al.*, 2013
Field application	Arable land and grassland	NH_3	band spreading vs surface spreading incorporation vs surface spreading injection vs surface spreading	Malgeryd, 1998; Smith *et al.*, 2000; Huijsmans *et al.*, 2001; Wulf *et al.*, 2002a; Misselbrook *et al.*, 2002; Rodhe & Karlsson, 2002; Huijsmans, 2003; Mattila & Joki-Tokola, 2003; Webb *et al.*, 2004, 2014; McGinn & Sommer, 2007; Sagoo *et al.*, 2007; Bhandral *et al.*, 2009
		N_2O	injection/incorporation vs surface spreading injection/incorporation vs band spreading	Sommer *et al.*, 1996; Flessa & Beese, 2000; Wulf *et al.*, 2002b; Velthof *et al.*, 2003; Webb *et al.*, 2014, 2004; Vallejo *et al.*, 2005; Weslien *et al.*, 2006; Rodhe *et al.*, 2006; Bhandral *et al.*, 2009; Sistani *et al.*, 2010; Thomsen *et al.*, 2010; Velthof & Mosquera, 2011
		NH_3	anaerobic digestate vs raw slurry separated liquid fractions vs raw slurry separated solid fractions vs raw slurry	Wulf *et al.*, 2002a; Mattila *et al.*, 2003; Amon *et al.*, 2006; Chantigny *et al.*, 2007, 2009; Balsari *et al.*, 2008; Bhandral *et al.*, 2009; Möller & Stinner, 2009; Dinuccio *et al.*, 2011, 2012; Monaco *et al.*, 2011; Nyord *et al.*, 2012
		N_2O	anaerobic digestate vs raw slurry separated liquid fractions vs raw slurry separated solid fractions vs raw slurry	Petersen, 1999; Vallejo *et al.*, 2006; Clemens *et al.*, 2006; Chantigny *et al.*, 2007; Fangueiro *et al.*, 2007, 2008b, 2010; Bertora *et al.*, 2008; Bhandral *et al.*, 2009; Möller & Stinner, 2009; Pereira *et al.*, 2010b; Thomsen *et al.*, 2010; Chiyoka *et al.*, 2011; Collins *et al.*, 2011; Dinuccio *et al.*, 2011; Schouten *et al.*, 2012

4.2.3 Derivation of the effect size and statistical analysis

Standardization of the literature results was undertaken through calculation of the effect size (i.e. a measure of comparing two variables). This allows quantitative statistical information to be pooled, and robust statistical comparisons to be made between effects from a range of studies that reported results based on different experimental variables. For each side-by-side comparison (i.e. observation), the natural logarithm of the response ratio was calculated as the effect size ($\ln R$):

$$\ln R = \ln (X_A/X_B)$$

Where X_A and X_B represent the mean emissions in treatment A (mitigation measure or alternative practice) and treatment B (reference practice), respectively, for NH_3, N_2O and CH_4. Log-transformation of the response ratio was carried out to stabilize the variance. For calculation of grouped effect sizes, a mixed-effects model was used and performed in the nlme package of R statistical software Version 3.1 (Pinheiro *et al.*, 2014). Mixed-effect models are preferable to fixed-effect models for statistical testing in ecological data synthesis because their assumption of variance heterogeneity is more likely to be satisfied (Gurevitch *et al.*, 2001). Experimental sites were considered as a random effect factor, to allow accounting for variances among studies. The $\ln R$ of individual pairwise comparison was the dependent variable. The mean effect size and the 95% confidence intervals (CIs) of each categorical group were estimated. The significance of the effects on emissions was statistically assessed at 0.05 level. In the graphs (forest plots), the "effect-size" (the mean value and 95% CIs) of each grouping was transformed back (i.e. exponentially transformed) and converted to a percentage change in gas emissions relative to the reference treatment.

4.2.4 Integrated assessment of management options

Introducing a new technology or management measure to mitigate emissions from a particular source may affect the emissions downstream in the manure management chain. To examine such possible effects, a scenario analysis was conducted.

Impacts of changes in technologies and management measures were compared with the reference scenario. We defined scenarios for three contrasting pig farming systems, namely a slurry-based system, a solid manure-based system, and a slurry-based system with slurry separation. The reference scenarios (systems) and the NH_3 emissions mitigation scenarios were derived from the UNECE Ammonia Guidance Document (Bittman *et al*., 2014). For the slurry-based system with slurry separation, we selected two contrasting separation techniques; a screw press (low separation efficiency) and a decanter centrifuge (high separation efficiency) (Hjorth *et al.*, 2010). The reference and alternative scenarios are described in Table 4-2.

Emissions of NH_3, N_2O and CH_4 were calculated from the model pig manure management chains, divided into emissions from enteric fermentation (CH_4), houses, storage of manure and field application of manure.

Emissions of NH_3 were calculated using the Tier 2 methodology of the EMEP/EEA air pollutant emission inventory guidebook (EMEP/EEA, 2013), based on the flow of TAN (total ammoniacal N) and emission factors. This methodology enables estimation of the effects of measures on NH_3 emissions at subsequent stages of the manure management chain. Emissions of N_2O were calculated using the IPCC guidelines. Indirect N_2O emissions resulting from NH_3 emissions were included in the assessment (IPCC, 2006). A default emission factor of 0.01 kg N_2O-N per kg NH_3-N emitted was adopted. Indirect N_2O emissions from nitrate leaching were not considered. Methane emissions from enteric fermentation and from storage of slurry and solid manure were based on IPCC guidelines (IPCC, 2006). The overall GHG emissions were calculated and presented as kg CO_2-eq, using the default values of 298 kg CO_2-eq for N_2O emissions and 25 kg CO_2-eq for CH_4 emissions (IPCC, 2007). Effects of new technologies and mitigation measures were estimated from the differences with the reference scenario. Average emission factors for the new technologies and mitigation measures were derived from the effect-size analysis of published data, as described before. Parameters used to calculate emissions from the reference systems and scenarios with mitigation measures and alternative practices can be found in the supplementary file (Table S1 and S2).

Table 4-2. Description of the reference scenario and the emission mitigation measures and technologies used for the scenario analysis

Scenario ID	Housing		Separation [a]	Storage		Field application
	Dietary CP content	Floor types		Slurry/liquid fractions	Solid manure/solid fractions	
			Slurry-based system			
Reference	Conventional	Slatted floor		No-crust cover		Surface spreading
Diet [b]	2% reduction	Slatted floor		Crusting		Surface spreading
Acidification	Conventional	Slatted floor		No-crust cover		Surface spreading
Straw cover	Conventional	Slatted floor		Straw cover		Surface spreading
Artificial film	Conventional	Slatted floor		Artificial film		Surface spreading
Band spreading	Conventional	Slatted floor		No-crust cover		Band spreading
Injection	Conventional	Slatted floor		No-crust cover		Injection
Acidification + injection	Conventional	Slatted floor		Acidification		Injection
Straw + injection	Conventional	Slatted floor		Straw cover		Injection
Artificial film + injection	Conventional	Slatted floor		Artificial film		Injection
Diet + acidification + injection	2% reduction	Slatted floor		Acidification		Injection
Diet + straw cover + injection	2% reduction	Slatted floor		Straw cover		Injection
Diet + artificial film + injection	2% reduction	Slatted floor		Artificial film		Injection
			Solid manure-based system			
Reference	Conventional	Deep litter			Composting	Surface spreading
Diet	2% reduction	Deep litter			Composting	Surface spreading
Stockpiling	Conventional	Deep litter			Stockpiling	Surface spreading
Compaction	Conventional	Deep litter			Compaction	Surface spreading
Incorporation	Conventional	Deep litter			Composting	Incorporation
Stockpiling + incorporation	Conventional	Deep litter			Stockpiling	Incorporation
Compaction + incorporation	Conventional	Deep litter			Compaction	Incorporation
Diet + stockpiling + incorporation	2% reduction	Deep litter			Stockpiling	Incorporation
Diet + compaction + incorporation	2% reduction	Deep litter			Compaction	Incorporation

Table 4-2 (continued)

Slurry separation-inclusive system						
Reference (slurry-based system)	Conventional	Slatted floor		No-crust cover		Surface spreading
SEP I (Screw press)	Conventional	Slatted floor	Screw press	No-crust cover	Composting	Surface spreading
SEP II (Centrifuge)	Conventional	Slatted floor	Centrifuge	No-crust cover	Composting	Surface spreading
SEP I +diet	2% reduction	Slatted floor	Screw press	No-crust cover	Composting	Surface spreading
SEP II + diet	2% reduction	Slatted floor	Centrifuge	No-crust cover	Composting	Surface spreading
SEP I + deep placement	Conventional	Slatted floor	Screw press	No-crust cover	Composting	Deep placement [c]
SEP II + deep placement	Conventional	Slatted floor	Centrifuge	No-crust cover	Composting	Deep placement
SEP I + artificial film	Conventional	Slatted floor	Screw press	Artificial film	Composting	Surface spreading
SEP II + artificial film	Conventional	Slatted floor	Centrifuge	Artificial film	Composting	Surface spreading
SEP I + artificial film + stockpiling	Conventional	Slatted floor	Screw press	Artificial film	Stockpiling	Surface spreading
SEP II + artificial film + stockpiling	Conventional	Slatted floor	Centrifuge	Artificial film	Stockpiling	Surface spreading

[a] separation efficiencies of N and DM (i.e. C), expressed as % in solid fraction relative to raw slurry, were 10% and 25% for screw press, and 30% and 60% for centrifuge (after Hjorth *et al.*, 2010).

[b] natural surface crust is assumed to be generated when dietary CP level was reduced, because of increased proportion of DM-rich faeces in slurry with lowering dietary CP. The effect of reducing dietary CP on animal N excreta and emissions from manure are considered, but its effect on CH_4 emissions from animal body is not considered in the scenario analysis.

[c] separated liquid fractions were injected into soil and solid fractions were immediately incorporated into soil after land spreading.

4.3 Results

4.3.1 Emissions from housing – effects of dietary CP manipulation and housing constructions

A total of 86 observations (37 from field-based experiments) were collected to analyse the effects of changes in crude protein (CP) content of the animal feed on NH_3 emissions from animal excretion in stables (i.e. from housing and indoor manure storage). The NH_3 emission decreased by 24 to 65% with a lowering of the CP content. The largest decrease was derived at a CP reduction of > 4% (Figure 4-1). There was a significant linear relationship between CP content in the diet vs manure pH, manure N content, amount of urine-N excreted and amount of total N excreted (Figure 4-2). The slopes of the two linear models (Figures 4-2a and b) show that one percent decrease in dietary CP content decreased the pH of the manure by on average 1.1% and the TN content of manure by 3.5%.

Data derived from 11 farm-scale studies were used to test the effects of animal houses with distinct floor constructions on emissions. Animal houses with alternative floors tended to have lower NH_3 emissions compared to the reference floor construction (Figure 4-1). The difference between slatted-floor/deep litter stables compared with solid-floor stables was statistically significant ($P < 0.01$). Buildings with slurries underneath slatted floor tended to have higher CH_4 emissions, but significantly lower N_2O emissions than buildings with deep-litter (Figure 4-3)

4.3.2 Emissions from slurry storages – effects of acidification and covers

Side-by-side comparisons were extracted from five studies in which the acidification-induced effects on emissions of NH_3 and/or CH_4 from stored slurries were estimated. One study was carried out at pilot scale and the other four in the laboratory; four studies used sulphuric acid (H_2SO_4) and one study hydrochloric acid (HCl). Statistically significant decreases in NH_3 emissions from acidified slurries (pH=4.5-6.5) were observed; the mean emission reduction was 83% and the 95% CIs ranged from 60% to 90% (Figure 4-1). Slurry acidification also led to a statistically significant reduction (on average, 87%) of CH_4 emissions during slurry storage (Figure 4-3).

A total of 12 studies compared the effects of covers on the emission of NH_3 from slurry storage, 10 of which were run at pilot or farm scales. Slurry storages covered by chopped straw, granules, artificial film, peat or oil had significant ($P < 0.01$) less NH_3 emissions compared to storages without such covers (Figure 4-1). In addition, the presence of a natural surface crust tended to lower emissions by 65% (14-89%) compared to storages without crust. Yet, the difference was statistically insignificant at the 0.05 level.

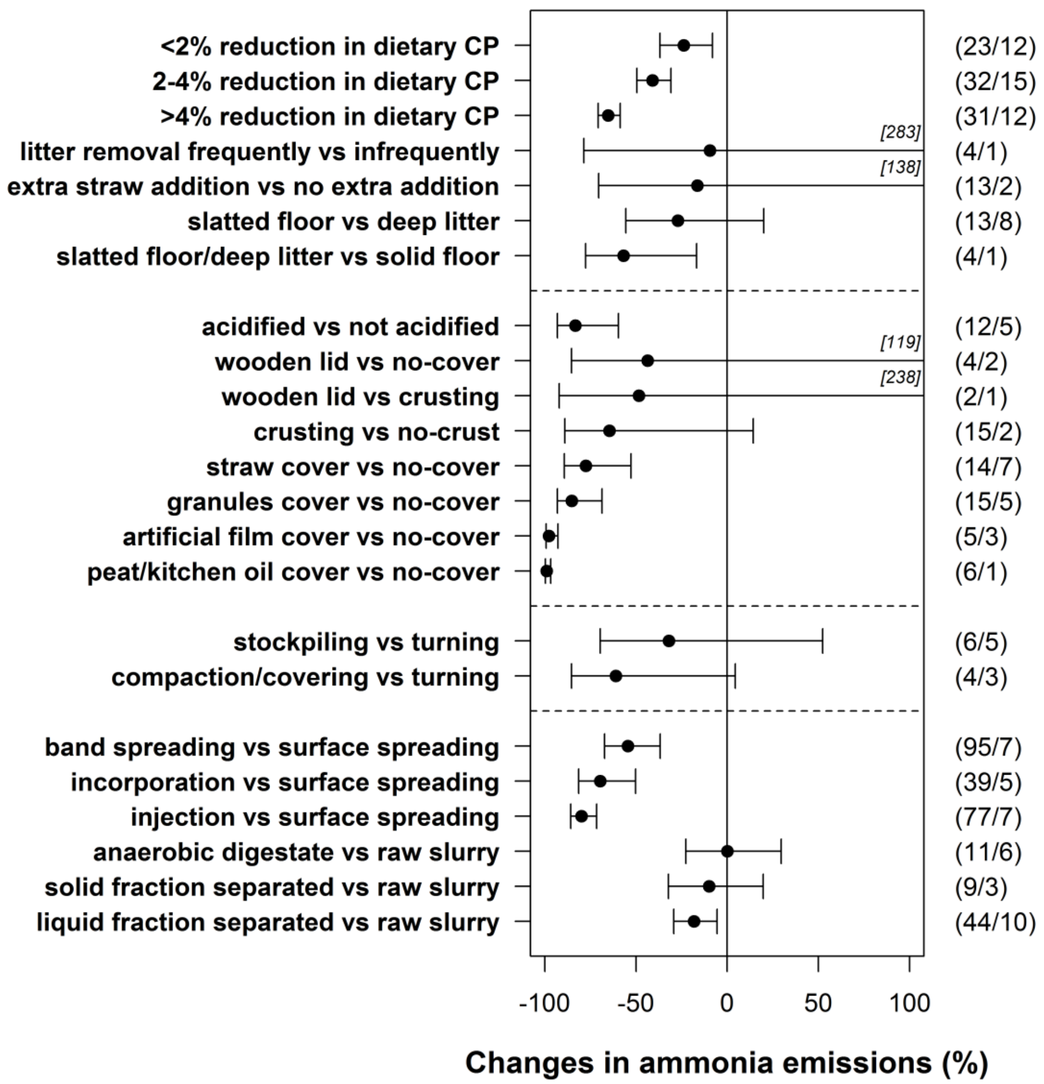

Figure 4-1. The mean change in NH_3 emissions as a percentage of the reference treatment, for a number of grouped side-by-side comparisons between treatments with vs without mitigation measures. Points show means of treatments, bars show 95% confidence intervals. The maximum values that exceed the scale of x-axis are shown in brackets. Numbers in the parentheses indicate the number of observations on which the statistical analysis was based, and the number of different studies from which the observations originated..

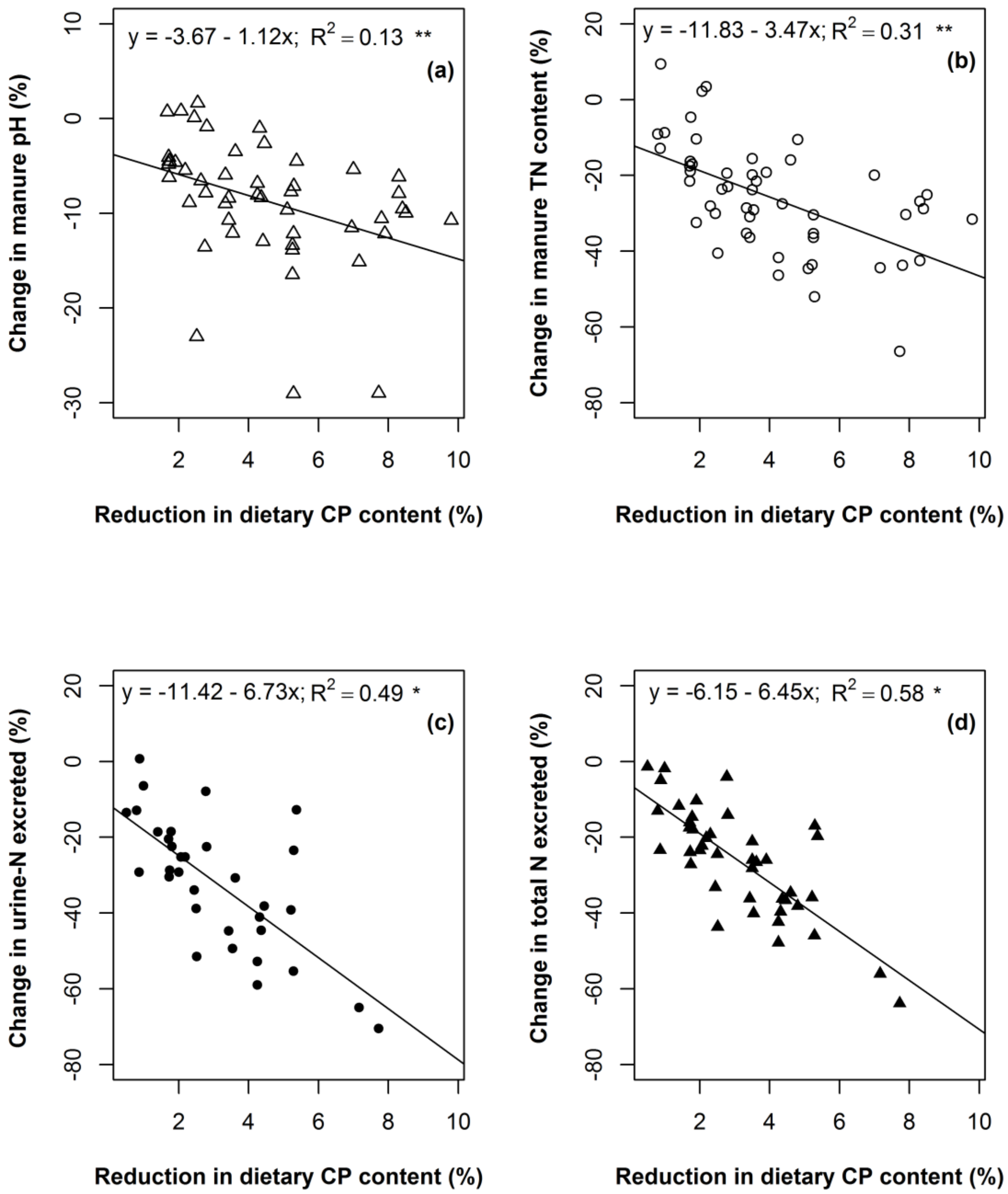

Figure 4-2. Relation between reduction in dietary CP content and change in manure pH (a), total N content of manure (b), urine-N excreted (c) and total (faeces and urine) N excreted (d), expressed as a percentage of the reference treatment. A CP reduction of 1 % equals to 10 g CP per kg DM-based feed. Significance asterisks ** = P <0.01, * = P <0.05.

A total of 9 studies (including 8 field-scale studies) were available to quantify the effect sizes of covers on CH_4 and/or N_2O emissions from slurry storage. Methane emissions were slightly suppressed by artificial film covers, but the effect was not statistically significant (Figure 4-3;

$P > 0.05$). Similarly, for all other grouped side-by-side comparisons, the changes in CH_4 emissions showed negative mean values. These effects were not statistically significant. Emissions of N_2O were enhanced by a factor of 8.6 when stored slurry was covered by chopped straw ($P < 0.01$). By contrast, slurry covered with artificial film decreased N_2O emissions by 98% ($P < 0.01$).

4.3.3 Emissions from solid manure storages – effects of compaction, stockpiling and covers

Five studies conducted at pilot scales were available to compare side-by-side NH_3 emissions from solid manure storages subjected to different treatments. Emissions of NH_3 from compacted or covered manure heaps were on average 61% lower in comparison to manure heaps subjected to composting (i.e. turning over manure heaps regularly; Figure 4-1). Yet, these differences were not statistically significant ($P > 0.05$). Static piling of manure (i.e. without turnover or disturbance) yielded lower emissions compared to manure heaps subjected to composting ($P > 0.05$; Figure 4-1). Apart from the meta-analysis, a number of additional studies without side-by-side comparisons were included in our database in order to maximize the number of studies for which NH_3 emissions were recorded. Based on all studies reviewed, the median NH_3 emission factor (% of TN) for composting was 39% (Table 4-3). Much lower emission factors were observed for static pilings (9%), covered heaps (9%) and compacted heaps (12%).

A total of 6 pilot-scale studies were available for the analysis of the effect size of manure heap management on CH_4 and N_2O emissions. As illustrated in Figure 4-3, storage of solid manure subjected to compaction, covering or stockpiling had larger CH_4 emissions than manure heaps subjected to composing; this difference was not significant. Compared to composting, two out of four studies reported significant higher CH_4 emissions from stockpiling, whereas only one study found significant lower CH_4 emissions from stockpiling. The effect sizes of these manure heap management practices on N_2O emissions were found to be inconsistent and not statistically significant.

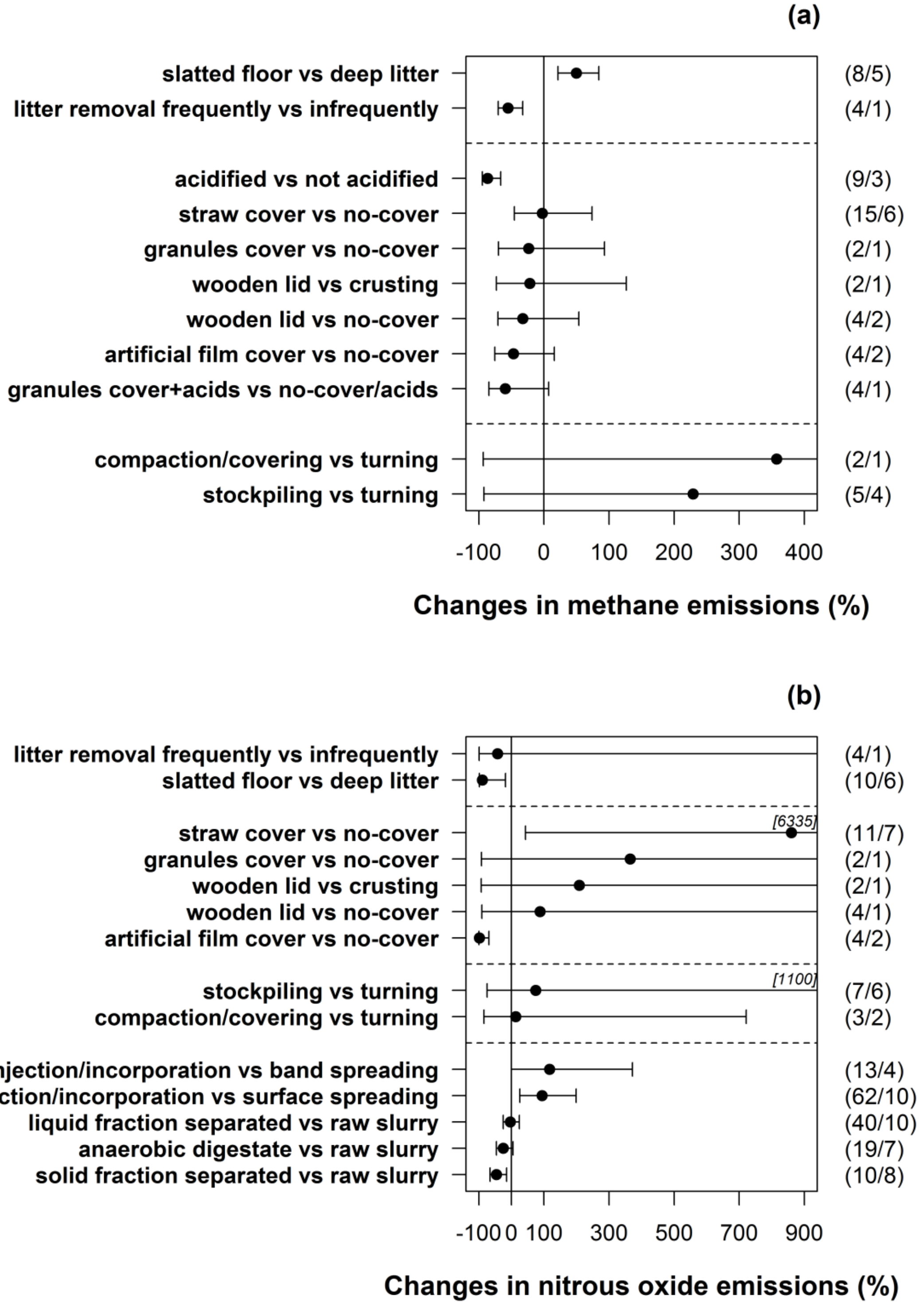

Figure 4-3. The mean change in methane (a) and nitrous oxide (b) emissions as a percentage of the reference treatment, for a number of grouped side-by-side comparisons between treatments with vs without mitigation measures. Points show means of treatments, bars show 95% confidence intervals. The maximum values that exceed the scale of x-axis are shown in brackets; some large variances (4-5 orders of magnitude) derived from limited studies are not shown. Numbers in the parentheses indicate the number of observations on which the statistical analysis was based, and the number of different studies from which the observations originated.

Table 4-3. Emission factors of ammonia from storage of solid manure and field application of manure.

Management stages	Management practices	Ammonia emission factors		
		Unit	Median [a]	Range [b]
Solid manure storage	Composting	% of total N into storage	38.9 (26/8)	17.3 - 45.3
	Stockpiling		9.0 (15/9)	1.8 - 23.5
	Compaction		12.4 (4/3)	1.1 - 17.4
	Covering		9.0 (4/4)	8.3 - 15.2
Field application	Surface spreading	% of total ammoniacal N applied	47.7 (229/20)	30.0 - 68.1
	Band spreading		20.9 (100/11)	13.5 - 31.5
	Incorporation		21.0 (39/5)	8.0 - 35.5
	Injection		11.1 (106/11)	4.9 - 21.2

[a] Numbers in the parentheses indicate the number of observations/studies; Data were derived from the references shown in Table1 and some additional studies referring storage of solid manure (Martins & Dewes, 1992; Petersen *et al.*, 1998; Sommer, 2001; Parkinson, 2004; Chadwick, 2005; Paillat *et al.*, 2005; Hansen *et al.*, 2006) and field application (Sommer & Olesen, 1991; Menzi *et al.*, 1998; Hansen *et al.*, 2003; Amon *et al.*, 2006; Chantigny *et al.*, 2007; Balsari *et al.*, 2008; Moal *et al.*, 2009; Dinuccio *et al.*, 2011, 2012; Huijsmans & Hol, 2011; Nyord *et al.*, 2012); These additional studies are not included in this meta-analysis.
[b] Range from first quartile to third quartile

4.3.4 Emissions after field application – effects of application methods and of processed manure products

Side-by-side comparisons of NH_3 emissions from manure following different application approaches were analysed based on results exclusively from field experiments. The emissions of NH_3 from manures following band spreading, incorporation and injection were 55% (range: 37-67%), 70% (50-82%) and 80% (72-86%) lower than that from surface broadcasted manures, respectively (Figure 4-1). These differences were statistically significant (P <0.01). Based on the reviewed studies, the median NH_3 emission factor (in % of TAN applied) for surface broadcasted manure was 48%, compared to the emission factors of 21%, 21% and 11% for band spread, incorporated and injected manure, respectively (Table 4-3). A total of 75 observations (69 from field experiments) were available to analyse the effect sizes of application technologies on N_2O emissions. Statistically higher emissions of N_2O (98%) were found for injection/incorporation of manure compared with surface broadcasted manure (P < 0.01) (Figure 4-3b).

Emissions of NH_3 were not significantly different between digestates and raw slurry following field application (Figure 4-1; P > 0.05). Significantly lower NH_3 emissions (18%) were found for separated liquid fraction (LFs) relative to raw slurry (P < 0.01; Figure 4-1), based on 44 observations (41 from field experiments). The overall effect of LFs on N_2O emissions did not differ from that of raw slurry (Figure 4-3b). Field-applied digestates and solid fractions showed on average 25% (P > 0.05) and 46% (P <0.01) lower N_2O emissions than field-applied untreated manure, respectively (Figure 4-3b).

4.3.5 Emissions from manure management chains – effects of scenario analysis

In the reference slurry-based system, emissions of NH_3 from animal housing, external slurry storages and slurry application to the field amounted to 43%, 22% and 35% of the total emissions, respectively (Figure 4-4a). The assessment of single mitigation measures showed that the total NH_3 emission was lowest (35% of the reference) for the acidification scenario, followed by the low dietary CP scenario (65% of the reference) and the slurry injection scenario (71% of the reference). Covering slurry storages with straw or an artificial film decreased NH_3 emissions during storage, but increased emissions following land application (Figure 4-4a). The largest decrease in NH_3 emissions was obtained when mitigation measures were combined (Figure 4-4a). Covering slurry storage with straw did not change the overall GHG emissions from the management chain (Figure 4-4b); the increased direct N_2O emissions were offset by decreased indirect N_2O and CH_4 emissions during storage. The "injection" scenario showed a higher total GHG emissions than the reference scenario (by 16%). Lowering dietary CP content decreased the whole-chain GHG emissions by 5%. Use of an artificial film cover reduced GHG emissions with 24%. The largest decrease (up to 50%) was shown in the scenarios with slurry acidification (Figure 4-4b).

In the reference solid manure-based system, emissions of NH_3 from housing, outdoor storages and following field application contributed to 38%, 25% and 37% of the total NH_3 emissions (Figure 4-5a). Reducing dietary CP content decreased the NH_3 emissions from the whole solid manure system by 29%. Emissions of NH_3 following the field application of stockpiled and compacted manures were higher than in the reference situation, suggesting that NH_3-N trapped by low-emission techniques during storage escaped at subsequent stages (i.e. during field application). Lowering dietary CP content of the animal feed and direct incorporation of manure into the soil decreased NH_3 emissions from stockpiled and compacted manures (Figure 4-5a). Direct N_2O emissions were relatively large in the solid manure-based systems (Figure 4-5b). The "Diet" scenario was the only scenario that had lower GHG emissions than the reference (by 15%), due to reduction in both direct and indirect N_2O emissions. The GHG emissions in scenarios with stockpiling, compaction or manure incorporation were higher than that in the reference system. The highest GHG emissions were found for combined stockpiling and incorporation (Figure 4-5b).

Introducing separation techniques in slurry-based systems did not affect total emissions of NH_3 and GHG much (Figure 4-6). This is due to the fact that decreased CH_4 emissions were counterbalanced by increased N_2O emissions. Centrifugation-based systems tended to have higher GHG emissions compared to systems with screw press (Figure 4-6b), which is related to differences in separation efficiencies. Emissions of GHG were reduced by a combination of low dietary CP level (7-10% reduction) and covering storages of LFs (6-20%).

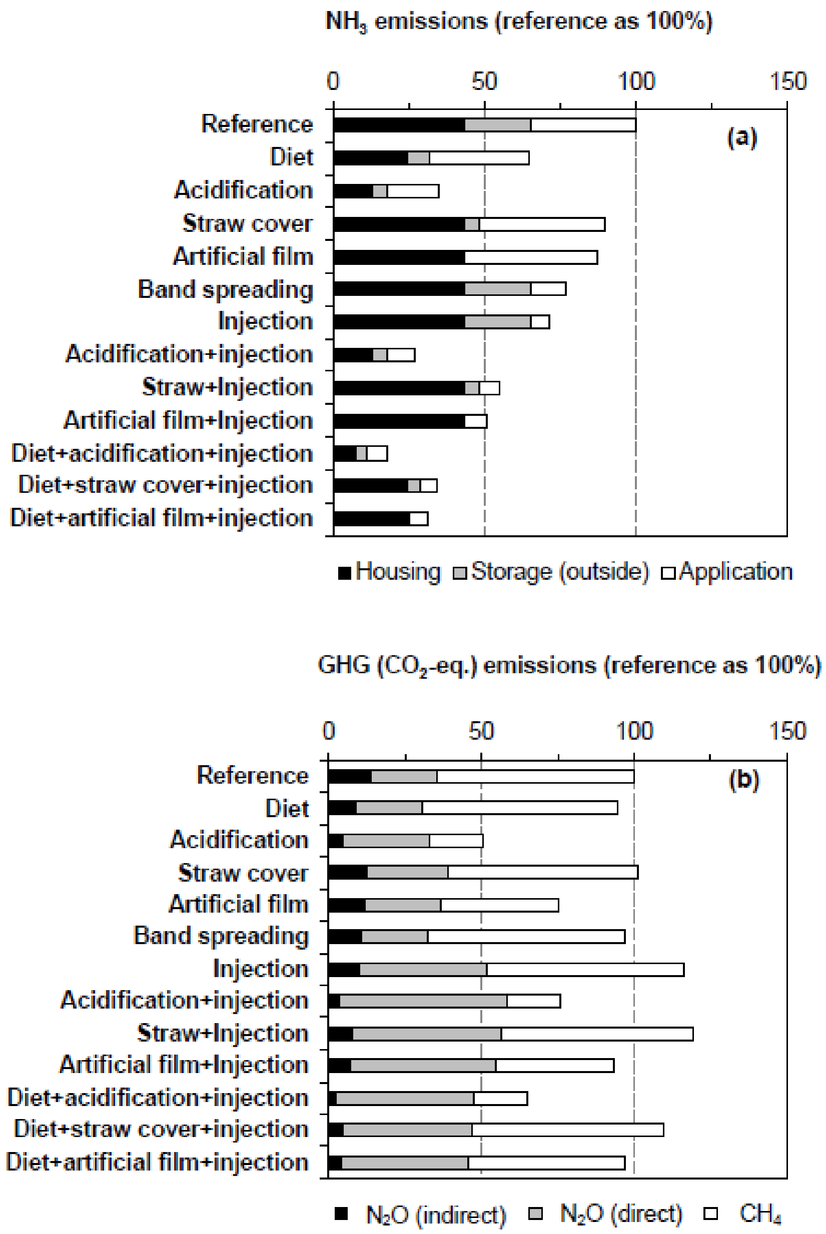

Figure 4-4. Impacts of mitigation measures on NH_3 (a) and GHG (b) emissions from slurry-based systems, expressed as percentage of the reference system. See Table 4-2 for description of scenarios.

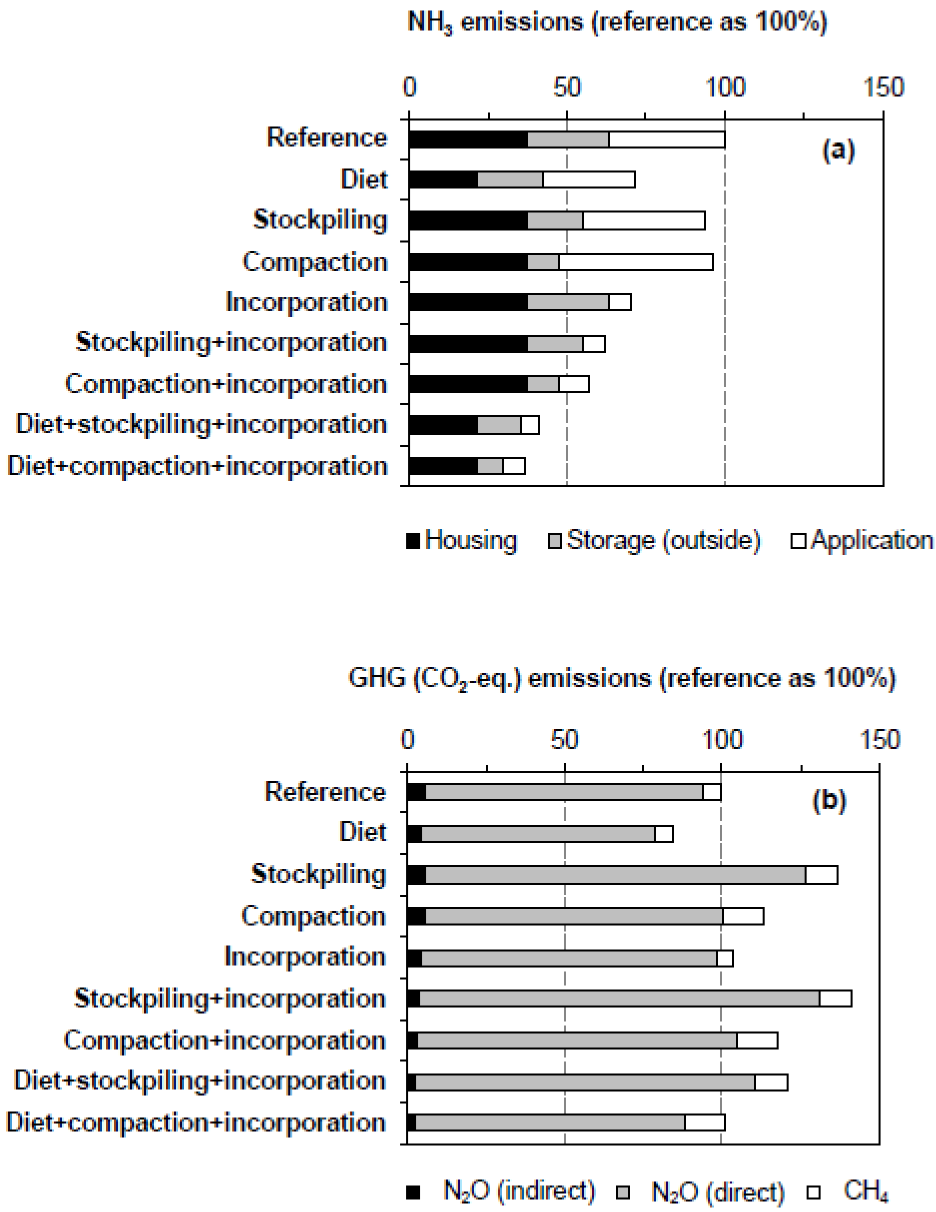

Figure 4-5. Impacts of mitigation measures on NH_3 (a) and GHG (b) emissions from solid manure-based systems, expressed as percentage of the reference system. See Table 4-2 for description of scenarios.

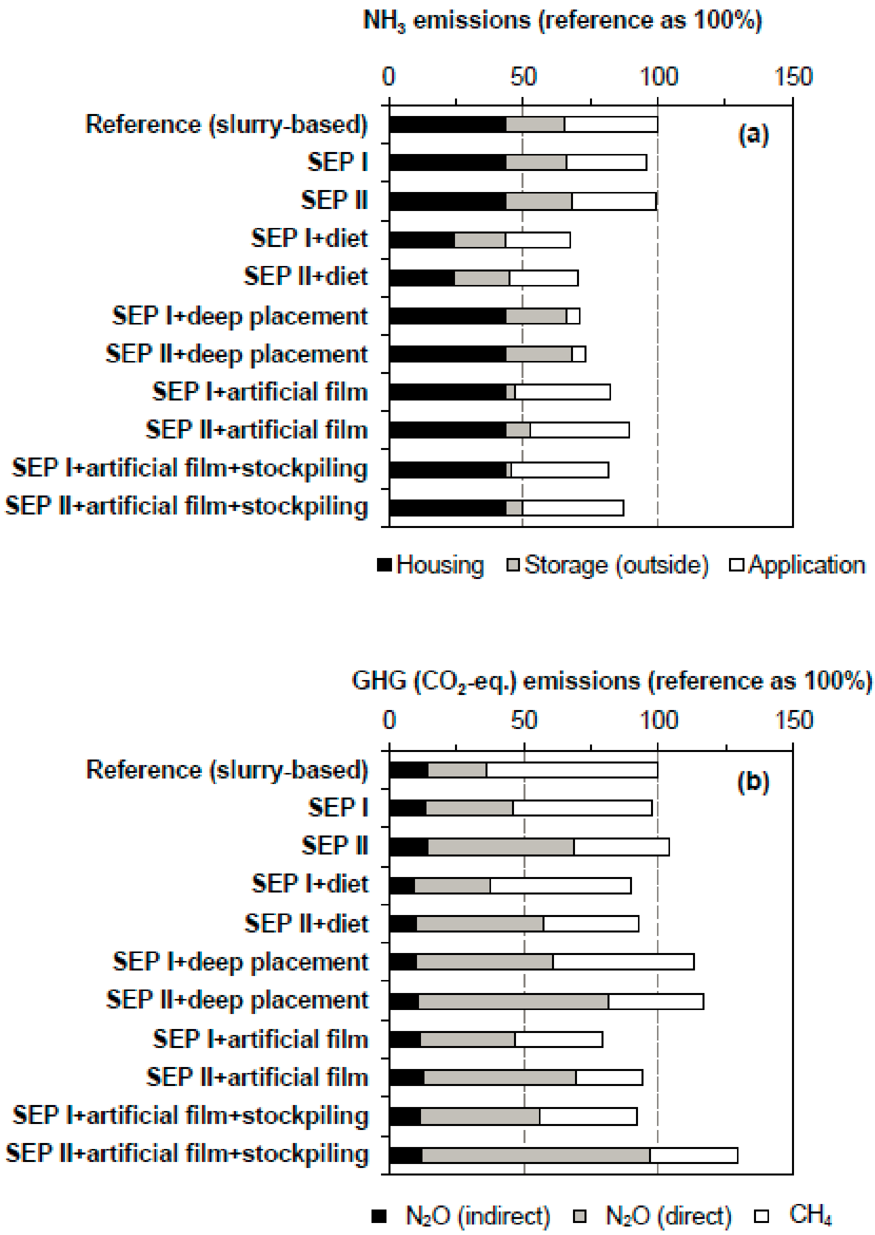

Figure 4-6. Impacts of mitigation measures on NH_3 (a) and GHG (b) emissions from slurry separation-inclusive systems, expressed as percentage of the reference (i.e. slurry-based) system. SEP I and SEP II respectively indicate screw press and centrifuge separation. See Table 4-2 for description of scenarios.

4.4 Discussion

4.4.1 Main findings

This study quantitatively assessed the impacts of NH_3 mitigation measures on emissions of NH_3, N_2O and CH_4 at individual stages of the manure management chain by means of a meta-analysis of 126 published studies. In addition, an integrated assessment of the impacts of packages of measures on NH_3 and GHG emissions from the whole management chains was carried out using various scenarios for slurry-based and solid manure-based systems. The main findings of our study are as follows: (i) lowering the CP content in feed significantly decreased NH_3 emissions at the housing stage (Figure 4-1) and also decreased the total GHG emissions from manure chains (Figures. 4-4,5,6), (ii) slurry acidification significantly decreased emissions of NH_3 and CH_4 from slurry storages (Figure 4-3), which leads to decreases in total GHG emissions from systems with acidified slurry (Figure 4-4b), (iii) covering slurry storages with straw significantly decreased NH_3 emissions (Figure 4-1) and increased N_2O emissions (Figure 4-3b), but the effects on total GHG emissions from the manure chains were relatively small (Figure 4-4b), (iv) stockpiling tended to decrease NH_3 emissions (Figure 4-1), yet might enhance emissions of N_2O and CH_4 during storage (Figure 4-3) and total GHG emissions (Figure 4-5b), (v) injection or direct incorporation of manure into soil significantly decreased NH_3 emissions (Figure 4-1), but significantly increased N_2O emissions (Figure 4-3b) and total GHG emissions from the manure management chain (Figures 4-6), (vi) the packages of NH_3 mitigation measures were effective in NH_3 emission mitigation, but had only minor impacts on GHG emissions, with the exception of acidification and stockpiling, and (vii) the manure N that is conserved by using mitigation measures can be used as crop available N, when low-emission field application techniques are applied.

The results collected did not allow to compare management options across animal species (e.g. pigs vs cattle). However, sufficient data were available for analysing the effects of lowering the CP content of feed for different animal species on NH_3 emissions during housing and following slurry injection. The results indicate that the differences between animal species in emission reduction were not significant ($P > 0.05$; data not shown). Data from both field-and laboratory-scale studies were included in our database as data solely from field-scale studies were insufficient. All data used in the meta-analysis is available as supporting information (Table S3). This database is expected to be updated in the future when new data especially from field-scale studies becomes available.

4.4.2 Dietary CP manipulation

Clearly, lowering CP content in animal feed is an effective strategy to reduce NH_3 emissions in the entire management chain (Figures 4-4,5,6). Lowering dietary CP deceases the N content of animal excreta. This is more pronounced for urine than for faeces (Figure 4-2). Urinary N is mainly in the form of urea and is easily converted into NH_3 by the enzyme urease present in faeces (Smits *et al.*, 1995; Canh *et al.*, 1998a; Misselbrook *et al*., 2005a; Galassi *et al.*, 2010). Lowering feed CP also decreases the pH of the manure (Figure 4-2a), which decreases the risk of NH_3 emissions. This decrease in pH is likely related to a decreased dietary electrolyte balance, to changes in the concentration of volatile fatty acids (VFA), and to a decreased acetic acid/ propionic acid ratio in the manure (Canh *et al.*, 1998b; Leek *et al.*, 2005; Velthof *et al.,* 2005; Hernández *et al.*, 2011). Caution must be taken to maintain animal performance when dietary CP manipulation is implemented (Frank & Swensson, 2002; Panetta *et al.*, 2006). When necessary, essential amino acids should be supplemented in accordance with animal requirements. The perspectives for lowering dietary CP also depend on the current diet (Bittman *et al*., 2014). According to our database, a 0.5-2% reduction in dietary CP was implemented for diets with a mean CP content of 14.4 % (on a DM-basis), while a 4% reduction was implemented for diets with a mean CP content of 18.1%.

The effects of dietary CP manipulation on CH_4 and N_2O emissions from animal excreta in housings were not quantitatively evaluated in the meta-analysis, as there were insufficient results. Montalvo *et al.* (2013) showed that reducing CP contents of pig diets decreased CH_4 emissions from manure, but did not significantly affect N_2O emissions. This is in agreement with the results of Velthof *et al.* (2005), who showed that the emission of CH_4 had significant correlations with the contents of dry matter, total C and VFAs in the manure. However, Le *et al.* (2009) and Lee *et al.* (2012) reported no significant impacts on CH_4 and N_2O emissions when low CP diets were used. Clark *et al.* (2005) found increased emissions of CH_4 from manure with decreasing dietary CP contents, presumably due to a slight reduction in digestibility of fibre that is the main substrate for methanogenesis with low CP contents (Külling *et al.*, 2001). The impacts of reducing dietary CP contents on enteric CH_4 emissions were reviewed recently by Dijkstra *et al.* (2011). They found no consistent effect of lowering dietary CP content on enteric CH_4 emission. The scenario analysis showed that lowering the CP content in feed decreased NH_3 and N_2O emissions from the whole management chains (Figures 4-4,5,6). This decrease is partly due to less N in manure and partly also because of a decrease in the pH of the manure. Decreasing the dietary CP content turned out to be a most promising strategy for abating N-based emissions.

4.4.3 Housing structure

The floor construction in animal houses has a significant effect on NH_3 emissions from indoor-stored manure (Groot Koerkamp *et al.*, 1998), partly because floor types might cause differences in micro-condition of manure storage. Animal houses with slatted floors or deep litter showed lower emissions than traditional houses with solid (concrete) floors (Figure 4-1). This is because urine as main source of NH_3 can be drained off quickly via openings of slatted floor into the relatively cool underground storage with low airflow or can be absorbed by deep straw (Kavolelis, 2006). Livestock housing with V-shaped solid floors with a gutter at the bottom of the V-groove to drain urine have low NH_3 emissions (Swierstra *et al.*, 1995; Braam *et al.*, 1997). The effectiveness of housing structure and management such as use of bedding on NH_3 mitigation may vary depending on environmental conditions. Significant relationships between NH_3 emissions and various climatic factors, such as outside temperature and wind speed, have been frequently reported for naturally ventilated dairy housings and pig buildings (Ivanova-Peneva *et al.*, 2008; Pereira *et al.*, 2010a; Schrade *et al.*, 2012; Wu *et al.*, 2012; Van Ransbeeck *et al.*, 2013; Saha *et al.*, 2014).

The emissions of N_2O from a deep-litter house were significantly higher compared to those from a slatted-floor house (Figure 4-3b). The formation of N_2O occurs during nitrification and denitrification processes. Nitrification requires aerobic conditions and denitrification requires anaerobic conditions. Both conditions can be found in deep-litter but not in slurries underneath slatted floors (Cabaraux *et al.*, 2009). Emissions of N_2O from housings without bedding materials are often hard to detect (Ngwabie *et al.*, 2009, 2011; Wu *et al.*, 2012; Saha *et al.*, 2014). More attention should be given to deep-litter induced N_2O emissions, particularly in the case of countries where adding bedding materials in housings is increasingly considered as a means to improve animal welfare. Conversely to N_2O, CH_4 emissions were significantly higher in housings with slatted floors and underground slurry stores than in housings with deep litter (Figure 4-3a). The anaerobic conditions and the moderate pH level of slurry under slatted floor favour CH_4 production. A high straw – manure ratio inhibits CH_4 production because of a greater manure aeration (Philippe *et al.*, 2011).

4.4.4 Acidification

Slurry acidification has been approved as a Best Available Technology (BAT) and is widely adopted in Denmark now (Petersen *et al.*, 2012). The slurry is acidified in a process tank by controlled addition of sulphuric acid to a pH of about 5.5. The acidified slurry is pumped back to the slurry pit in the animal house or to outdoor storage tanks (Petersen *et al.*, 2012). Such acidification technology has potential to reduce NH_3 emission from housing, storage and field application (Kai *et al.*, 2008). The pH of the slurry is a key factor affecting NH_3 mitigation potential. The effectiveness of acidification to mitigate NH_3 emissions is significantly weakened when the pH of the acidified slurry goes up to a level of 6.0-6.5 (Dai & Blanes-Vidal, 2013; Petersen *et al.*, 2014; Wang *et al.*, 2014). Therefore, a pH of 6.0 is recommended as the upper limit for a successful emission abatement. There is limited information on emissions after land application of acidified slurry. Recent field experiments have shown that acidification results in a long-term inhibition of CH_4 emissions from slurry storages (Petersen *et al.*, 2014). The reduction in CH_4 emissions could be attributed to inhibited methanogenesis because of the acidic conditions and high sulphide concentrations (Ottosen *et al.*, 2009; Petersen *et al.*, 2012). High sulphide concentrations may impair indoor air quality and animal performance. Because emissions of both NH_3 and CH_4 were decreased by acidification, the overall GHG emissions from acidified slurry were low (Figure 4-4b). Acidification of slurry to $\leq$ 6.0 is therefore a possible technique to minimizing GHG emissions from slurries.

4.4.5 Natural surface crust and artificial covers

Formation of a natural surface crust on stored manure is known to be effective in reducing NH_3 emissions during slurry storage. Crusting is enhanced by gasification, i.e., the release of gases, including CO_2 and CH_4 during anaerobic storage. During this process bubbles tend to combine around fibre particles, helping to raise floating particles to the surface (Smith *et al.*, 2007). A limiting factor of generating a slurry crust is the dry matter content in slurry, and no crust is formed when the dry matter content is less than 1% (Misselbrook *et al.*, 2005a). The dry matter content is also influenced by diet ingredients and manure treatment such as slurry separation (Misselbrook *et al.*, 2005a; Dinuccio *et al.*, 2008; Fangueiro *et al.*, 2008b). The ammonia mitigation potential is larger with artificial covers than with natural crusts (Figure 4-1). Artificial covers minimize the emitting surface and may adsorb ammonia (Hörnig *et al.*, 1999; Portejoie *et al.*, 2003). The effects depend on environmental conditions (Sommer, 1997). Emissions of NH_3 were higher in summertime than in wintertime, and such temperature effect seems to be more pronounced for slurry without crust and cover (Balsari *et al.*, 2006; Clemens *et al.*, 2006; Petersen *et al.*, 2013).

The published impacts of artificial covers and natural crusts on CH_4 emissions during slurry storage are inconsistent (Figure 4-3a). Husted (1994) indicated that a natural surface crust reduced CH_4 emission from slurry by a factor of 11-12, although this effect decreased with increasing slurry temperature. A 38% decrease of CH_4 emissions was reported by Sommer *et al*. (2000) when slurry was covered by a natural surface crust, or straw or floating Leca® (lightweight expanded clay aggregate). This reduction in CH_4 emissions was related to the oxidation of CH_4 in the surface crusts (Petersen *et al.*, 2005; Petersen & Ambus, 2006). However, Clemens *et al.* (2006) and Petersen *et al.* (2013) did not find a significant influence on CH_4 emissions by covering slurry with straw. Berg *et al.* (2006) reported relatively high emission from slurry covered with straw, perlite or Leca® compared to uncovered slurry. Methane emissions are known to be highly dependent on ambient air temperature or slurry temperature (Husted, 1994; Sommer *et al.*, 2000, 2007), which results in seasonal variations in CH_4 emissions. The magnitude of such seasonal variations are dependent on many factors including manure characteristics, environmental conditions, and management practices (Clemens *et al.*, 2006; Rodhe *et al.*, 2012; Petersen *et al.*, 2013).

Straw covers significantly increased emissions of N_2O from slurry storages (Figure 4-3b). This may be due to an interface between the slurry and the air-filled surface covers, which favours both nitrification and denitrification (Sommer *et al.*, 2000; Petersen *et al.*, 2013). However, there were no significant differences in the overall GHG emissions between systems with or without straw covers (Figure 4-4b), because of the reduction of NH_3 emissions and associated indirect N_2O emissions. The variation in straw cover-induced N_2O emissions was large (Figure 4-3b), which is partly related to variations in environmental conditions. There is evidence that increases in N_2O emissions from slurry storage with covers (especially, surface crust and chopped straw) are more pronounced in summer than in winter (Sommer *et al.*, 2000; Berg *et al.*, 2006; Rodhe *et al.*, 2012; Petersen *et al.*, 2013).

4.4.6 Stockpiling, compaction and covering

Composting is considered as a good measure for recycling manure because it produces a stabilised and sanitised end-product for agriculture, with relatively low transport costs because of reduced volume and mass (Bernal *et al.*, 2009). However, losses of N during composting can be high, especially via NH_3 emissions (Figure 4-1) and are affected by the frequency of turning the manure heap (Sommer & Dahl, 1999; Parkinson, 2004)(Martins & Dewes, 1992). Methane emissions can be enhanced under anaerobic conditions in covered, compacted or static heaps compared to heaps subjected to turning (Figure 4-3a). However, CH_4 emissions from heaps of solid manure with these alternative management measures are also (very) low, in case air temperature is relatively low (e.g. in winter) (Amon *et al.*, 2001; Chadwick, 2005; Hansen *et al.*, 2006). This indicates that measures to mitigate CH_4 emissions from manure heaps should be especially implemented during warm seasons. Increases in N_2O emissions during stockpiling (Figure 4-3b) increased GHG emissions from solid manure-based systems (Figure 4-5b). However, the mechanism behind the stockpiling-induced increase in N_2O emission has not been well-studied and therefore requires further research (McGinn & Sommer, 2007; Sagoo *et al.*, 2007; Shah *et al.*, 2012).

4.4.7 Low-NH_3-emission application technologies

The calculated NH_3 reduction efficiencies of low-emission field application techniques are well in agreement with the findings of Webb *et al.* (2010). Slurry injection has the highest NH_3 mitigation potential (Figure 4-1). However, a side-effect of slurry injection is increased N_2O emissions (Figure 4-3b). Such increases in N_2O emissions are likely to increase the overall GHG emissions of the whole system when no combined mitigation measures (such as dietary CP manipulation and acidification) are applied (Figure 4-4b). The calculated effects of slurry injection or incorporation into the soil on N_2O emissions were highly variable (Figure 4-3b) and related to many factors, e.g. types and application rates of manure, soil properties (e.g. soil texture and moisture content), vegetation and climate (Velthof *et al.*, 2003). For example, Velthof & Mosquera (2011) reported higher N_2O emissions from injected manures than from surface-applied manures, but the effects were variable due to the large variations in soil moisture and rainfall. Webb *et al.* (2014) concluded that the impacts of immediate incorporation on N_2O emissions may be related to soil type, with a greater possibility of emission increases on coarse sandy soils.

4.4.8 Field application of processed manure

The effect-size analysis of 44 side-by-side comparisons indicated that the liquid fraction of separated slurry had a significantly smaller NH_3 emissions following application to land than untreated slurry (Figure 4-1). Most comparisons were made on manures applied via surface broadcasting (36 out of these comparisons). The lower NH_3 emissions have been attributed to the low dry matter content in the liquid fraction that allows rapid infiltration of manure into soil (Bhandral *et al.*, 2009; Dinuccio *et al.*, 2012; Nyord *et al.*, 2012). Infiltration into the soil reduces NH_3 losses from surface applied slurry, partly because the rate of diffusion of NH_3 in the soil is relatively low, and the sorption of NH_4^+ to soil colloids reduces the concentration of ammonium in the soil solution (Sommer & Hutchings, 2001). The dry matter content in the liquid fraction may differ greatly between separation technologies and also because of the dry matter content in original slurry (Hjorth *et al.*, 2010). Inefficient removal of solids from raw slurry may decrease the infiltration rate of the liquid fraction, which decreases the potential of lowering NH_3 emissions from liquid fractions (Mattila *et al.*, 2003; Dinuccio *et al.*, 2011).

No significant difference in N_2O emissions was observed between liquid factions and raw slurry following land application. Conversely, solid fractions have shown lower N_2O emissions than raw slurry in terms of percentage of total N applied (Figure 4-3b). As the solid fraction contains less available N (particularly NH_4^+) than raw slurry, there is likely to be less nitrification and, by that, less emission of N_2O (Fangueiro *et al.*, 2007; Bertora *et al.*, 2008). Slurry separation did not considerably influence the total GHG (CO_2-eq.) emissions from the whole management chain (Figure 4-6b). Field application of digested substrates tended to have lower N_2O emissions compared to raw slurry (Figure 4-3b). This is in agreement with the review of Nkoa (2013) in which the environmental risk of soil application of anaerobic digestates has been assessed. It has been hypothesized that the lower N_2O emissions from digestates are the consequence of less easily degradable C, hence less energy source for denitrifiers (Rochette *et al.*, 2000; Vallejo *et al.*, 2006).

4.4.9 The manure management chain

Our study provides a quantitative analysis of the effect sizes of a range of key manure management technologies on NH_3, N_2O and CH_4 emissions from the manure management chain. Lowering the dietary crude protein (CP) content of the animal feed and acidification of slurry in storages are management strategies that effectively reduced NH_3 emissions and GHG emissions from all subsequent stages of the manure management chain (including both indirect and direct N_2O emissions, and CH_4 emissions). Strategies aimed at decreasing NH_3 emissions from manure storages must be combined with low-NH_3-emission manure application techniques, to ensure that the NH_3 trapped during storage does not escape following field application. Slurry injection greatly decreases NH_3 emissions but enhances N_2O emissions. Joint adoption of slurry injection with a low dietary CP intake, and covering manure storages or acidification of slurry in storages greatly decreases the risk of pollution swapping. Compaction, static stockpiling and covering of solid manure tend to decrease NH_3 emissions and increase CH_4 and N_2O emissions compared to manure heaps subjected to composting. However, the number of observations underlying these latter effects is low. Slurry separation in liquid and solid fractions does not significantly affect emissions of NH_3 and GHG compared to a reference system without slurry separation. Given the possible synergistic and antagonistic effects, and the fact that mitigation measures are increasingly implemented in practice, selecting the proper combinations of measures becomes more important, in order to successfully minimize the whole-chain ammonia and GHG emissions.

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Supporting Information

Additional Supporting Information can be found in the online version of this article:
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CHAPTER 5

Nutrient recovery and emissions of ammonia, nitrous oxide and methane from animal manure in Europe: effects of manure treatment technologies

This Chapter has been submitted to Environmental Science & Technology (accepted subject to revision): Y. Hou, G.L. Velthof, J.P. Lesschen, I.G. Staritsky, O. Oenema

Abstract

Animal manure contributes considerably to ammonia (NH_3) and greenhouse gas (GHG) emissions in Europe. Various treatment technologies have been implemented to reduce emissions and to facilitate its use as fertilizer, but a systematic analysis of these technologies has not been carried out yet. Here, we present an integrated assessment of manure treatment effects on NH_3, nitrous oxide (N_2O) and methane (CH_4) emissions from manure management chains in EU-27 in 2010, using the model MITERRA-Europe. Whole-chain effects of implementing twelve treatment technologies on emissions and nutrient recovery were further explored through scenario analyses; the level of implementation corresponded to levels currently achieved by forerunner countries. Manure treatment decreased national GHG emissions from manures by 0-17% in 2010, with the largest contribution from anaerobic digestion; the effects on NH_3 emissions were small. Scenario analyses indicate that increased implementation of acidification, thermal drying, incineration and pyrolysis may decrease both NH_3 (9-11%) and GHG (11-18%) emissions; nitrification-denitrification treatment decreased NH_3 emissions, but increased GHG emissions. The nitrogen recovery (% of nitrogen excreted in housings that is applied to land) would increase from a mean of 57% (in 2010) to 61% by acidification, but would decrease to 48% by incineration. Promoting optimized manure treatment technologies can greatly contribute to achieving NH_3 and GHG emission targets of EU regulations.

5.1 Introduction

Animal manure is a main source of plant-available nutrients, but also a major emission source of ammonia (NH_3) and greenhouse gases (GHG) - nitrous oxide (N_2O) and methane (CH_4). Manure from animal production is responsible for about 40% of the global anthropogenic NH_3 and N_2O emissions (Galloway *et al.*, 2004; Oenema *et al.*, 2005). Approximately 35-40% of the global anthropogenic CH_4 emissions are associated with the livestock sector (about 6% from manure management and the remaining from enteric fermentation) (Steinfeld *et al.*, 2006). In Europe, animal manures contribute about 65% to the total anthropogenic NH_3 emissions, 40% to N_2O emissions and 10% to CH_4 emissions (Oenema *et al.*, 2007, 2014; EEA, 2014a, 2014b). Farm animals excreted 9.7 Tg N and 1.7 Tg P in the European Union (EU) in 2010, equivalent to about 95% and 160% of the total use of mineral N and P fertilizers (Eurostat; Hou *et al.*, 2016).

Emissions of NH_3, N_2O and CH_4 may occur simultaneously from different sources of manure management systems that typically include animal houses, manure storages, manure application to land and droppings from grazing animals in pastures. Introducing a management measure may have interactive effects on emissions of these gases from a specific source; it may also influence emissions downstream in the system and hence the nutrient recovery from the manure (Sommer *et al.*, 2009; Velthof *et al.*, 2009; Hou *et al.*, 2015). The whole manure management chain needs to be considered therefore, when assessing effects of measures on gaseous emissions and nutrient recycling and recovery.

Manure treatment technologies have been increasingly applied in practice in Europe during the last few decades, driven by the specialization/intensification of animal production and the tightened enforcement of EU environmental policies (Oenema *et al.*, 2011). Manure treatment creates management opportunities to better use the nutrients and organic matter in manure. Treatment may induce changes in physical, chemical and/or biological properties of the manure and hence influence emissions of NH_3 and GHG throughout the whole management chain. An inventory reported that 7.8% of manure production in the EU was processed in 2010, but with large variations among countries (range 0-35%) (Foged *et al.*, 2011). While the growth in implementation of manure treatment is in general lauded as an environmental success (Sommer *et al.*, 2013), there is a need for systematic environmental assessment of manure treatment, also to provide guidance for the further development of proper manure management strategies.

Only few studies have assessed losses of N and GHG emissions from the animal production systems at country and EU scales (Oenema *et al.*, 2009; Lesschen *et al.*, 2011; Weiss & Leip, 2012). Manure treatment techniques are usually not considered in these large-scale studies, and as a consequence their environmental impacts have not been systematically addressed. Yet, a large number of laboratory and pilot experiments have been carried out to analyze NH_3 and GHG emissions from processed manures, most of them typically focused on a specific gas/substance or emission source (Petersen *et al.*, 2007; Thangarajan *et al.*, 2013; Hou *et al.*, 2015). Whole-farm (or life cycle) assessments were mostly conducted on the basis of specific farm-scale characteristics. Manure management systems and the implementation of NH_3 abatement measures (e.g. low-emission stables, storage systems and application methods) vary greatly among farms and countries (UNFCCC; Asman *et al.*, 2011). These farm and country-specific contexts may also influence the performance of manure treatment technologies. There is as yet little information about the potential effects of manure treatment technologies in the EU. In particular, there is insufficient understanding of possible synergistic and antagonistic effects of manure treatment on emissions of NH_3, N_2O and CH_4 and on nutrient recovery.

The objectives of this study are to assess the contribution of manure treatment techniques to emissions of NH_3, N_2O and CH_4 at national level for the EU-27 in 2010, using the improved model MITERRA-Europe (Velthof *et al.*, 2009; Lesschen *et al.*, 2011). Further, we explored the whole-chain impact potentials of treatment technologies on gaseous emissions and nutrient recovery through scenario analyses. We also executed an uncertainty analysis.

5.2 Materials and methods

System boundary. The whole chain from 'animal excretion, in-house and outdoor manure storage, manure treatment, application of manure to land, and deposition of urine and faeces in pastures during grazing' in the EU-27 was considered in this study. The system boundary, the main flows of nutrients embodied in animal manures and the possible manure treatment techniques are illustrated in Figure 5-1, defined according to literature (Velthof *et al.*, 2009; Foged *et al.*, 2011; Lesschen *et al.*, 2011). Faeces and urine are deposited by grazing animals in pastures and produced by housed animals in housings. Excreta from housings are applied to agricultural land after a period of storage in either liquid or solid form, or in some cases are treated by certain technologies. The main treatment technologies currently applied in Europe are solid-liquid separation, anaerobic digestion, acidification, biological aerobic N removal (i.e. nitrification-denitrification), composting, (thermal or bio-) drying and incineration (Foged *et al.*, 2011). The use of treatment technologies depends on manure types (e.g. liquids, slurry and solid manure). Treated manure products are typically returned to soil, as fertilizers or soil amendments (Foged *et al.*, 2011; Sommer *et al.*, 2013). In addition, several NH_3 mitigation measures have been adopted alongside the manure management chain, but depending on country (Oenema *et al.*, 2009).

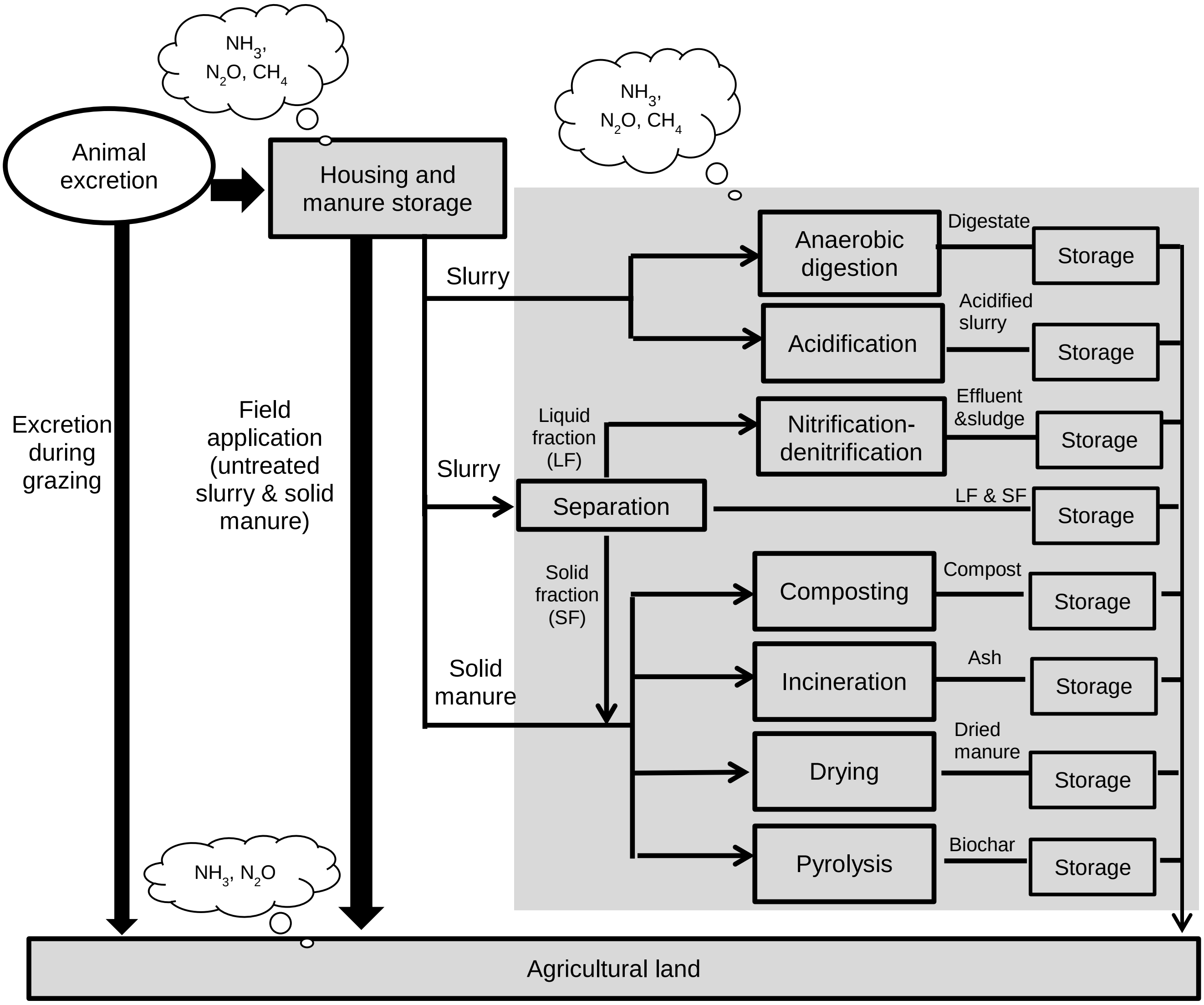

Figure 5-1. Schematic representation of the manure management chain with possible manure treatment technologies (highlighted in grey) discussed in this study. The arrows indicate the main flows of manure products. The clouds show emission sources of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) from animal manure.

Emission sources. Emissions of NH_3, N_2O and CH_4 from the manure management chain of the EU-27 were quantified at national scales in 2010. Animal categories include dairy cows, other cattle, pigs, poultry, sheep and goats, which are the main categories in Europe (Lesschen *et al.*, 2011). Emission sources include emissions of NH_3, N_2O and CH_4 from animal manure in animal housing, manure storage and treatment systems, and emissions of NH_3 and N_2O from manure applied to land and deposited in pastures (Figure 5-1). Indirect N_2O emissions were excluded. As our study focuses on manure management, enteric CH_4 emissions were not considered.

MITERRA-Europe and data sources. For calculations, the model MITERRA-Europe was further developed and used. MITERRA-Europe is an integrated environmental assessment model which calculates the N and P losses, and GHG emissions on a deterministic and annual basis, using statistical data of agriculture at EU country and regional levels (Velthof *et al.*, 2009; Lesschen *et al.*, 2011). Country specific NH_3 emission factors (EFs) are based on information from the GAINS model, and quantification of N_2O and CH_4 emissions are based on the IPCC guidelines (Klimont & Brink, 2004; Asman *et al.*, 2011).

The N and P excretion for each animal category and country was quantified on the basis of a three-year average (2009-2011), using the nutrient balance of feed intake and animal production (Hou *et al.*, 2016). Data about manure management systems (i.e. animal grazing, daily spreading, liquid and solid based systems) were sourced from the national GHG inventory reports (NIRs) to UNFCCC (UNFCCC, 2016). Data about the degree of implementation of NH_3 mitigation measures and their abatement efficiencies were obtained from GAINS (Klimont & Brink, 2004; Asman *et al.*, 2011), supplemented with information from NH_3 mitigation guidance (UNECE) and review articles (e.g. Melse & Timmerman, 2009; Bittman *et al.*, 2014; Hou *et al.*, 2015; Pardo *et al.*, 2015; Van der Heyden *et al.*, 2015). The IPCC default N_2O-N EFs were adopted to quantify emissions of N_2O from manure management systems, manure application and deposition (IPCC, 2006). For calculating CH_4 emissions, country-specific methane conversion factors (MCFs) for each animal category and manure management system were obtained from NIRs to UNFCCC, considering the allocation of manure management systems among climate zones. Data on nutrient excretion (Table S1) and emission factors (Table S2) are shown in supporting information (SI).

Emissions of GHG, including N_2O and CH_4, were converted to CO_2-equivalent (CO_2-eq), considering the global warming potential of 25 and 298 times of that of CO_2 for CH_4 and N_2O emissions, respectively.

Emissions from manure treatment. A 'manure treatment' module was developed to assess environmental effects of the main treatment technologies in the manure management chain (Figure 5-1). Information on the use of treatment techniques in each country in 2010 was derived from an inventory report on manure treatment activities and NIRs to UNFCCC (UNFCCC, 2016; Foged *et al.*, 2011). Parameters related to emissions (NH_3, N_2O and CH_4) from manure treatment, storage and field application (treated manure), and nutrient recovery are shown in Table S3. Specific characteristics of referenced technologies are follows:

Solid-liquid separation. Three groups of mechanical separator were included: i) screw and filter pressing, ii) non-pressurized filtration and iii) centrifugation and sedimentation. Their separation efficiencies (i.e. the mass of a nutrient element in separated solid fraction, expressed as % of the mass of this element in raw slurry) varied from 10-33% for N and 15-69% for P, which were higher when flocculates and multivalent cations (coagulation-flocculation) were added (Table S3).

Slurry acidification. Acidification involves the addition of concentrated acid (e.g. sulfuric acid) to the slurry in house under slatted floors each day, to lower the pH to 5.5. A fraction of the acidified slurry is then transferred to an outside storage tank without further treatment (Kai *et al.*, 2008; Petersen *et al.*, 2016). The average NH_3 abatement efficiency was 65% in housing, 83% in outdoor storage and 40% during application; a reduction of 87% for CH_4 emissions from housing and storage systems was found (Kai *et al.*, 2008; Hou *et al.*, 2015; Petersen *et al.*, 2016).

Anaerobic digestion. Anaerobic digestion of animal slurry in EU is dominantly operated under mesophilic conditions (Foged *et al.*, 2011). Biogas production from slurry was quantified as function of volatile solid inputs, the yield of CH_4 and the biogas composition (Hamelin *et al.*, 2011). Biogas is considered to be composed of 55-65% CH_4 and 35-45% CO_2, depending on country. Leakage is assumed as 1% of the gross biogas production (Miranda *et al.*, 2015). Liquid and solid fractions were produced from digested slurry, when mechanical separators were used as post treatment.

Nitrification-denitrification treatment. This technology includes a separation unit (pre-treatment), a nitrification-denitrification process (with liquid fractions as input) and a settlement/separation unit (post-treatment). The resulting sludge and liquid effluent (post-treatment), and solid fractions (pre-treatment) are stored before application. Emission factors during treatment and storage were derived from studies conducted in Belgium, France and the Netherlands, where this technique has been applied in practice (Table S3) (Willers *et al.*, 1996; Béline & Martinez, 2002; Loyon *et al.*, 2007; Bernet & Béline, 2009; Melse & Timmerman, 2009)**.**

Composting. Manure sources used for composting consist of solid manure and solid fractions separated from slurry (Foged *et al.*, 2011). Emissions of NH_3 from composting were considered to be 52% higher on average than those from conventional storage, whereas emissions of CH_4 (71%) and N_2O (49%) were lower on average; these percentages were derived from a meta-analysis study (Pardo *et al.*, 2015).

Thermal drying. Poultry manure and separated solid fractions are dried at temperate of 80-150^oC, which is commonly followed by pelletizing as post treatment (Foged *et al.*, 2011). Gaseous emissions from the dryer must be recovered to avoid NH_3 emissions, and air scrubber was assumed to be installed with average NH_3 abatement efficiency of 85% (Foged *et al.*, 2011; Ghaly & Alhattab, 2013). Methane emissions were assumed to be negligible under aerobic conductions.

Bio-drying. This technique is used to treat poultry manure and solid fractions separated from slurry to lower moisture content (by 40-60%) and to facilitate transport. This technique has not been widely used in EU (Foged *et al.*, 2011). Emissions of NH_3 are assumed to be increased (by 121%) relative to static piling due to the forced aeration (Pardo *et al.*, 2015). Changes in N_2O and CH_4 emissions were not always consistent, therefore EFs were assumed the same as that for static piling (Pardo *et al.*, 2015).

Incineration. Industrial-scale incineration of poultry manure exists in several EU countries (e.g. the Netherlands and UK), with the net energy surplus being used for electricity generation (Foged *et al.*, 2011; Billen *et al.*, 2015). The gases released from this process (with gas cleaning installation) mainly consist of CO_2 and N_2, while other emissions (NH_3, N_2O, NOx and CH_4) are minor or negligible (Billen *et al.*, 2015). Ash residues contain all P input from feedstocks and no or only a limited fraction (less than 2%) of the C and N input (Brassard *et al.*, 2014; Christel *et al.*, 2014; Billen *et al.*, 2015; Fernandez-Lopez *et al.*, 2015).

Emissions from application of treated manure. Emissions of NH_3 and N_2O following the application of treated manure were estimated from their N contents and the specific EFs (Table S2 and Table S3). Emission factors were basically obtained from review articles (e.g. meta-analysis), and also from a number of experimental studies.

Defining scenarios. Effects of treatment technologies on emissions of NH_3, N_2O and CH_4 from manure, and the N and P flows in the manure management chain of the EU-27 were further examined through scenario analyses. The assessment focuses on exploring the whole-chain effects and the interactions between gas emissions when certain treatment technologies are implemented. Scenarios are summarized in Table 5-1. Twelve scenarios (S1-12) were compared with the reference, a situation without manure treatment. For all scenarios, we assumed that an equivalent of 20% of total manure (N) removed from animal houses in each country was processed. For all technologies, the same amounts of treated manure are considered, allowing technology comparison. The assumption of 20% implementation corresponded more or less to the upper bound of the degree of implementation of specific treatment in EU countries at present. For example, 11% of total slurry produced was acidified in Denmark in 2010 (Foged *et al.*, 2011), and nearly 20% in 2015. In Italy, 24% of total slurry was treated by solid-liquid separation (Foged *et al.*, 2011). Anaerobic digestion was applied to 13-24% of pig and cattle slurry in Germany, 20-35% in Italy, and 30% of pig slurry in Cyprus (UNFCCC). Nearly one third of chicken manures in the Netherlands was incinerated in 2010, and 15% in UK (Foged *et al.*, 2011; Billen *et al.*, 2015).

Scenarios include increased implementation of single techniques to treat raw slurry (S1-5) and solid manure (S6-9), and also advanced combination of technologies (S10-12). For scenarios with raw slurry as feedstock (S1-5, 10-12), treatment technologies were considered to be applied to slurry removed from animal houses; the amount of slurry that is treated in each scenario equals to 20% of total manure (i.e. the sum of slurry and solid manure) removed from houses, in terms of N. There is an exception for acidification (S4, S10) that is applied to slurry in the houses (i.e. before removal). For scenarios with solid manure as feedstock, technologies were applied to solid manure removed from houses; the amount equals to 20% of total (slurry and solid) manure removed. The total amounts of solid manure (or slurry) that are assumed to be treated may be less than 20% of total manure removed from houses in few countries where manure was produced dominantly as slurry (solid manure).

Although studies about pyrolysis of solid manure have been conducted, this technique has not yet been commercially implemented in Europe (Foged *et al.*, 2011). In scenario (S9), slow pyrolysis of manure at temperature of 600°C was considered with the purpose of producing a stabilized biochar. The mass (C, N and P) balance in pyrolysis is detailed in SI (Table S3). Three treatment systems with combinations of technologies were designed (S10-12) based on literature (Gioelli *et al.*, 2016; Regueiro *et al.*, 2016). In scenario S10, we assumed that slurries that are acidified in houses (as S4) are separated by decanter centrifuge before outdoor storage. In scenario S11, slurries removed from houses are separated (as S2), and followed by acidifying the liquid fractions and by pyrolysis of the solid fractions. In scenario S12, we assumed that all digested slurries from anaerobic digestion (as S3) are acidified and immediately separated (centrifuge).

Uncertainty analysis. Uncertainty analysis was carried out to achieve insight in how variation in the key parameters in the model affected the results, using a Monte Carlo (MC) based method (Zhu *et al.*, 2015). Six groups of parameter were included in this analysis: i) animal numbers, ii) parameters related to nutrient excretion, iii) parameters used to quantify emissions from housing and manure storages (e.g. emission factors), iv) parameters related to manure treatment, v) parameters related to manure application, and vi) manure treatment activity data. Parameter uncertainty is shown in Table S4. The model output uncertainty in response to uncertainty of the parameters was quantified for the year 2010, the reference (without manure treatment) and the scenarios (1000 MC runs each). The difference in emissions between the reference and each treatment scenario was statistically analyzed by comparing the MC simulating outputs, using Tukey HSD test. In addition, the uncertainty contribution of six parameter groups to the overall uncertainty was analyzed for the year 2010.

Table 5-1. Description of manure treatment scenarios[a]

Scenarios	Origin of feedstock	Brief description of treatment systems	Manure products	
			Liquid form	Solid form
S1: Screw press	Slurry (cattle, pigs)	Screw press; LF, SF are not treated further	LF (liquid fraction)	SF (solid fraction)
S2: Decanter centrifuge	Slurry (cattle, pigs)	Decanter centrifuge; LF, SF are not treated further	LF	SF
S3: Anaerobic digestion (AD)	Slurry (cattle, pigs)	Mesophilic digesters	Digestate; LF of digestate	SF of digestate
S4: Acidification (Acid)	Slurry (cattle, pigs)	Acidifying slurry in housing and storage	Acidified slurry	-
S5: Nitrification-denitrification	Slurry (cattle, pigs)	Nitrification-denitrification [b]	Effluents	Sludge; SF
S6: Composting	Solid (cattle, pigs, poultry)	Composting	-	Compost
S7: Thermal drying	Solid (cattle, pigs, poultry)	Thermal drying, with air scrubbers	-	Dried pellets
S8: Incineration	Solid (cattle, pigs, poultry)	Pre-drying and incineration	-	Ash residues
S9: Pyrolysis	Solid (cattle, pigs, poultry)	Slow pyrolysis operated at ~600 °C	-	Biochar
S10: Acid-centrifuge	Slurry (cattle, pigs)	Acidification -> centrifuge	Acidified LF	Acidified SF
S11: Centrifuge-acid, pyrolysis	Slurry (cattle, pigs)	Centrifuge -> acidification (LF); pyrolysis (SF)	Acidified LF	Biochar
S12: AD-acid-centrifuge	Slurry (cattle, pigs)	Anaerobic digestion -> acidification -> centrifuge	Acidified digested LF	Acidified digested SF

[a] Parameter inputs (e.g. emissions factors) are shown in SI (e.g. Table S3); [b] Decanter centrifuge is considered as pre-treatment unit.

5.3 Results

Manure management chain in 2010. Effects of manure treatment on NH_3 and GHG emissions were relatively small in 2010 (Figure 5-2). Approximately 6% of total N excreted by housed animals was treated in EU-27; countries with excretion being treated above this EU average were all in the EU-15 (except Cyprus; Table S1). Manure treatment altered GHG emissions by -17% to 1% at country levels, and on average by -4% at EU level. This was mainly caused by anaerobic digestion (Figure 5-2). Germany, Italy, Denmark, Spain, and the Netherlands had the largest absolute GHG abatement due to manure treatment, sharing 89% of the total abatement in EU-27. In these five countries, anaerobic digestion abated GHG emissions from manure from 1% (in Spain) to 9% (in Germany). Acidification abated 5% of GHG emissions in Denmark, separation 5% in Italy, and composting nearly 3% in Spain (Figure 5-2).

Effects of treatment techniques on NH_3 emissions were minor in most EU countries (Figure 5-2). Reduction in NH_3 emissions was relatively large in Denmark due to acidification (5%), in Belgium (4%) due to nitrification-denitrification treatment, and in Netherlands and UK (2-3%) due to incineration.

Total NH_3 emissions from the manure management chain in EU were 2.5 Tg N and GHG emissions were 86.9 Tg CO_2-eq in 2010 (Figure 5-2a). The proportions of N and P excreted in housings that were applied to land (corrected for NH_3 emissions) were 57% (a range of 52-68% among countries; Table S1) and 98%, respectively (Table 5-2).

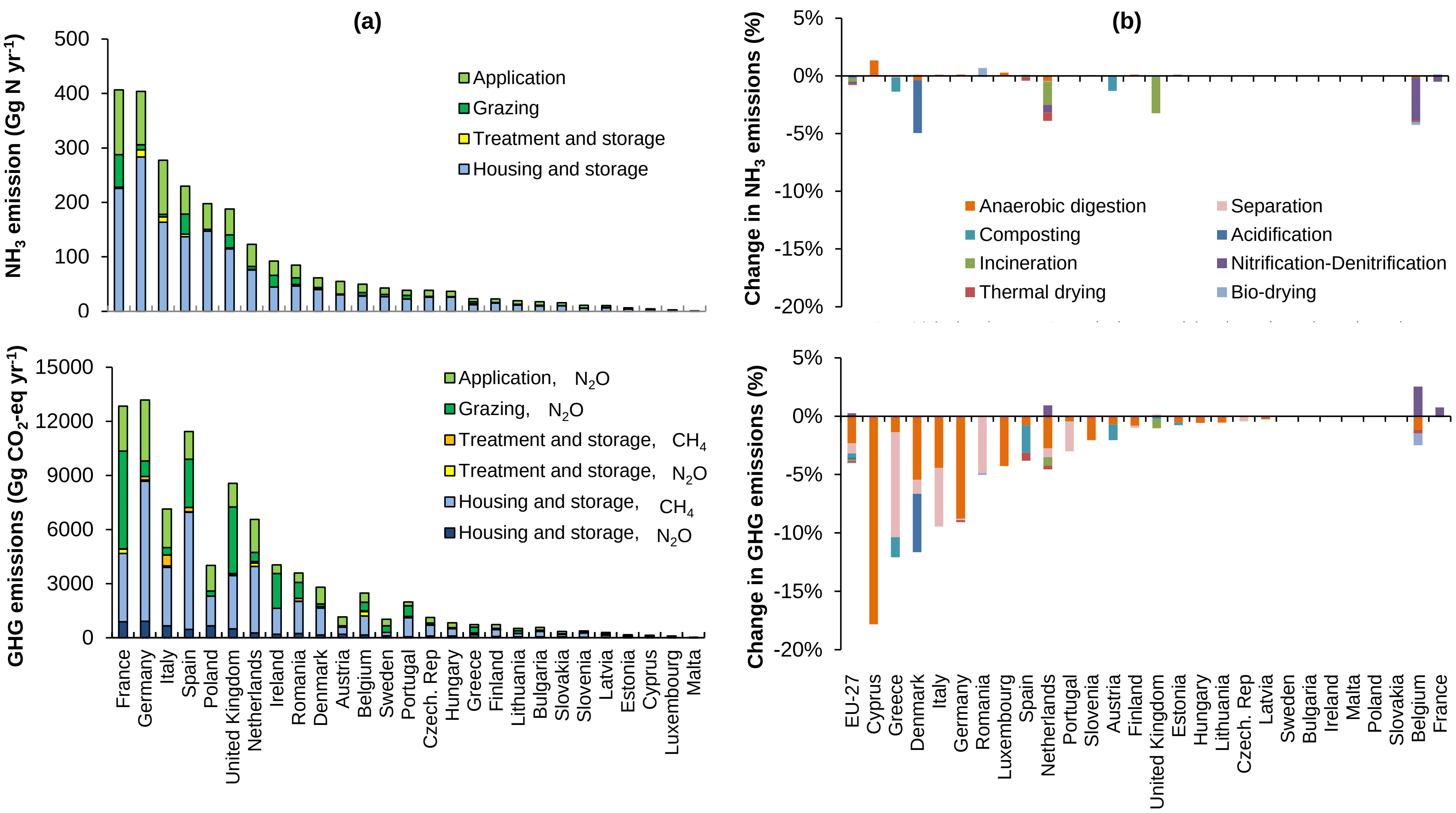

Figure 5-2. Emissions of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) from manure management chains in countries of EU-27 in 2010 (a), and estimated effects of current manure treatment on NH_3 and greenhouse gas (GHG) emissions by comparing situations with and without treatment (b) (positive = increased emission; negative = emission mitigation).

Scenario analyses - NH_3 and GHG emissions. Scenarios with increased implementation of manure treatment technologies (20% of all manure produced in housings) in the EU-27 were compared with the reference (without manure treatment; Figure 5-3).

Increased implementation of slurry separation by screw press (S1) or decanter centrifuge (S2) decreased total GHG emissions by 8% and 12%, respectively. This reduction is due to decreased CH_4 emissions from storage; the greater reduction from decanter centrifuge is related to the larger fraction of storage of separated solid manure. However, total NH_3 emissions changed only marginally. Increased adoption of anaerobic digestion (S3) decreased total GHG emissions by 19% (due to a reduction of 17% in CH_4 emissions), while NH_3 emissions were minimally affected. Increased implementation of acidification (S4) decreased both NH_3 emissions (mainly from housing and storage) and GHG emissions by 10% and 18%, respectively. Nitrification-denitrification treatment (S5) decreased both NH_3 emissions from storage and treatment systems (by 3%) and from manure applied to land (5%). Emissions of N_2O increased by 28% due to nitrification-denitrification treatment, though partly off-set by lower CH_4 emissions (18%). This leads to an increase of 6% in the whole-chain GHG emissions (Figure 5-3).

Composting of solid manure (S6) slightly changed total NH_3 emissions; increased emissions during composting were offset by lower emissions following compost application to land. GHG emissions decreased by 7% relative to the reference. Thermal drying (S7), incineration (S8) and slow pyrolysis (S9) of solid manure decreased both NH_3 (9-11%) and GHG emissions (11-12%). In these three scenarios (S7-9), decreased GHG emissions are due to reduction in both CH_4 emissions from storage systems and N_2O emissions from field relative to the reference; NH_3 emissions decreased during manure storages and application to land (Figure 5-3).

Acidification followed by separation of acidified slurry (S10) decreased NH_3 and GHG emissions nearly similar as implementing acidification alone (S4). Centrifuge separation followed by acidification of liquid fractions and pyrolysis of solid fractions (S11) decreased NH_3 emissions by 7% and GHG emissions by 20%. Anaerobic digestion in combination with acidification and centrifuge (S12) had lower NH_3 emissions compared to anaerobic digestion alone (S3), and had similar reduction potential of GHG emissions (Figure 5-3).

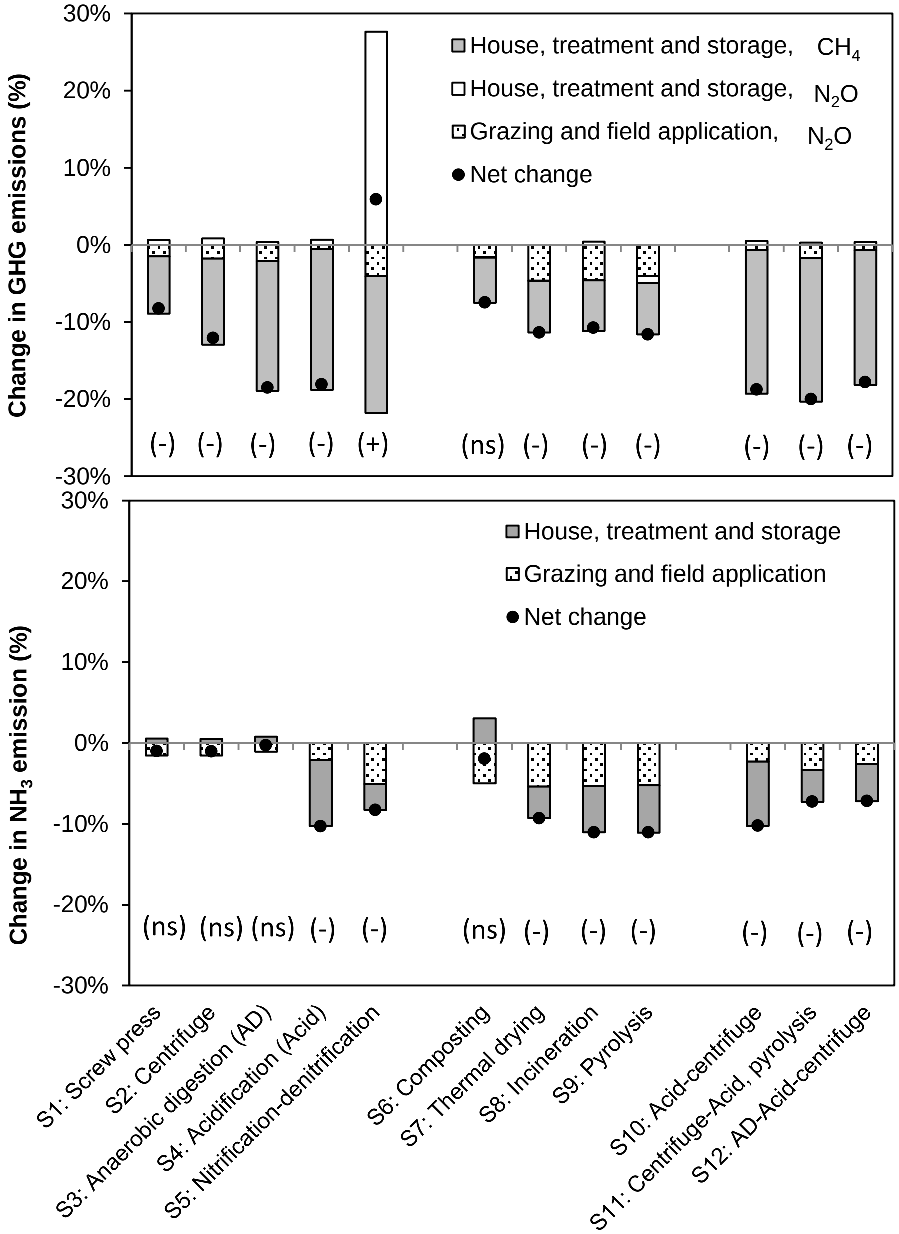

Figure 5-3. Changes in emissions of nitrous oxide (N_2O) and methane (CH_4) (upper panel) and in ammonia (NH_3) emissions (bottom panel) from the manure management chain following the implementation of manure treatment scenarios, relative to a situation without manure treatment (see text and Table 5-1). For each scenario, it was assumed that 20% of total manure produced in housings in a country was treated. (-)/(+) indicates significant difference (lower/higher); (ns) indicates nonsignificant difference.

Scenario analyses – nutrient recovery and content. The amount of total N applied to land, in terms of percentage of total N excreted in housing in EU-27, increased from 57% to 60-61% by acidification treatment (S4, S10, S12; Table 5-2). These N recovery percentages however decreased to 48% with incineration (S8) and to 52% with nitrification-denitrification (S5) and slow pyrolysis (S9). Other technologies changed only marginally N recovery (Table 5-2).

For specific treatment systems (treated manure), the N recovery varied from 2% (through incineration) to 87% (through acidification). The N/P ratio varied from 3.4 to 10.7 in liquid manure products from treatment, compared to a mean of 3.5 in raw slurry. For solid manure products, the N/P ratio ranged from 0.1 to 3.2, compared to 3.0 in raw solid manure (Table 5-2).

Uncertainty. The uncertainty (expressed as coefficient of variation) was 16% for total NH_3 emissions, 20% for GHG emissions and 6% for the N recovery in 2010. Parameters related to emissions from housing and storages are the main factor contributing to the overall uncertainty in total NH_3 emissions (70%) and GHG emissions (39%). Parameters related to emissions from manure application contributed 50% to the overall GHG emission uncertainty. Manure treatment activity data and associated emission parameters contributed less than 1% to the overall emission uncertainty in 2010 (Figure S1).

The differences in emissions between reference and scenarios were statistically significant, except for S1-3 (separation techniques and anaerobic digestion; regarding NH_3 emissions) and for S6 (composting; both NH_3 and GHG emissions) (Figure 5-3).

Table 5-2. Amounts of nitrogen (N) and phosphorus (P) in manure applied to land in the EU-27 in 2010 and in scenarios[a]. Amounts are expressed in percent of the amounts of N and P excreted in housing (i.e. N and P recovery).

	All manure (treated and untreated)		Treated manure			
	N recovery (%)	P recovery (%)	N recovery (%)	P recovery (%)	N/P ratio in manure products[b]	
					Liquid form	Solid form
In 2010	57	98	-[c]	-	3.5 (untreated)	3.0 (untreated)
Scenarios: single technique, slurry						
S1: Screw press	58	98	63	100	3.6	2.0
S2: Decanter centrifuge	58	98	61	100	7.5	1.2
S3: Anaerobic digestion (AD)	58	98	62	100	3.4; 5.0	1.2
S4: Acidification (Acid)	61	98	87	100	4.4	-
S5: Nitrification-denitrification	52	98	33	100	8.6	2.9; 1.1
Scenarios: single technique, solid manure						
S6: Composting	58	99	60	100		3.2
S7: Thermal drying	58	99	57	100		3.1
S8: Incineration	48	99	2	100		0.1
S9: Pyrolysis	52	99	25	100		2.1
Scenarios: combined techniques, slurry						
S10: Acid-centrifuge	61	98	84	100	10.7	1.5
S11: Centrifuge-acid, pyrolysis	57	98	61	100	9.7	0.5
S12: AD-acid-centrifuge	60	98	74	100	10.0	1.4

[a] Emissions of ammonia were subtracted from the N in applied manure for calculating the N recovery; [b] manure products from respective treatment technologies are explained in Table 5-1 (liquid manure products from AD include digested slurry and liquid fraction separated from digestate; solid manure products from nitrification-denitrification include sludge and solid fraction separated from slurry). [c] not applicable

5.4 Discussion

Manure treatment in EU. Anaerobic digestion and solid-liquid separation are by far the most popular treatment technologies in Europe, and anaerobic digestion had the largest contribution to overall GHG mitigation of all treatment technologies (Figure 5-2). Germany and Denmark have been most successful in promoting anaerobic digestion. The success is due to national government support (e.g. investment support for construction and subsidies on bioenergy delivery) and the enforcement of EU environmental regulations (e.g. Nitrates Directive, Renewable Energy Directive) (Edwards *et al.*, 2015). The Danish government proposed a target of using 50% of the manure produced for renewable energy by 2020, which would essentially be met through an expansion of biogas plants (Danish Agrifish Agency). Slurry separation has been adopted by livestock farms in many EU countries (particularly in Italy and Portugal), where GHG emissions might be decreased due to the adoption of this technique (Figure 5-2). This reduction is mainly because of the lower CH_4 emissions from storage of separated solid fractions relative to that from storage of raw slurry, which is in line with other whole-farm scale assessment (Sommer *et al.*, 2009). Also, we assumed that separation is near the source of production, i.e. the slurry was not stored prior to separation. Anaerobic digestion and slurry separation have not changed NH_3 emissions much (Figure 5-2).

The adoption of other treatment technologies was concentrated in a few specific countries, therefore their contributions were limited at EU scale. Slurry acidification with the purpose of NH_3 abatement also reduces CH_4 emissions from slurry during storage (Hou *et al.*, 2015). The reduction in CH_4 emissions is attributed to the inhibition of methanogenesis, because of the acidic conditions and the high concentrations of sulphate (which is an electron acceptor) and sulphide (which is toxic) (Ottosen *et al.*, 2009; Petersen *et al.*, 2012). However, this technique is only used in Denmark. Poultry incineration occurs in Netherlands and UK; there is evidence that CH_4 and N_2O emissions are negligible, while the electricity production 'saves' emissions by replacement of fossil fuel combustion (Billen *et al.*, 2015). Composting occurs in many countries in small scale units; a meta-analysis indicates that composting tends to decrease N_2O and CH_4 emissions compared to conventional static storage of solid waste, yet with large variations in the magnitude of decrease (Pardo *et al.*, 2015). Although manure treatment is on average still marginal in Europe, most techniques (e.g. anaerobic digestion, acidification, incineration) contribute to significant decreases in GHG emissions and/or NH_3 emissions. This indicates the importance of taking manure treatment into account for national emission inventories.

Implications of scenario analyses. Our scenario analyses indicate that processing an equivalent of 20% of total manure production from housings will decrease NH_3 emissions by 0 to 11% and will alter GHG emissions by -20 to +6% from the manure management chain in EU-27, relative to the reference (Figure 5-3). Both NH_3 and GHG emissions decreased relatively strong in scenarios with acidification (S4, S10-12), and to a lesser extent also with thermal drying (S7), incineration (S8) and pyrolysis (S9) (Figure 5-3).

Manure N is a major source (81%) of NH_3 emissions from agriculture in Europe. It has been reported that implementation of an optimal combination of NH_3 mitigation measures (covered manure storages, low-emission application etc.) may decrease total NH_3 emissions in EU-27 by 316 Gg N (Oenema *et al.*, 2009). Our results suggest that manure treatment, implemented to treat 20% of total manure production from housings, have comparable emission abatement (180-275 Gg N, referring to S4-5, S7-12). Some manure treatment technologies may also have relatively large potential to mitigate GHG emissions (Figure 5-3), compared to conventional NH_3 abatement measures that minimally affect CH_4 and N_2O emissions, or increase N_2O emissions (Oenema *et al.*, 2009; Velthof *et al.*, 2009).

Nitrification-denitrification treatment increased GHG emissions because of the increased N_2O emissions from the reactor, while CH_4 emissions decreased (Figure 5-3). Emissions of N_2O from nitrification-denitrification may range from 1 to 20% of the slurry N input (Willers *et al.*, 1996; Béline & Martinez, 2002; Loyon *et al.*, 2007; Melse & Timmerman, 2009). The IPCC default N_2O EFs for raw slurry in storages have a much smaller range: 0-0.5% (IPCC, 2006). Emissions of N_2O from nitrification-denitrification treatment may be decreased by increasing the residence time of the slurry in the denitrification reactor or by a better control of the molasses addition (through measurement of the redox potential) (Béline & Martinez, 2002; Melse & Verdoes, 2005). These unwanted side effects have to be minimized; this is especially important for Brittany (France) and Flanders (Belgium) where this technique has been implemented to decrease the manure N surplus (Bernet & Béline, 2009).

Increasing the efficiency of manure N and P use as fertilizer is economically beneficial because it lowers the need for chemical fertilizers. However, scenarios with nitrification-denitrification (S5), incineration (S8) and slow pyrolysis (S9) indicate a low recovery fraction available for application to land (Table 5-2). These technologies cannot be considered as sustainable from a resource use efficiency point of view, as they convert the majority of N (about 65% to 100%) to a form (dinitrogen gas) that cannot be utilized anymore for fertilization (Melse & Timmerman, 2009; Billen *et al.*, 2015). During these treatments (S5, S8-9) a significant fraction of the carbon in the manure is also lost, and thus less organic matter is available for improving soil quality. In the scenario with manure incineration, the N loss equals to 8% of mineral N fertilizer consumed in EU in 2010. In livestock-rich regions that produce more manure N than crops need, these technologies may help to lower manure N surpluses. For scenarios with acidification (S4, S10, S12), the manure N recovery increased because of the N saved from volatilization (Table 5-2). The N availability in acidified slurry increases the N fertilizer equivalent value, by about 25% compared to raw slurry (Kai *et al.*, 2008; Sørensen & Eriksen, 2009).

Phosphate rock as a non-renewable resource is scarce especially in EU where more than 95% of mineral P fertilizer used has been imported from outside EU (van Dijk *et al.*, 2015). For all manure treatment scenarios, we estimated that the total P recovery did not decrease (Table 5-2). Nevertheless, the P availability for crops (in terms of first year P fertilizer equivalent value) varies, from less than 20% for ash residues to 80%-100% for acidified slurry or separated liquid fractions (Sommer *et al.*, 2013). Manure treatment generates manure products that vary in the ratios between nutrient (e.g. N, P and potassium), such as the N/P ratios (0.1-10.7; Table 5-2). This provides opportunities to better use manure and to better meet crop nutrient demands. Depending on between-farm variations in crop rotations and soil fertility, farmers may choose the manure product with the appropriate N/P ratio.

Green energy production from anaerobic digestion and incineration of animal manure can be viewed as pathways to contribute to achieving the EU Renewable Energy target; at least 27% of the total energy needs have to come from renewables by 2030 (European Commission, 2009). Green energy production leads to CO_2 emission reduction due to the replacement of heat and electricity produced from fossil fuels. We estimate that the net energy production is 1.7-2.8 Mtoe (Million Tonnes of Oil Equivalent) for the scenario with anaerobic digestion (S3), assuming an energy surplus of 50-80% in biogas plants, used for heat and electricity cogeneration (Sommer *et al.*, 2009; Miranda *et al.*, 2015). The avoided CO_2 emissions from fossil fuels are 7.6-12.1 Tg CO_2-eq, assuming an emission factor of 0.102 kg CO_2 MJ^{-1} for power production from coals (Sommer *et al.*, 2009; Miranda *et al.*, 2015). This is equivalent to 9-14% of GHG emissions estimated for the manure management chain in EU-27 in 2010.

Our scenario analyses illustrate that various treatment technologies can be viewed as measures to mitigate GHG and/or NH_3 emissions from the manure management chain.

Increasing the implementation of these technologies in practice may contribute to achieving the NH_3 emission targets of the National Emission Ceiling Directive (Directive 2001/81/EC), and the GHG emission targets of the Kyoto Protocol (UNFCCC). Slurry acidification, incineration and pyrolysis are technologies that reduce both NH_3 and GHG emissions. Acidification also increases the N recovery, while incineration and pyrolysis may reduce manure N surpluses in regions with high animal density. Solid-liquid separation produces manure products with diverse N and P contents, which allows farmers to better meet crop-specific nutrient demands. Combing anaerobic digestion with acidification or with other NH_3 mitigation measures (for storage of digested slurry) is needed to achieve abatement of both GHG and NH_3 emissions.

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Supporting information:

Table S1. The calculated nutrient excretion by the livestock in the EU-27 in 2010.

Country	Total excretion (Gg)				Manure management systems (% of total N excretion)				The amount of N excretion being treated, as % of total N excreted in housing	N applied to land, as % of N excreted in housing
	Nitrogen	Phosphorus	Volatile solid[a]	Carbon[a]	Grazing	Housing (slurry)	Housing (solid)	Daily spreading		
Austria	187	32	2980	1518	5	37	58	0	6	60
Belgium	238	42	3386	1743	28	42	29	1	12	62
Bulgaria	69	14	1248	639	22	6	72	0	<0.1	57
Cyprus	14	3	189	99	0	32	68	0	8	56
Czech. Rep	122	23	2237	1144	13	35	51	1	1	56
Denmark	253	53	2952	1537	8	74	18	0	13	68
Estonia	24	4	420	213	32	25	44	0	2	54
Finland	81	15	1189	610	11	49	41	0	2	64
France	1730	282	26047	13263	45	27	29	0	1	55
Germany	1235	224	19809	10166	10	59	31	0	9	63
Greece	136	28	2261	1151	59	7	33	0	18	53
Hungary	111	24	1943	1019	9	16	75	0	<0.1	53
Ireland	468	69	6676	3357	61	29	10	0	<0.1	56
Italy	870	145	14743	7558	10	39	50	0	12	56
Latvia	39	6	745	377	29	25	47	0	<0.1	54
Lithuania	72	12	1251	634	30	25	45	0	<0.1	54
Luxembourg	12	2	192	96	45	38	17	0	8	58
Malta	3	1	33	18	0	28	72	0	<0.1	53
Netherlands	548	87	6789	3523	13	67	19	0	8	61
Poland	606	115	11015	5681	7	10	83	0	<0.1	53
Portugal	176	28	2772	1426	52	16	32	0	2	52
Romania	360	69	6699	3412	42	11	45	2	5	54
Slovakia	53	9	949	486	13	12	74	0	<0.1	54
Slovenia	33	6	667	338	12	56	32	0	1	56
Spain	972	168	15286	7856	45	24	31	0	5	55
Sweden	169	25	2434	1237	30	35	35	0	<0.1	59
United Kingdom	1131	181	16877	8592	57	13	30	1	6	54
EU-27	9716	1669	151790	77691	31	32	36	<0.5	6	57

[a] VS per animal category and country was determined as function of feed VS intake and VS digestibility (IPCC, 2006; CSIRO, 2007; Hou *et al*., 2016). The constant ratios of TC/VS in excretion (0.5 for ruminants, 0.54 for pigs and 0.58 for poultry) were used to quantify carbon content in excretion.

Table S2 Emission factors of ammonia (NH_3), nitrous oxide (N_2O), dinitrogen (N_2) and nitrogen oxides (NO_x) from animal manure in the EU-27 at country levels in 2010, based on the model MITERRA-Europe (Velthof *et al.*, 2009; Lesschen *et al.*, 2011) and additional calculations.

Country	NH_3				N_2O			N_2 and NO_x
	Housing (% N excreted in housing)	Storage of (raw) manure (% of N in storage)	Raw manure application (% of N applied)	Grazing (% of N deposited by grazing animals)	Storage of (raw) manure (% N in storage)	Raw manure application (% of N applied)	Grazing (% of N excreted by grazing animals)	Storage of raw manure (% N excreted in housing)
Austria	13	5	19	7	0.38	1.0	1.8	7
Belgium	13	5	14	8	0.34	1.2	2.0	5
Bulgaria	13	6	18	8	0.38	1.1	1.4	10
Cyprus	12	13	14		0.36	1.2		8
Czech. Rep	14	12	16	8	0.32	1.2	1.9	7
Denmark	9	11	10	7	0.26	1.5	2.0	3
Estonia	13	12	19	8	0.37	1.0	2.0	7
Finland	12	10	12	8	0.33	1.1	2.0	5
France	13	12	18	8	0.30	1.1	1.9	6
Germany	15	13	12	8	0.29	1.2	1.9	5
Greece	13	14	18	4	0.31	1.1	1.1	10
Hungary	14	14	16	6	0.34	1.2	1.6	9
Ireland	18	8	20	8	0.38	1.0	1.9	4
Italy	12	12	19	5	0.31	1.2	1.2	7
Latvia	13	12	19	8	0.36	1.1	2.0	7
Lithuania	13	12	19	8	0.37	1.1	2.0	7
Luxembourg	13	10	20	8	0.32	1.0	2.0	4
Malta	15	9	20		0.32	1.0		8
Netherlands	10	8	13	7	0.22	1.6	1.9	3
Poland	15	13	14	8	0.38	1.1	2.0	9
Portugal	14	15	18	7	0.24	1.1	1.8	7
Romania	13	12	17	8	0.41	1.0	1.6	9
Slovakia	12	12	17	7	0.38	1.2	1.7	9
Slovenia	14	7	22	5	0.32	1.0	1.8	5
Spain	13	15	15	8	0.30	1.3	1.7	7
Sweden	13	11	14	8	0.33	1.2	2.0	6
United Kingdom	15	11	15	4	0.36	1.2	1.6	8

Table S3 Summary of key parameters and data sources used for the environmental assessment of manure treatment technologies in this study.

Manure treatment technologies	Parameter codes	Values used in the model	Description	Data sources
Acidification	$Re_{build\_acid\_NH3}$	65%	NH_3 and CH_4 reduction factors during in-house (Re_{build_acid}), outdoor storage ($Re_{storage\_acid}$), and application (Re_{applic_acid}) of acidified slurry, relative to raw slurry	(Kai *et al.*, 2008; Petersen *et al.*, 2016)
	$Re_{storage\_acid\_NH3}$	83%		(Hou *et al.*, 2015)
	$Re_{storage\_acid\_CH4}$	87%		(Hou *et al.*, 2015)
	$Re_{applic\_acid\_NH3}$	40%		(Kai *et al.*, 2008; Fangueiro *et al.*, 2015a, 2015b)
Solid-liquid Separation	Et_{sep_N}	10%; 22%	Separation efficiency (Et_{sep}) of N, P and VS, for i) screw and filter pressing; ii) combined with coagulation/ flocculation	(Møller *et al.*, 2007b, 2000, 2002, 2007a; Converse & Karthikeyan, 2004; Melse & Verdoes, 2005; Rico *et al.*, 2007; Campos *et al.*, 2008; Fangueiro *et al.*, 2008, 2009; Hjorth *et al.*, 2010; Hamelin *et al.*, 2011; Popovic *et al.*, 2012)
	Et_{sep_P}	15%; 32%		
	Et_{sep_VS}	29%; 55%		
	Et_{sep_N}	33%; 47%	i) non-pressurized filtration; ii) combined with coagulation/ flocculation	
	Et_{sep_P}	41%; 81%		
	Et_{sep_VS}	49%; 76%		
	Et_{sep_N}	26%; 47%	i) centrifugation and sedimentation; ii) combined with coagulation/ flocculation	
	Et_{sep_P}	69%; 81%		
	Et_{sep_VS}	58%; 76%		
	$Re_{applic\_LF\_NH3}$	18%	NH_3 reduction factors for application of liquid fraction, relative to raw manure	(Hou *et al.*, 2015)
	$Re_{applic\_SF\_N2O}$	45%	N_2O reduction factors for application of solid fraction, relative to raw manure	(Hou *et al.*, 2015)
Anaerobic digestion	f_{biogas_leak}	1%	Biogas leakage, % of biogas production	(Miranda *et al.*, 2015)
	f_{biogas_CH4}	55-65%	CH_4 in biogas composition	(Hamelin *et al.*, 2011; UNFCCC)
	$If_{storage\_dig\_NH3}$	50%	Increase in NH_3 emission from storage of digested slurry, relative to raw slurry	(Sommer, 1997; Clemens *et al.*, 2006; Koirala *et al.*, 2013)
	$Re_{storage\_dig\_CH4}$	78.5%	CH_4 reduction factors from storage of digested slurry, relative to raw slurry; N_2O emissions are assumed to be unchanged.	(Miranda *et al.*, 2015)
	f_{Deg_vs}	0.45-0.6	Degradable rates of VS of manure in digester, varied among animal categories	(Møller *et al.*, 2004a, 2004b; Hamelin *et al.*, 2011, 2014)
	$Re_{applic\_dig.LF\_NH3}$	18%	NH_3 reduction factors for application of digested liquid fraction; NH_3 EF from digested slurry are considered to be same as raw manure.	(Hou *et al.*, 2015)
	$Re_{applic\_dig/dig.LF\_N2O}$	25%	N_2O reduction factors for application of digested slurry and liquid fraction Of digestate.	(Hou *et al.*, 2015)

Nitrification and denitrification treatment (NDN)	EF_{NDN_N2O}	9%	N_2O emission factor (EF) during NDN; a range 1-20% was reported in literature	(Willers *et al.*, 1996; Béline & Martinez, 2002; Melse & Verdoes, 2005; Loyon *et al.*, 2007)
	EF_{NDN_NH3}	0.5%	NH_3 EF during NDN; a range 0.1-0.8%	(Willers *et al.*, 1996; Melse & Verdoes, 2005)
	EF_{NDN_TN}	70%	Total N loss during NDN, a range 52-80%	(Béline & Martinez, 2002; Beline *et al.*, 2008; Riaño & García-González, 2014)
	EF_{NDN_CO2}	15%	CO_2 EF during NDN; a range 52-80%	(Loyon *et al.*, 2007)
	EF_{NDN_CH4}	0.25%	CH_4 EF during NDN; a range 0.04-0.34%	(Melse & Verdoes, 2005; Loyon *et al.*, 2007)
	$EF_{NDN\_slud\_CH4}$	0.5%	CH_4 EF during storage of sludge	(Loyon *et al.*, 2007)
	$EF_{NDN\_slud\_NH3}$	1.5%	NH_3 EF during storage of sludge	(Loyon *et al.*, 2007)
	$EF_{NDN\_slud\_N2O}$	0.1%	N_2O EF during storage of sludge	Assumption
	$EF_{NDN\_slud\_N2}$	0.5%	N_2 EF during storage of sludge	Assumption
	$EF_{applic\_\_NDN\_slud\_NH3}$	1%	NH_3 EF during application of sludge	
	$Re_{applic\_SF\_N2O}$	45%	N_2O reduction factors for application of sludge, relative to raw manure, considered same as separated solid fraction	(Hou *et al.*, 2015)
Composting	$If_{compost\_NH3}$	52%	Increasing factors ($If_{compost}$) or reduction factors ($Re_{compost}$) for emissions during composting, relative to conventional storage	(Pardo et al., 2015)
	$If_{compost\_CO2}$	38.5%		
	$Re_{compost\_CH4}$	71%		
	$Re_{compost\_N2O}$	49%		
	$EF_{applic\_compost\_NH3}$	1%	NH_3 EF during application of compost	(Chadwick *et al.*, 2011)
	$EF_{applic\_compost\_N2O}$	1%	N_2O EF during application of compost	(IPCC, 2006; Chadwick *et al.*, 2011)
Incineration	$EF_{incinerate\_NH3}$	0.03%	NH_3 EF of incineration; a range 0.01-0.06%	(Brassard *et al.*, 2014; Billen *et al.*, 2015)
	$EF_{incinerate\_N2O}$	0.3%	N_2O EF of incineration; a range 0.04-0.7%	(Brassard *et al.*, 2014; Billen *et al.*, 2015; Fernandez-Lopez *et al.*, 2015)
	$EF_{incinerate\_NOx}$	0.35%	NO_x EF of incineration; a range 0.1-0.4%	(Brassard *et al.*, 2014; Billen *et al.*, 2015)
	$f_{incinerate\_residue\_N}$	2%	Fraction of N residue, a range 1.4-4%	(Christel *et al.*, 2014)
	$f_{incinerate\_residue\_C}$	1%	Fraction of C residue, a range 0.05-2.2%	(Brassard *et al.*, 2014; Christel *et al.*, 2014; Fernandez-Lopez *et al.*, 2015)
	$EF_{applic\_ash\_NH3/N2O}$	Negligible	NH_3 and N_2O EF of application of ash	Assumption
Thermal drying	$EF_{thermal\text{-}dry\_\_NH3}$	Poultry: 25%,Pig: 30%,Cattle:30%	NH_3 EFs during drying (Pig: 25-36%; Cattle: 25-36%; Poultry: 21-29%)	(Sistani *et al.*, 2001; Maurer & Müller, 2012; Ghaly & Alhattab, 2013; Pantelopoulos *et al.*, 2016)
	$Re_{dry\_scrubber\_NH3}$	85%	NH_3 reduction factor by air scrubber; a range 71-99%	
	$EF_{thermal\text{-}dry\_\_N2O}$	0.02%	N_2O EF, a range 0.01-0.5%	
	$EF_{thermal\text{-}dry\_\_NOx}$	0.02%	NO_x EF, assumed to be same as N_2O	Assumption
	$EF_{thermal\text{-}dry\_\_N2}$	5%	N_2 EF during thermal drying	Assumption
	$EF_{applic\_dry\_\_NH3}$	negligible	NH_3 EFs during application	(Rodhe & Karlsson, 2002)

Bio-drying	If_{biodry_NH3}	121%	Increase in NH_3 emission during bio-drying, a range 28-243% was reported in literature	(Ramirez *et al.*, 2012; Sadaka & Ahn, 2012; Pardo *et al.*, 2015)
	$EF_{applic\_bio-dry\_NH3}$	1%	Assumption, considering the limited available N content in dried manure	
Slow pyrolysis	$f_{biochar\_C}$	35% (pigs); 44% (cattle, poultry)	Fraction of C in biochar, as % of C input; a range 32-56%	(Kim *et al.*, 2009; Ro *et al.*, 2010; Cantrell *et al.*, 2012; Song & Guo, 2012; Azuara *et al.*, 2013; Li & Takarada, 2013; Troy *et al.*, 2013; Wnetrzak *et al.*, 2013; Baniasadi *et al.*, 2016)
	$f_{biochar\_N}$	25% (pigs); 30% (cattle, poultry)	Fraction of N in biochar, as % of N input a range 5-40%	
	$f_{biochar\_p}$	99%	Fraction of P in biochar, as % of P input a range of 97-100% was reported	
	$EF_{applic\_biochar\_NH3}$	1%	NH_3 EFs during application of biochar	(Kookana *et al.*, 2011; Cayuela *et al.*, 2014)
	$EF_{applic\_biochar\_N2O}$	0.5%	N_2O EFs from application of biochar	

Table S4. A summary of uncertainty in model parameters used for the Monte Carlo (uncertainty) analysis (CV= coefficient of variation; SD =Standard deviation).[a]

Parameter codes (if applicable, see Table S3)	Brief description	Group[b]	Distribution type	CV	SD	Spatial correlation, national level	Source (uncertainty)
-	Livestock numbers	LAD	normal	0.05		0.5	(Zhu *et al.*, 2015)
E_{xcreta_N}	N excretion factors	EXE	normal	0.1		0.5	(Hou *et al.*, 2016)
E_{xcreta_vs}	VS excretion factors	EXE	normal	0.1		0.5	(Hou *et al.*, 2016)
$f_{TC/VS}$	The TC/VS ratio in excretion	EXE	normal	0.25		1	This study[c]
$EF_{build\_slurry/solid\_NH3}$	NH_3 emission factors (EFs) in housing	EFH	normal	0.25		0.8	(Zhu *et al.*, 2015)
$EF_{storage\_lurry/solid\_NH3}$	NH_3 EFs in storage	EFH	normal	0.25		0.8	(Zhu *et al.*, 2015)
$EF_{storage\_slurry/solid\_N2O}$	N_2O EFs in storage	EFH	Lognormal		0.35	0.5	(Zhu *et al.*, 2015)
$EF_{storage\_slurry/solid\_N2}$	N_2 EFs in storage	EFH	Lognormal		0.5	0.5	(Zhu *et al.*, 2015)
$MCF_{slurry/solid}$	Methane conversion factors	EFH	normal	0.25		0.8	(Zhu *et al.*, 2015)
f_{acid_slurry}	Fractions of acidified slurry	TAD	normal	0.25		0.2	This study
$f_{separate/compost/thermal-dry/biodry\_slurry}$	Fractions of slurry used for separation, composting, thermal or bio-drying	TAD	normal	0.25		0.2	This study
f_{AD_slurry}	Fractions of slurry used for anaerobic digestion	TAD	normal	0.25		0.2	This study
$f_{compost/incineration/thermal-dry\_solid/biodry\_solid}$	Fractions of solid manure used for incineration, composting, thermal or bio-drying	TAD	normal	0.25		0.2	This study
$Re_{storage\_acid\_NH3}$	NH_3 reduction factor, acidification	EFT	normal	0.47		1	(Hou *et al.*, 2015)
$Re_{storage\_acid\_CH4}$	CH_4 reduction factor, acidification	EFT	normal	0.44		1	(Hou *et al.*, 2015)
Et_{sep}	Separation efficiency	EFT	normal	0.25		1	This study
$Yield_{biogas\_CH4}$	Biogas-CH_4 yield	EFT	normal	0.25		1	This study
f_{biogas_CH4}	Biogas composition	EFT	normal	0.25		1	This study
f_{Deg_vs}	Degradation rates, in AD	EFT	normal	0.25		1	This study
$If_{storage\_dig\_NH3}$	NH_3 increase factor, digestate stored	EFT	normal	0.5		1	This study
$Re_{storage\_dig\_CH4}$	CH_4 reduction factor, digestate stored	EFT	normal	0.25		1	(Miranda *et al.*, 2015)
EF_{NDN_NH3}	NH_3 EFs, NDN treatment	EFT	normal	0.5		1	This study
EF_{NDN_N2O}	N_2O EFs, NDN treatment	EFT	normal	0.5		1	This study

EF_{NDN_N2}	N_2 EFs, NDN treatment	EFT	normal	0.25		1	This study
$f_{incinerate\_residue}$	Fractions of nutrients in ash, incinerate	EFT	normal	0.25		1	This study
$EF_{incinerate\_NH3/N2O/NOx}$	EFs from incineration	EFT	normal	0.25		1	This study
$EF_{thermal-dry\_NH3/N2O/NOx/N2}$	EFs from thermal-dry	EFT	normal	0.5		1	This study
$Re_{dry\_scrubber\_NH3}$	NH_3 reduction efficiency, air scrubber	EFT	normal	0.1		1	(Ghaly & Alhattab, 2013)
If_{biodry_NH3}	NH_3 increase factor, during bio-drying	EFT	normal	0.5		1	This study
$If_{compost\_NH3}$	NH_3 increase factor, composting	EFT	normal	0.59		1	(Pardo *et al.*, 2015)
$Re_{compost\_N2O}$	N_2O reduction factor, composting	EFT	normal	0.79		1	(Pardo *et al.*, 2015)
$Re_{compost\_CH4}$	CH_4 reduction factor, composting	EFT	normal	0.5		1	(Pardo *et al.*, 2015)
$f_{biochar\_C\&N\&P}$	Nutrient fractions remained in biochar	EFT	normal	0.25		1	This study
$EF_{applic\_slurry/solid/compost(etc.)\_NH3}$	NH_3 EFs for applied manure products	EFA	normal	0.25		0.8	(Zhu *et al.*, 2015)
$EF_{graz\_slurry\_NH3}$	NH_3 EFs from excreta of gazing animal	EFA	normal	0.25		0.8	(Zhu *et al.*, 2015)
$Re_{applic\_LF/acid}$	NH_3 reduction factors, applied (separated) liquid fraction and acidified slurry	EFA	normal	0.5		1	(Hou *et al.*, 2015)
$EF_{applic\_slurry//compost(etc.)\_\_N2O}$	N_2O EFs for applied manure products	EFA	Lognormal		0.28	0.5	(Zhu *et al.*, 2015)
$Re_{applic\_dig/sep.SF(etc.)\_N2O}$	N_2O reduction factors, applied treated manure	EFA	normal	0.49		1	(Hou *et al.*, 2015)
$EF_{graz\_slurry\_N2O}$	N_2O EFs from excreta of gazing animal	EFA	Lognormal		0.57	0.5	(Zhu *et al.*, 2015)

[a] The uncertainty analysis was performed according to the methodology described by Zhu *et al.* (2015) who analyzed the uncertainty of the MITERRA model for global animal production; the uncertainties (CV or SD) were also mainly derived from Zhu *et al.* (2015), and other meta-analysis studies (Hou *et al.*, 2015; Miranda *et al.*, 2015; Pardo *et al.*, 2015).

[b] LAD: livestock activity data (i.e. animal numbers); EXE: parameters used for calculating nutrient excretion; EFH: parameters (e.g. emission factors) used for calculating emissions from housing and manure storages; EFT: parameters used for calculating emissions during manure treatment; EFA: parameters used for calculating emissions during manure application to land; TAD: manure treatment activity data.

[c] For these parameters, little information on uncertainties was available. We assigned the CV of these parameters using three categories: high uncertainty (CV = 0.5), moderate uncertainty (CV = 0.25) and low uncertainty (CV = 0.1), following (Kros *et al.*, 2012; Zhu *et al.*, 2015). Low uncertainty was used for parameters derived from good-quality statistical databases; high uncertainty was used for parameters based on expert knowledge or derived from model estimates or limited published data.

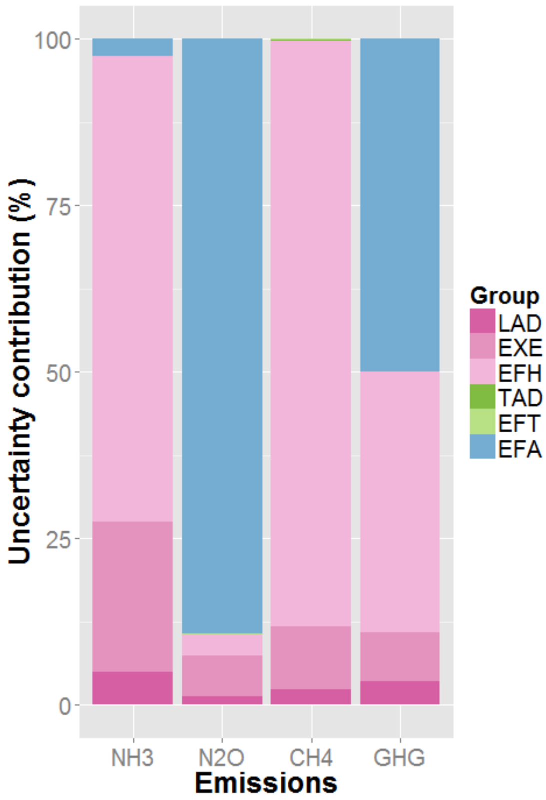

Figure S1. Uncertainty contribution of different groups of model inputs and parameters to the total emissions of NH_3, N_2O, CH_4 and GHG from animal manure in EU-27 in 2010. (LAD: livestock activity data (i.e. animal numbers); EXE: parameters used for calculating nutrient excretion; EFH: parameters (e.g. EFs) used for calculating emissions from housing and manure storages; EFT: parameters used for calculating emissions during manure treatment; EFA: parameters used for calculating emissions during manure application to land; TAD: manure treatment activity data; see details in Table S4)

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CHAPTER 6

Stakeholder perceptions of manure treatment technologies in Denmark, Italy, the Netherlands and Spain

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Abstract:

Animal manures are valuable sources of plant nutrients, bioenergy and organic matter for enhancing soil quality, but are also associated with a range of environmental issues. Manure treatment technologies have been developed in Europe to better use animal manures and to reduce their environmental impact, but the adoption of these technologies in practice is regionally diverse and still limited. Also, little is known about the opinions of stakeholders towards manure treatment. This study aimed to identify stakeholder perceptions of (1) which factors can facilitate and hinder the implementation in practice, (2) which technologies have the most potential for successful adoption, and (3) how farm characteristics and scale of treatment operations affect priorities for technology adoption. This analysis used data from a survey of various stakeholders engaged in manure treatment in four European countries (Denmark, Italy, the Netherlands and Spain) that have large areas of high animal density, but diverse socio-economic, political and environmental conditions. Pressure from governmental regulations was perceived as a key factor that stimulated manure treatment in all four countries (70% of respondents). Processing manure to produce bioenergy was considered important in Denmark and Italy, but less important in Spain and the Netherlands. The major barriers to technology adoption were related to economic factors (lack of investment capital, high processing cost and a long payback time; 45-60% of respondents), while there was relatively little concern regarding transport, noise burden and health risk. Slurry separation and anaerobic digestion were perceived to have the greatest potential for a common adoption in practice. Other preferred technologies were more country-specific (e.g. acidification in Denmark, composting in Spain, and drying and reverse osmosis in Netherlands). Farm characteristics and scale of operation were identified as important factors for technology adoption. The implementation of manure treatment in practice was facilitated by the pressure from environmental regulations, and was hindered by financial barriers. Manure treatment will therefore remain a regional activity. Policy measures and outreach strategies to alleviate the main barriers and to encourage the adoption of manure treatment are suggested.

6.1 Introduction

Animal manures are valuable sources of plant nutrients, soil organic matter and bioenergy. However, following the introduction of relatively cheap inorganic fertilizers from the 1950s onwards, animal manures were increasingly considered as a waste, especially in affluent countries (e.g., in Europe and North America; Van der Meer, 1987). Recently, inappropriate use and inefficient recycling of animal manures, particularly in regions with high animal density, have exerted a series of negative impacts on the environment, e.g. eutrophication of ecosystems, soil acidification and global warming (Steinfeld *et al.*, 2006). In Europe, the livestock sector is currently responsible for about 80% of total European ammonia (NH_3) emissions, 10-17% of greenhouse gas (GHG) emissions, 40-50% of diffuse nitrogen (N) and 70% of inorganic phosphorus (P) losses to inland and coastal water (Leip *et al.*, 2015). In response, a series of governmental policies have been implemented by the European Union (EU) and some of its Member States to improve the utilization of manure nutrients in agriculture and therefore decrease their environment impact (Oenema *et al.*, 2011). These policies have contributed towards the development of manure treatment technologies, which are important for achieving cleaner production in livestock husbandry.

Historically, manure has always been treated and used for various purposes. Attempts to produce biogas from manure date back to the 10th century B.C. (Bond & Templeton, 2011). Efforts to recover specific nutrients or to increase the agronomic value of manure date from the second half of the 20^{th} century (Van der Meer, 1987). Manure has been dried and used as fuel and building material probably as long as there has been animal agriculture. A wide range of new manure treatment technologies have been developed and are now available in Europe. These technologies are considered to be of great importance for the development of sustainable agricultural systems and societies (Foged et al., 2011a; Sommer et al., 2013). Several technologies (e.g. slurry acidification, anaerobic digestion) are used to decrease ammonia and/or GHG emissions from animal manure, and thereby decrease the risk of climate change and acidification of ecosystems. Technologies have been developed to produce renewable energy from manure, for instance, through anaerobic digestion (i.e. biogas production) and incineration (Billen *et al.*, 2015; Kimming *et al.*, 2015). Manure-based bioenergy production decreases CO_2 emissions by substituting fossil fuel for power and electricity production, and therefore is a crucial contributor to the development of bio-economy. Other technologies (e.g. solid-liquid separation, drying, composting, reverse osmosis) have been developed to improve manure handling and transportation characteristics (Sommer et al., 2013). In addition, various manure-based products resulting from these treatment technologies provide opportunities for better nutrient management in agriculture. These products may reduce unnecessary mineral fertilizer use and so the associated resource use and environmental pollution from fertilizer production (Sommer et al., 2013).

Implementation of manure treatment technologies in practice is however limited and regionally scattered in the EU. Less than 10% of the total animal manure production (excluding excreta of grazing animals) was processed in the EU-27 in 2010, with large variations between countries (Foged *et al.*, 2011a). The extent to which treatment technology advances in a country can be influenced by governmental policies and the perceptions of key stakeholders. Environmental policies and legislations vary between EU countries. Although EU Directives set the framework in which all Member States must create legislations directed at civilians/industries to attain the EU-scale objectives, Member States have some flexibility to implement these Directives (Oenema *et al.*, 2011). For example, there is flexibility in the design of national action programs and the use of mitigation measures and techniques in the Nitrates Directive (1991/676/EC) and National Emission Ceiling Directive (2001/81/EC). In addition, differences in farming systems and environmental conditions in the EU, combined with the complexity of manure management and nutrient recycling, can also affect the adoption of treatment technologies (Sommer et al., 2013). To facilitate the proper development of manure treatment technology, there is a need to improve understanding of the reasons for the limited and scattered implementation of these treatment technologies in practice in the EU, especially in regions with high animal density.

While extensive research has been conducted to evaluate the technical, environmental and economic performance of manure treatment technologies in EU, stakeholder opinions regarding the factors influencing manure treatment in practice have not received significant consideration. The diffusion and exploitation of cleaner technologies relies on a combination of factors including governmental policies, financial incentives, technical and service support, and social acceptance (Montalvo, 2008). A better understanding of needs and perceptions of stakeholders from both the supply and demand side is essential to allow for successful innovations for sustainable production and consumption to be shared, spread and scaled up (Blok *et al.*, 2015). The development of manure treatment involves stakeholders across government, industry, academia, extension services and agricultural production sectors.

Integration between policy fields, expert bodies and types of expertise is increasingly required in framing and assessing these EU environmental policies (Kowarsch, 2015). Stakeholders from different sectors may have diverse opinions regarding the objectives of a policy measure as well as on the relevant actions needed to achieve it (Petit & van der Werf, 2003; Van Dam & Junginger, 2011). Policy makers and researchers generally have a broad picture of environmental issues and manure management at regional and national scales. In contrast, the experience of individual farmers is more tied to a particular farm environment, and their decisions are shaped mostly by local socio-economic conditions (Ingram, 2008; Asai *et al.*, 2014). Agricultural advisors have an fair understanding of a group of farmers and their farms through regular contact, enabling them to develop a geographically broad impression of the farming community (Ingram, 2008). Increased understanding among stakeholders involved in the system can help to overcome barriers to the adoption and exploitation of manure treatment technologies.

Few studies have been conducted to investigate stakeholder perceptions of factors influencing the adoption of manure treatment technologies. Examples include studies focusing on composting (Viaene *et al.*, 2016), slurry separation (Gebrezgabher *et al.*, 2015) and anaerobic digestion (Hoppe & Sanders, 2014; Dahlin *et al.*, 2015) in several EU countries. A study in the Netherlands reported that farmer attitudes toward the various properties of manure separation technology were important determinants of adoption. Farmer attitudes were positive towards the agronomic attributes of separation such as the ability to use nutrients (e.g. N and P) in manure optimally, but the economic benefits were generally not appreciated (Gebrezgabher *et al.*, 2015). Barriers to on-farm composting in Belgium were studied based on interviews with stakeholders, which found that strict regulation, considerable financial investment, and lack of experience and knowledge were hindering on-farm composting (Viaene *et al.*, 2016). An analysis of stakeholder perceptions in the biogas production chain in several EU countries indicated that biogas producers and digestate suppliers face many risks and challenges, primarily linked to high financial cost (and sometimes little incentives), legal constraints for operation and market barriers to digestate application (Hoppe & Sanders, 2014; Dahlin *et al.*, 2015). These studies have illustrated that the adoption of manure treatment technology is likely to be affected by a wide range of diverse socio-political, environmental and agronomic factors. There is a need for better understanding of stakeholder perceptions of factors that currently influence manure treatment and also their perspectives regarding successful adoption of these technologies in future.

This study aimed to provide empirical insights into: (1) what stakeholders perceive as important to facilitate or hinder the implementation of manure treatment in practice, (2) stakeholder views of the technologies that have the most potential for successful adoption, and (3) how the preference of technologies with the most potential differs between farm types, farm sizes, and scale of treatment operations. To achieve these objectives, a survey of stakeholders from various groups was conducted in four EU countries: Denmark, Italy, the Netherlands and Spain. All selected countries have large areas of high animal density, but diverse political, farming and environmental contexts.

6.2 Methods

This section includes a description of the countries surveyed (Section 6.2.1), stakeholder categories (Section 6.2.2), the questionnaire structure (Section 6.2.3) and the methods regarding data collection and analysis (Section 6.2.4).

6.2.1 Country selection and context

Denmark (DK), Italy (IT), the Netherlands (NL) and Spain (ES) were selected to represent European countries that have highly-intensive animal production, and as a result, large pressure for manure handling and management (Figure 6-1). Average livestock densities are 1.9 and 3.6 livestock units (LU) per ha of utilized agricultural area in DK and NL, respectively (compared to the EU-27 average of 0.8 LU ha^{-1}). In the north of IT (e.g. Lombardy and Veneto regions) and in some regions of ES (e.g. Catalonia and Murcia regions) livestock densities are also higher than 1.5 LU ha^{-1} (Figure 6-1).

These four countries were also selected because they vary in governmental policies, manure management systems and environmental conditions (Table 6-1). All four countries need to comply with the Nitrates Directive, which aims to protect water quality by promoting good farming practices and preventing the pollution of groundwater and surface waters by nitrate from agricultural sources (including animal manure). The implementation of the Nitrates Directive has had a great influence on manure management (Velthof *et al.*, 2014). The whole territories of DK and NL have been designated as the so-called “Nitrate Vulnerable Zones" (NVZs), while the NVZs cover approximately 21% of total agricultural area in ES and 32% in Italy. Derogations have been granted for specific regions/farms in DK, IT and NL, which allow them to go beyond the limit of 170 kg N ha^{-1} of manure application, while there is no derogation in ES. Renewable energy action plans differ between these countries, e.g. the use of animal manures for renewable energy production (Table 6-1). Soil organic matter is key to soil quality and productivity, and plays a major role in modifying chemical, microbiological and physical properties in ways that improve soil fertility. Mean organic carbon contents in the top soils are < 15 g C kg^{-1} in most regions of ES, while > 30 g C kg^{-1} on average in NL (Reijneveld *et al.*, 2009; de Brogniez *et al.*, 2015). The organic carbon content of the soil may affect decisions about the most suitable use of manure as a source of organic matter to improve soil quality (Diacono & Montemurro, 2010). In DK and NL, manure management systems of dairy cattle are dominantly slurry-based, in contrast to the large fraction of solid-based systems in ES and IT (Table 6-1).

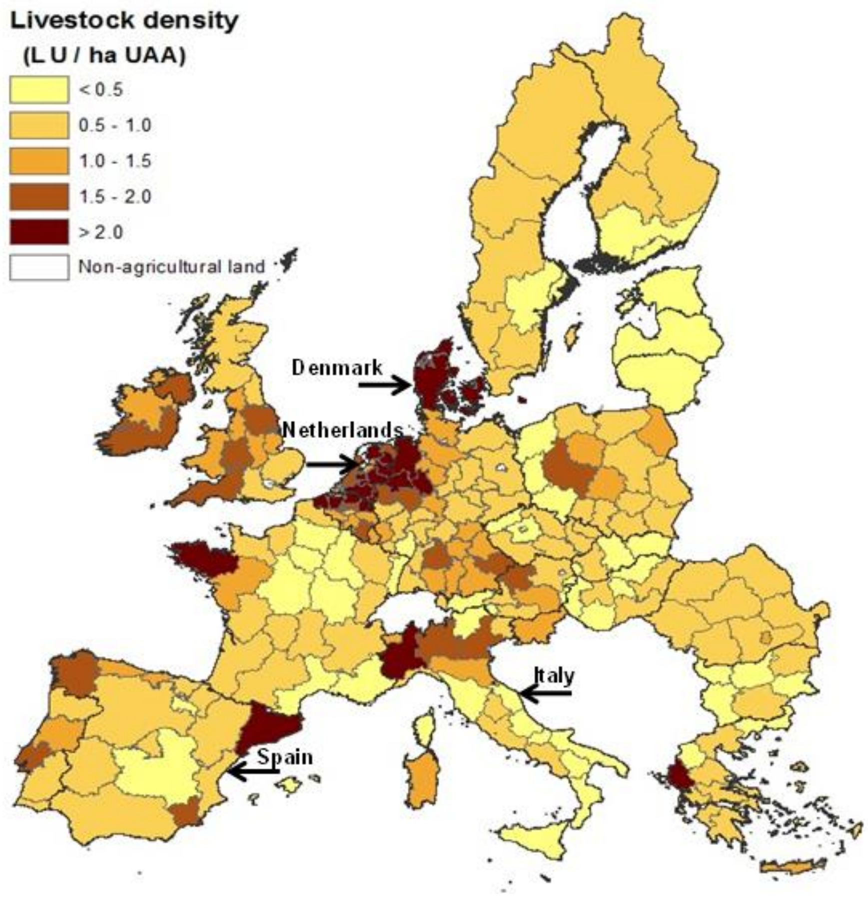

Figure 6-1. Livestock density in the EU-27, expressed in livestock units (LU) per ha utilized agricultural area (UAA). Data were from Eurostat (2010) for the year 2010.

Table 6-1. A comparison of political and agri-environmental characteristics selected for the four European countries.

	DK	NL	ES	IT
Policies				
-Nitrates Directive				
NVZs (% total agricultural area)	100	100	21	32
Derogation granted	Yes	Yes	No	Yes
-Renewable Energy (RE) Directive [a]				
RE from manure in 2006 (ktoe ,kilotonne of oil equivalent)	~70	0	~1.6	n.a.
Estimates in 2020 (ktoe)	~145	~98	~143	n.a.
Agri-environmental conditions				
-Average soil organic carbon in top soils (g kg^{-1}) [b]	20-30	30-40	<15	15-20
-Manure management systems (% of manure N from housing) [c]				
Dairy cow	92% (slurry)	99% (slurry)	70% (solid)	60% (solid)
Other cattle	60% (solid)	83% (slurry)	99% (solid)	60% (slurry)
Pigs	95% (slurry)	99% (slurry)	90% (slurry)	99% (slurry)

[a] Source from National renewable energy action plans; No information (n.a.) available for Italy
[b] Adhikari et al., 2014; de Brogniez et al., 2015; Reijneveld et al., 2009
[c] Information from National inventory reports (NIR) to UNFCCC (the United Nations Framework Convention on Climate Change) for the year 2010.

6.2.2 Stakeholder groups

Six stakeholder groups with expertise in the domain of manure treatment were chosen for this study: (i) livestock farmers; (ii) members of the board of farmers' organizations; (iii) agricultural advisors and consultants; (iv) developers and users of treatment technologies from industry (also including contractors with manure treatment facilities); (v) employees of public authorities (working on the development and control of agri-environmental policies); and (vi) researchers from academic institutions (with expertise in animal manure treatment) (Table 6-2).

6.2.3 Questionnaire design

The survey consisted of 62 questions divided into five sections. Section 1 dealt with respondents' experience in manure treatment. Section 2 related to opinions on factors that stimulate and hinder the implementation of manure treatment in practice. The selection of these factors (presented in the questionnaire) was based on peer-reviewed studies (e.g., Gebrezgabher et al., 2015; Hoppe and Sanders, 2014; Montalvo, 2008) and views of experts (including the authors) in the research of farm-based studies in the surveyed countries. Section 3 aimed to investigate stakeholder opinions about the technologies that have the most potential for successful adoption. Eight common treatment technologies were listed in the questionnaire: solid-liquid separation, acidification, anaerobic digestion, biological nitrogen removal, composting, drying, combustion/ incineration, and membrane filtration/ reverse osmosis (Foged *et al.*, 2011a). For each technology there were four follow-up questions to investigate why, how and where the selected technologies had the greatest adoption potential (considering farm type, farm size and scale of operation, and the benefits of each technology). Section 4 collected demographic information, including employment categories (to distinguish between stakeholder groups) and farm characteristics (in the case of farmers). The final section allowed respondents to submit any other comments and to give contact information (if they wished to receive the results of the study). Respondents could write additional comments and suggestions for each question (under the response 'other').

6.2.4 Data collection and analysis

The survey was performed through both face-to-face interviews and online questionnaires, with support from the academic institutions that participated in the joint EU project ReUseWaste [1] . The electronic version of the questionnaire was designed using SurveyMonkey^{TM}. The questionnaire used for face-to-face interviews was the same as that used for the online survey. Data were collected between April 2014 and June 2015.

Survey dissemination strategies differed between countries. In DK, surveys were disseminated by researchers from the University of Copenhagen via an email that described the purpose and background context of the survey, and included a link to the online survey. Agricultural advisors were contacted via a database of advisors obtained from the Danish agricultural extension service (110 advisors were randomly selected; 32 of them completed the survey, Table 6-2). A list of other stakeholders was prepared via personal contacts. For instance, the questionnaires were emailed to 18 researchers with expertise in manure management and treatment (in Aarhus University, University of South Denmark and the University of Copenhagen), 20 officers in local and national governmental department (e.g. the Danish Environmental Production Agency, the Danish AgriFish Agency), and to the chairmen of 45 farmers' organizations in DK. Similarly to DK, all surveys were disseminated via email and completed online in NL. Requests were sent to target stakeholders (except for researchers) via the secretaries of two large (branch) organizations i.e., CUMELA and Nutrient Platform, and of the main farmers' organization LTO.[2] Furthermore, a selection of 20 researchers from Wageningen UR with expertise in manure management and treatment were asked to complete the questionnaire. In ES, the questionnaire was completed online by

[1] ReUseWaste: http://www.reusewaste.eu/
[2] CUMELA: http://www.cumela.nl/, Nutrient Platform: http://www.nutrientplatform.org/, LTO: http://www.lto.nl/.

stakeholders from the research, industry, extension service and policy communities who were selected and contacted by researchers from the Spanish National Research Council (CSIC). The questionnaire was completed via face-to-face interviews with farmers, instead of by an online survey, as it was considered that farmers would generally have limited access to the internet and were not familiar with online questionnaires. Livestock farmers were visited at their homes (one by one) in regions of high livestock density (Murcia and Catalonia) by researchers from CSIC. These farmers were selected via the contact of local agricultural advisors and also according to their willingness to participate. A hard copy of the questionnaire was presented to the respondents. Interviewers were instructed not to present their own opinions, but only to clarify the questions in case farmers did not understand. Results of the face-to-face interviews were uploaded to the SurveyMonkey™ database and analyzed alongside online responses. In IT, stakeholders from the research, industry, extension service, and policy communities were surveyed during two national agricultural meetings held in 2014 (November and December), and further interviews were subsequently conducted via personal contacts of researchers from the University of Turin. Efforts were made to ensure the privacy of the face-to-face interviews, and respondents were interviewed one by one. Respondents representing livestock farmers and members of farmers' organizations came mainly from areas where animal husbandry is highly intensive, i.e. Northern Italy (Piedmont, Lombardy, Emilia Romagna and Veneto). In total across all the countries 291 surveys were completed; each stakeholder group had between 18 and 75 respondents (see Table 6-2 and Section 6.3).

Table 6-2. Overview of respondents as number per country.

	DK	NL	ES	IT	Total
Do you have experience in manure treatment? [a]					
Yes	73	66	55	45	239
No	9	13	7	23	52
Total	82	79	62	68	**291**
What is your job? [b]					
Farmer	10	18	35	12	75
Representative in a farmer organization	10	3	3	5	21
Agricultural advisor	32	6	1	10	49
Technology developer/ user in company	17	30	4	7	58
Employee in the public authority	8	1	3	6	18
Researcher	7	18	15	23	63
Respondents skipped this question	7	9	3	6	25

[a] single answer
[b] multiple answers

A draft of the survey was sent to researchers (more than 20 in total) in the four countries to improve clarity and reduce the chance of misinterpretation. Comments and suggestions on the draft questionnaires were used to modify the survey before distribution. The same survey was disseminated to the four target countries, but translated (into Danish, Dutch, Italian and Spanish). The link to the English version of the questionnaire (as example for Denmark) was attached in Appendix A.

Data downloaded from the SurveyMonkey™ were compiled and analyzed using R version 3.0.0 (e.g. Crosstab function) and Microsoft Excel 2010. The number of positive ticks to each option of a question (i.e. the number of respondents) was recorded. Results were analyzed by individual countries and also with the sum of all countries. Since there were multiple-response questions in the questionnaire, the absolute number of respondents referring to each answer of a question was converted to the percentage of the total number of respondents who answered the question. This conversion allowed for the comparison of different variables listed in a question, as well as a comparison between countries.

6.3 Results

Table 6-2 provides an overview of the number of respondents per stakeholder group and country. In total, 291 questionnaires were completed: 28% in DK, 22% in ES, 23% in IT, and 27% in NL. A total of 82% of the respondents had experience with manure treatment (Table 6-2). More than 50% of those had experience with manure separation and anaerobic digestion, except for respondents in ES (Figure 6-2). Over 70% of respondents in DK had experience with slurry acidification. In ES, most respondents (40%) had experience with composting. Respondents from NL had more experience with manure drying and membrane filtration (or reverse osmosis) (Figure 6-2). Few respondents answered that they had experience with alternative treatment technologies that were not offered as possible responses in the question, e.g. ammonia stripping from liquid manure, phosphorus recovery, or evaporation of liquid manure.

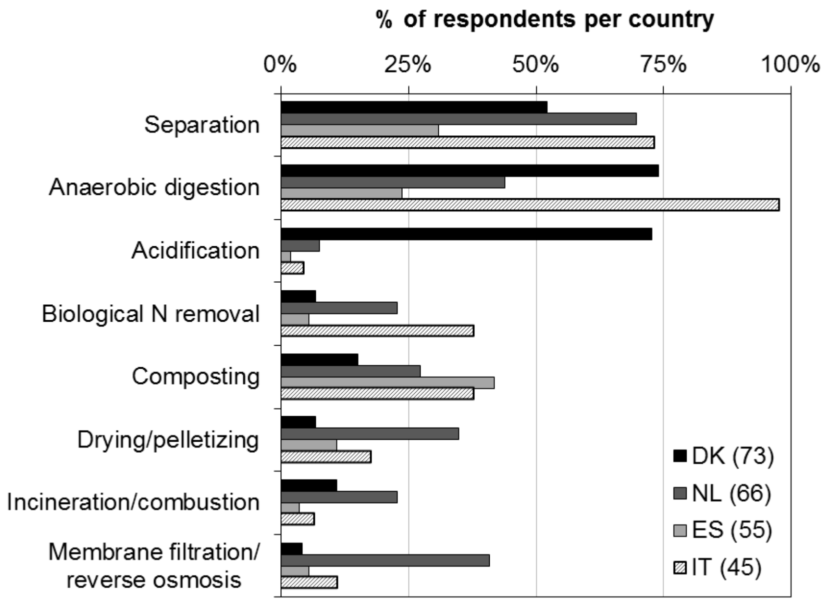

Figure 6-2. Response to the question: "please indicate the treatment technique(s) in which you are involved." (multiple responses permitted). The number of respondents per country with experience in manure treatment is shown in the legend.

6.3.1 Factors that stimulate and hinder adoption

Pressure from environmental policies was perceived to be the most important factor affecting the implementation of manure treatment in practice (70% of total respondents), which was the case for respondents from all four countries and all stakeholder groups (Figure 6-3; Appendix B.1). The need to facilitate the export of manure from the farm (47%, especially in DK and NL) was also highlighted by many respondents. The need to achieve renewable energy targets by producing bioenergy from manure was ranked relatively highly in DK and IT. Compared to the other factors considered, the need to efficiently use manure nutrients due to increased fertilizer prices was considered relatively important in ES. For all countries, controlling diseases, pathogens and odor was considered the least important among the six factors defined in the survey (Figure 6-3).

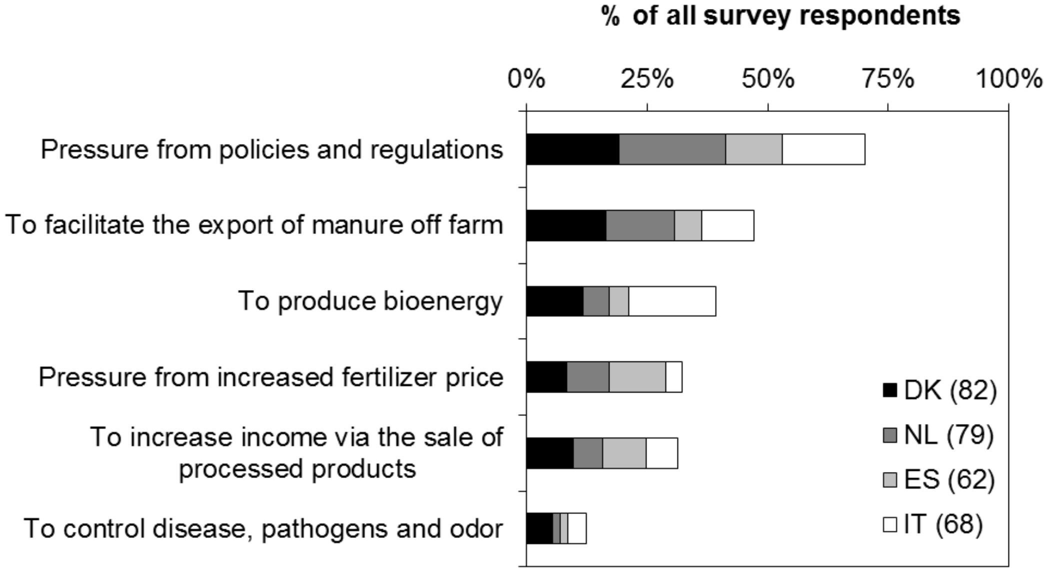

Figure 6-3. Responses to the questions (expressed as % of respondents from all survey countries): "please indicate the top three reasons that can stimulate farmers to apply manure treatment techniques." The number of respondents that answered this question is shown in the legend.

Economic factors were the main barriers to the implementation of manure treatment in practice, namely the lack of investment capital (60% of total respondents), high processing costs (52%), and long payback period (45%). These barriers were perceived to be important for all countries (Figure 6-4) and by all stakeholder groups (Appendix B.2). Legal constraints (32% of all respondents, highest at 45% in NL) and lack of knowledge (32% of all respondents, especially in ES and IT) were chosen by a moderate number of respondents. Transport, noise burdens and health risks were not seen as important barriers among all stakeholder groups (Figure 6-4; Appendix B.2). Interestingly, livestock farmers and agriculture advisors had relatively little concern about the market for manure processing products (Appendix B.2). This suggested that these farmers were possibly interested in using processed organic fertilizers, which is confirmed by the results from a parallel study on farmer perceptions of organic fertilizers in Denmark (Case et al., unpublished results).

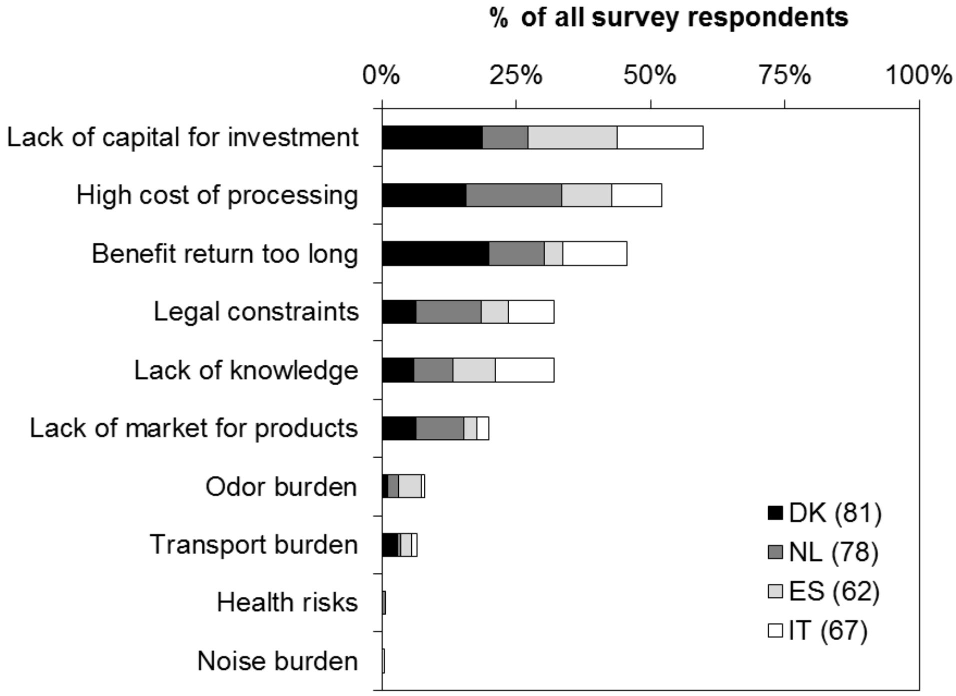

Figure 6-4. Responses to the questions (expressed as % of respondents from all survey countries): "please indicate the three most important constraints / barriers to the adoption of manure treatment technologies." The number of respondents that answered this question in each country is shown in the legend.

6.3.2 Preferred treatment technologies

Stakeholders indicated that manure separation and anaerobic digestion had the greatest potential for a common adoption in practice (36% and 42% of total respondents, respectively). Other technologies appear to be more country specific. There was a relatively high adoption potential for slurry acidification in DK (47%) and composting in ES (44%), while drying of solid manure fractions and membrane filtration (or reverse osmosis) of liquid fractions were considered positively in NL (Figure 5-5).

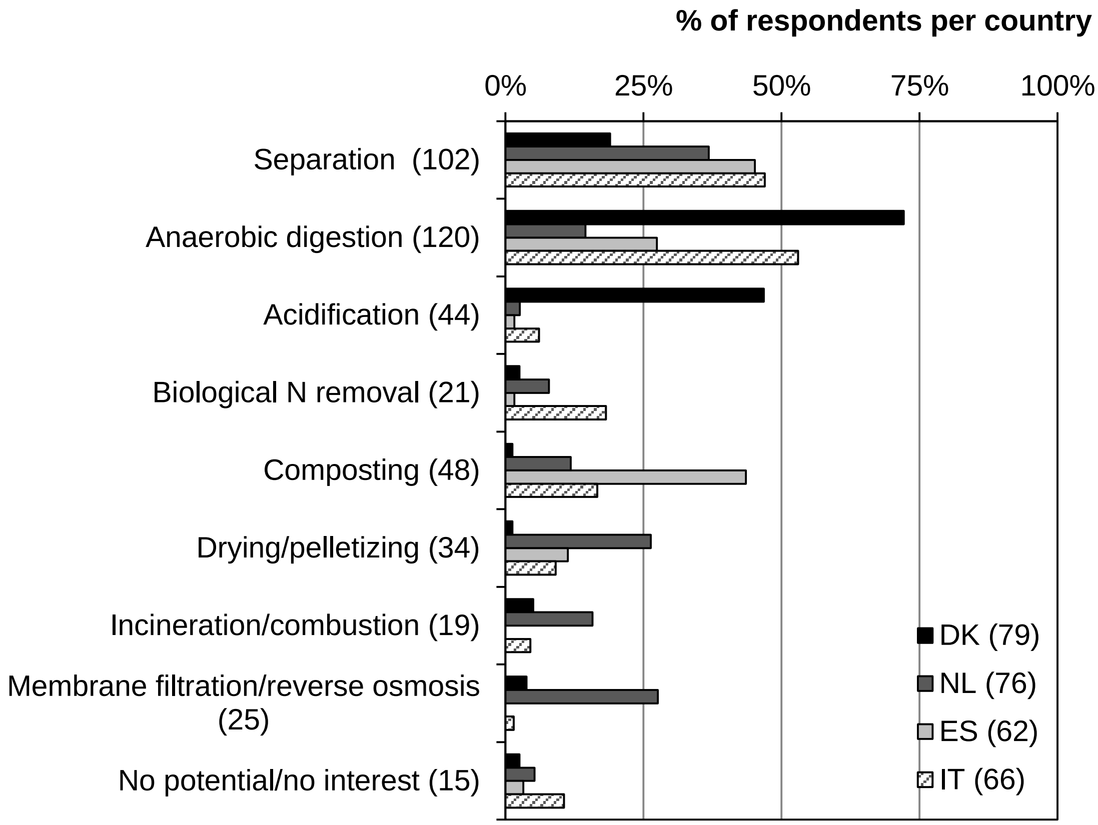

Figure 6-5. Response to the question: "which techniques have the most potential to be applied in your country during the next 10 years?" The number of respondents (answered this question) for each country is shown in the legend. The number of respondents for each technology/answer is indicated in the Y-axis label.

6.3.3 Preferred farm structure and scale of operation

Figure 6-6 shows that livestock farms with a limited area of land were considered to have a relatively high adoption potential for all of the manure treatment technologies considered with the exception of slurry acidification (Figure 6-6a). This exception is possibly due to the fact that farms with sufficient land are more willing, or are required to use techniques that reduce ammonia losses from on-farm storage and application of manures. Overall, manure treatment was considered to be more applicable to pig and cattle farms than to poultry farms (Figure 6-6a).

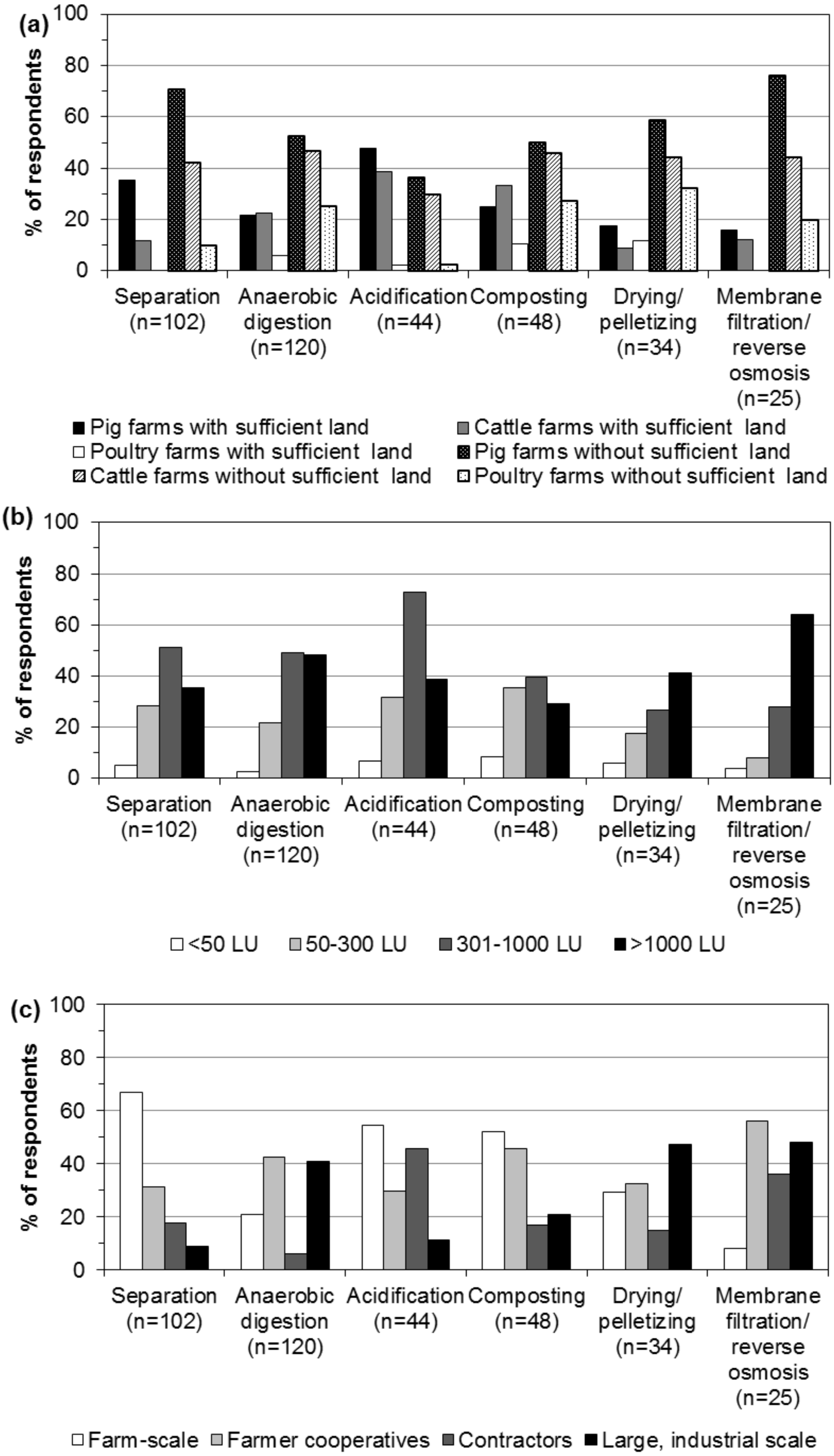

Figure 6-6. Responses to indicate which farm types (a), sizes (b, LU=livestock unit) and operation scales (c) have the most potential for adoption of respective technologies (multiple answers), expressed as % of respondents for all four countries. The number (n) of respondents is shown for each technology. Results referring to biological nitrogen removal and incineration treatment are not shown due to limited number of responses.

Manure treatment was considered to be less applicable to small livestock farms (i.e. <50 LU). Drying and reverse osmosis technologies were perceived most appropriate for large livestock farms (> 1000 LU) (Figure 6-6b). Stakeholders had different views regarding the optimal scale of the manure treatment plant (Figure 6-6c). Separation (67% of respondents), acidification (55%) and composting (52%) were perceived to be most applicable at the farm scale. Anaerobic digestion, drying (pelletizing) and membrane filtration were considered to be most applicable at the industrial scale and for farmer cooperatives (Figure 6-6c).

6.3.4 Benefits of manure treatment

Table 6-3 shows respondent perceptions of the benefits of manure treatment. A reduction in manure disposal costs and an increase in the fertilizer value of separated liquid and solid fractions were ranked as the main benefits of manure separation. For anaerobic digestion, the main benefits included bioenergy production, the increased fertilizer nitrogen value of digestate, and the reduction of odor and gaseous emissions during further processing and field application. Mitigation of ammonia emissions during slurry storage and application, and the increased fertilizer N value of slurry were ranked as the main benefits of slurry acidification. Increased organic matter quality of manure and improved soil quality after field application were ranked as the main benefits of composting.

6.4 Discussion

Currently, less than 10% of the animal manure produced in EU is treated and most farmers have little knowledge about manure processing technologies (Foged *et al.*, 2011a). In the present study the survey was disseminated to stakeholders involved in manure treatment, directly or indirectly. This explains why over 80% of the respondents described themselves as having at least some experience with manure treatment. Most of the stakeholders contacted within each group (farmers, farmers' organizations, extension service, industry, policy and research) were considered to be forerunners in the whole domain of the development, implementation and management of manure treatment technologies. By exploring the views of these stakeholders engaged with manure treatment, a better understanding of the future perspectives of manure processing is possibly achieved.

Table 6-3. Summary of responses to the questions asking about the benefits of each respective technology (for all four countries), measured in % of the total number of respondents for each question.

	% of respondents
What are the top three benefits of separation? (Number of respondents: n =102)	
To reduce cost of manure disposal	47
To increase fertilizer value of liquid fractions	39
To increase fertilizer value of solid fractions	34
To use solid factions for biogas production	27
To use solid fractions for composting	25
To reduce ammonia emissions from liquid fractions after field application	18
To use solid factions as bedding materials	16
What are the top three benefits of anaerobic digestion? (n=120)	
To produce bioenergy	88
To increase fertilizer nitrogen value of digestate	58
To reduce odor and gaseous emissions during processing	43
To reduce odor and gaseous emissions after field application of digestate	42
To increase soil quality after field application of digestate	13
To increase fertilizer phosphorus value of digestate	8
What are the top three benefits of acidification? (n=44)	
To reduce ammonia emissions during field application	82
To reduce ammonia emissions during storage	73
To increase fertilizer nitrogen value of slurry	68
To increase fertilizer sulfur value of slurry	27
To reduce greenhouse gas emissions during storage	25
What are the top three benefits of composting? (n=48)	
To improve the organic matter quality	54
To remove pathogens	46
To reduce the volume and mass of the manure	42
To improve soil quality after field application of compost	42
To increase economic value as compost products	40
To reduce ammonia emissions after field application of compost	19
To homogenize the manure	13
What are the top three benefits of drying/ pelletizing? (n=34)	
To facilitate export	59
To increase the market value of the manure	53
To reduce costs of transporting manure surplus off farm	41
To increase soil quality after field application of dried products	29
To decrease ammonia emissions after field application of dried products	26
What are the top three benefits of membrane filtration/ reverse osmosis? (n=25)	
To increase fertilizer effectiveness of nitrogen as concentrates	64
To make a K fertilizer	48
To reduce cost of transporting phosphorus surplus off farm	44
To remove organic matter from liquid manures	28
To reduce ammonia emissions after field application of concentrates	12

6.4.1 Key factors that stimulate manure treatment in practice

Pressure from environmental policies and regulations was identified as the most important stimulus for the implementation of manure treatment systems (Figure 6-3). This may reflect the fact that current policies and regulations implemented in these four countries have influenced stakeholder decisions on manure handling and management activities. A number of policies have been implemented by the EU and United Nations (UN) bodies to reduce environmental pollution from animal manures (Oenema *et al.*, 2011), which play an important role in stimulating manure treatment activities in Europe. The EU Nitrates Directive sets up the maximum application limit of manure in NVZs, equivalent to 170 kg N ha^{-1} $year^{-1}$ (European Commission, 1991). This limit obliges livestock farms to treat and/or to transport the excess manure to other farms. The EU National Emission Ceiling Directive (European Commission, 2001) aims to reduce emissions of ammonia (NH_3) (including from manures), and thereby stimulate the development of certain manure treatment technologies (Bittman *et al.*, 2014). For example, acidifying slurry was introduced as one of the options for obligatory NH_3 mitigation measures by Danish regulations in response to these EU Directives. On the other hand, the use of manure treatment may remain marginal in regions that have low pressure from these regulations. The authors conducted also a similar survey in Portugal, but the number of responses from targeted stakeholders was small and hence results are not shown. A low response rate from Portuguese stakeholders (in particular farmers) may reflect that the interest for manure treatment is low in regions that have sufficient land for application of the manure produced, as well as low pressure from governmental legislation. These results revealed that variations between countries in manure treatment have a strong relationship with variations in livestock density and national policies.

Producing bioenergy from animal manures was identified as an important reason for the adoption of manure treatment in practice, in particular in DK and IT (Figure 6-3). Anaerobic digestion produces biogas that can be used directly for heating, for combined thermal and electricity generation, or to upgrade to bio-methane that has similar characteristics to natural gas (Bernet & Béline, 2009). Using animal manures as feedstock for biogas production has advantages compared to using energy crops, such as less competition with food production and higher mitigation potential of greenhouse gas emissions (De Vries *et al.*, 2012). Further, the digestate can serve as an improved organic N fertilizer (Table 6-3). The development of biogas production in European countries has been influenced strongly by environmental regulations and the EU Renewable Energy Directive (Edwards *et al.*, 2015). The growth of anaerobic digestion in DK is largely due to policy incentives such as increased investment support for construction of biogas plants, the implementation of fossil energy taxes or renewable energy tariff subsidies and the government support strategies to increase interactions between various social groups (Raven & Gregersen, 2007). Italy has also witnessed an extraordinary growth in biogas generation from animal manures and other agricultural biomass in the last few years, which is largely due to the biogas support programs implemented in Italy (the introduction of Tradable Green Certificate and feed-in-tariff, and increased investment subsides) (Chinese et al., 2014). In comparison, manure-based biogas producers in NL and ES face many financial and socio-political challenges (Fierro *et al.*, 2014; Hoppe & Sanders, 2014), which may explain why biogas production was perceived as less attractive in these two countries (Figure 6-3 and Figure 6-4).

The pressure from increased fertilizer price was perceived to be an important factor for stimulating manure treatment in ES (Figure 6-3), which is in line with the conclusion from a study that investigated the existing experience on manure treatment in Catalonia, a region with high animal density in ES (Flotats *et al.*, 2009). The increase in prices of mineral fertilizers could explain the recent growth in composting facilities in Catalonia, in order to recover nutrients in organic forms and produce soil organic amendments that are economically valuable (Flotats *et al.*, 2009). The need to facilitate off-farm manure export was considered to be relatively important in NL and DK, where the average LU is high and a large portion of farms have been involved in manure exchange (Asai et al., 2014); it appears to be less important in ES partly due to the average low animal density (Figure 6-1).

6.4.2 Key barriers to manure treatment in practice

The most important barriers to the implementation of manure treatment in practice were related to economic factors (Figure 6-4). This corresponded with findings from several other studies. Results from a survey among 111 Dutch dairy farmers indicated that nearly half of respondents strongly disagreed with the statement that low cost of manure separation is a reason for them to consider the use of manure separation, while only 13% of respondents agreed (Gebrezgabher *et al.*, 2015). Substantial upfront investments, subsidies not being granted, and increased price of co-feedstock were identified as important barriers for biogas producers in NL (Hoppe & Sanders, 2014). In the present study, most respondents (who perceived that anaerobic digestion had the most potential for adoption) stated that subsidies for upfront investment and/or energy production were vital for anaerobic digestion of animal slurries in practice (data not shown). This confirms results from previous studies that subsidies play a large role in the profitability of biogas plants (Gebrezgabher *et al.*, 2010; Chinese *et al.*, 2014; Riva *et al.*, 2014).

A number of respondents brought up legal constraints as an important issue hindering the implementation of manure treatment (Figure 6-3). A Dutch respondent indicated that "Licensing can be very restrictive in realizing initiatives, due to lack of objective knowledge (on manure processing) among local residents and licensing authorities". Likewise, a stakeholder study indicated that legal permits to operate biogas plants were difficult to attain in NL, partly because municipalities did not yet have specific biogas polices in place and therefore there were few staff trained in how to deal with permit requests for co-digestion plants (Hoppe & Sanders, 2014). A Danish respondent also stated that "It is difficult or impossible to get authority approval for treatment operations, because of the resistance of the local community". Therefore, outreach strategies should be developed to provide more information to local residents, authorities, and extension services regarding the benefits and risks of manure treatment so as to increase social acceptability.

6.4.3 Differences in priorities of technology adoption and operation structure

The choice of prioritized technologies generally corresponded with the technologies for which respondents had experience (see Figure 6-2) and the status of manure processing activities in the countries surveyed (Foged *et al.*, 2011a). An EU inventory study reported that slurry separation was used most in IT and ES; anaerobic digestion was predominantly applied in Germany, followed by IT and DK; and acidification operations were mainly located in DK, while ES had the largest share of composting operations (Foged *et al.*, 2011a). In this study, composting was identified to have considerable growth potential in ES (Figure 6-4). This is partly because of the low soil organic matter content of arable land in ES (< 15 g C kg^{-1}; de Brogniez et al., 2015) and the ability to improve soil quality following the application of compost (Bernal *et al.*, 2009). Composting was not ranked highly in DK and NL, where soil organic matter contents are relatively high (de Brogniez et al., 2015). Solid-liquid separation, drying of solid fractions and reverse osmosis of liquid fractions (to concentrates) were considered as attractive technologies for livestock farms with a limited area of land in NL (Figure 6-4). This may have been chosen due to the need to comply with policy regulations. Obligatory manure treatment was introduced in NL in 2013, which designated that livestock farms with a manure surplus have to treat and/or to export a certain percentage of the surplus. Thus, the need to transport manures can greatly increase the use of treatment technologies that reduce the volume of liquid (separation and reverse osmosis) and solid fractions (drying and pelletizing). Manure-based anaerobic digestion was prioritized in DK (Figure 6-4), mainly due to Danish government policy. The Danish government proposed a target of using 50% of the manure produced in DK for renewable energy by 2020, which would need to be met through a strong expansion of biogas plants and capacity (Danish Ministry of Food, Agriculture and Fisheries, 2009).

Farm size and treatment plant operation structure are important for the implementation of manure treatment technologies (Flotats et al., 2009; Gebrezgabher et al., 2015). Clearly, land-limited large farms with >300 LU (representing farms with high animal density) have larger potential (or need) for the adoption of manure treatment than small farms (Figure 6-6). Separation and composting were generally considered to be farm-scale treatment technologies, while manure drying and reverse osmosis were considered most applicable at large, industrial scales (Figure 6-6). The complexity of the management and the costs of investments and processing varied among treatment technologies (Foged *et al.*, 2011b). This may explain why the potential and suitability of technology adoption is related to the scale of farm and plant operations. Solid-liquid mechanical separation and composting are generally considered to be less complex in operation and of relatively low cost, compared to treatments such as anaerobic digestion and reverse osmosis (Flotats *et al.*, 2009; Foged *et al.*, 2011b). The annual gross costs (investment and operational costs) vary from 0.5-3 € t^{-1} of inputs for mechanical separation and slurry acidification to 8-14 € t^{-1} for anaerobic digestion and reverse osmosis, and their net processing costs on the basis of total N treated vary from 0.15-3 € kg^{-1} of N (Møller *et al.,* 2000; Foged *et al.,* 2011b). Processing manure in a cooperative form has advantages to reduce financial risks (to individual farmers) and treatment costs, and to make manure treatment viable for small- and mid-sized farms (Møller *et al.*, 2000; Flotats *et al.*, 2009; Swindal *et al.*, 2010).

6.5 Conclusions and recommendations

Understanding the opinions of stakeholders closely engaged in manure treatment can enhance the effectiveness of programs designed to stimulate diffusion and exploitation of these technologies. Such an understanding is an essential part of attaining EU environmental and renewable energy targets. Based on the main findings from the present study, policy requirements, outreach strategies and future research needs are suggested.

Policy requirements. Pressure from governmental legislation was identified as the key stimulant of technology adoption, while barriers to adoption were mainly related to economic factors. It is recommended that policies for the promotion of manure treatment must be economically appealing to attract new adopters (farmers and industries). Long-term financial support schemes (e.g. subsidies) must be developed to encourage potential adopters to invest, considering the long-term investment requirements of manure treatment. It is also necessary to improve permit request procedures to facilitate their acquisition for operations. Large variations in technology preference between countries, farm types and scale of operation were observed in this study. These variations need to be considered when developing policy support schemes and marketing strategies.

Outreach strategies. More information should be conveyed to livestock farmers (especially those with large, land-limited farms) and other technology users regarding the different aspects of a specific technology, i.e. financial viability, optimal operation conditions (e.g. farm size, operation scale), regulations and incentives, and the agronomic and environmental performance of the technology. Better dissemination of this information to users would alleviate the lack of knowledge and experience and thus to assist with their decisions on technology adoption. Resources should be allocated to enable face-to-face, direct mail contact, as well as internet sources for dissemination of information. Outreach strategies need to be developed to convey these important environmental benefits of manure treatment to local residents so as to increase social acceptability.

Future research needs. This study emphasizes the importance of understanding stakeholder perceptions in countries with large areas of high animal density where manure treatment should be prioritized. However, manure treatment should not be limited to these regions, considering the potential benefits of manure treatment (e.g. not only environmental but also agronomic benefits). Thus, future research addressing the perceptions of stakeholders in regions with contrasting farming systems and socio-political conditions will complement the present findings and provide a more complete picture of the development of manure treatment.

Understanding stakeholder opinions about the development of manure treatment can assist in the design of policies and outreach strategies, leading to a better use of animal manures and a sustainable production and management chain. The results from this study can serve as a basis for such efforts.

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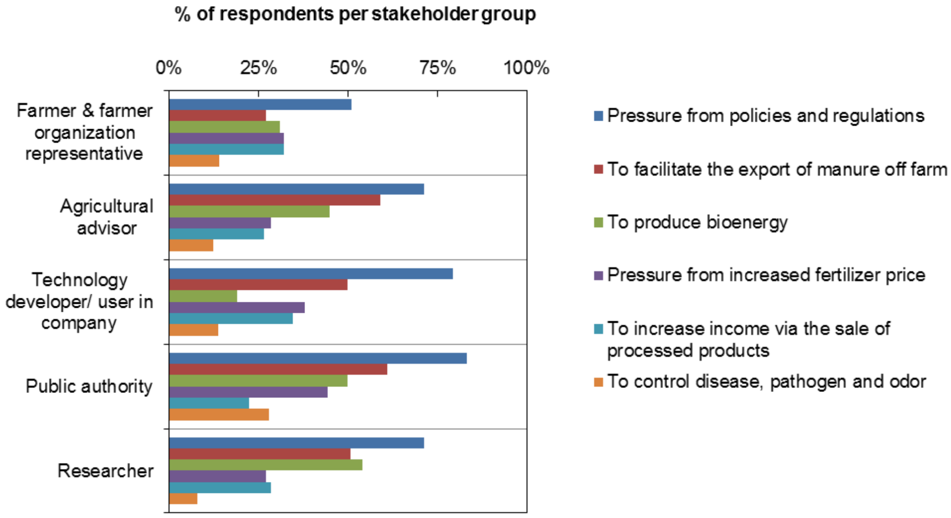
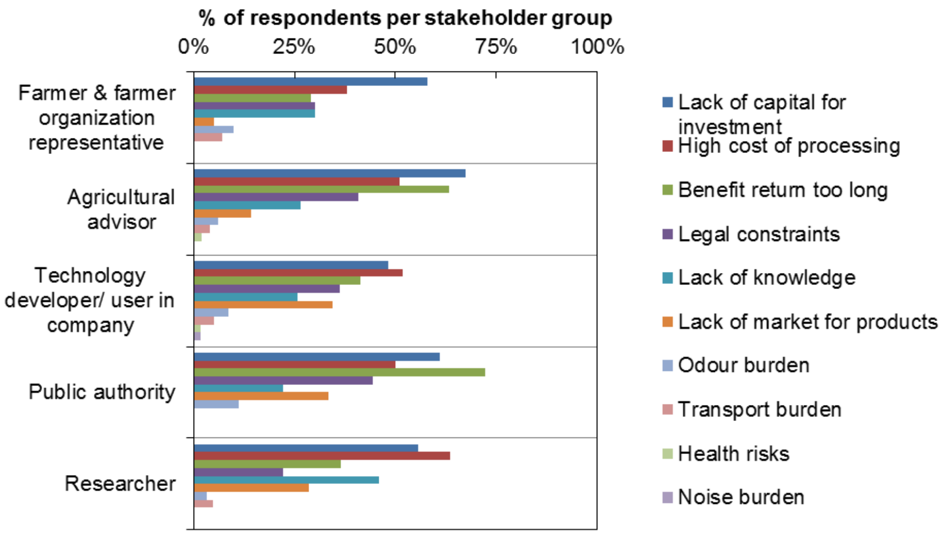
Appendix A: The link to online questionnaire (as example for Denmark):

https://www.surveymonkey.com/r/reusewasteDK_EN.

Appendix B:



B.1. Responses to the question (multiple responses permitted): "please indicate the top three reasons that can stimulate farmers to apply manure treatment technologies." (presented as % of respondents per stakeholder group, for all survey countries). Due to a relatively low number of responses from farmer organizations, this stakeholder group was aggregated with the farmer group.



B.2 Responses to the question (multiple responses permitted): "please indicate the three most important constraints/ barriers for adoption of manure treatment technologies." (presented as % of respondents per stakeholder group, for all survey countries). Due to a relatively low number of responses from farmer organizations, this stakeholder group was aggregated with the farmer group.

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CHAPTER 7

General discussion

7.1 Introduction

Animal manures are major sources of plant nutrients and soil organic matter. However, when not properly managed, these manures release also considerable amounts of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) into the air, and nitrogen (N) and phosphorus (P) losses to water bodies via leaching and runoff, which create a range of unwanted environment impacts. In response, a large number of policy measures and manure treatment technologies have been developed, and part of these have been implemented in practice. Introducing a measure or technology to mitigate emissions from one source in the manure management chain may affect emissions also downstream in the chain, and may affect emissions of other pollutants, which may lead to synergistic or unwanted environmental side-effects (Sommer *et al.,* 2009; Velthof *et al.*, 2009). The trade-offs and co-benefits of emission mitigation measures and manure treatment technologies are poorly understood, especially when considering the whole manure management chain. Moreover, the effects of combinations of measures and technologies have not been well analyzed yet at regional and national scales.

The overall objective of my PhD thesis research is (i) to enhance the quantitative insight into the effects of emission mitigation measures and treatment technologies on the emissions of NH_3, N_2O and CH_4, and the recovery of N and P from animal manure in the whole chain from animal feeding to manure application to land, and (ii) to explore the effects of combinations of measures and technologies to mitigate these emissions consistently. This PhD thesis research was part of the Marie Curie Training Program ReUseWaste (Recovery and Use of Nutrients, Energy and Organic Matter from Animal Waste). In total eleven PhD students and two post docs have been working on various specific mitigation measures and treatment technologies. My research focused on the integral analysis of the whole manure management chain, and on the up-scaling and synthesis of research results, including those from literature. My research was split-up in four parts:

i) To allow for sound comparisons between countries and estimates at EU level, a harmonized and transparent methodology has been developed for the estimation of feed use and N and P excretion by the main animal categories, which is an essential first step for estimating manure-sourced emissions throughout the manure management chain (Chapters 2 and 3).

ii) To provide quantitative insights into the possible side-effects of NH_3 emission mitigation measures, effects of these measures on emission of NH_3, N_2O and CH_4 at individual stages of the chain were assessed, using a meta-analysis approach. Further, the whole-chain impacts of a suite of NH_3 mitigation measures on these emissions were explored for alternative manure management systems through scenario analyses, with emission parameters derived from the meta-analysis (Chapter 4).

iii) To estimate the environmental effects of treatment technologies in EU, gaseous emissions (NH_3, N_2O and CH_4) and the recovery of N and P from animal manure in EU were assessed at country levels for the year 2010, using the improved MITERRA-Europe model, supplemented with a ‘manure treatment’ module developed in this study. Mitigation potentials of various treatment technologies and the associated impacts on nutrient recycling of manure in the EU were further examined through scenario analyses (Chapter 5).

iv) To increase the understanding related to the factors that affect the adoption of treatment technologies in practice, a survey of stakeholder perceptions of manure treatment technologies was conducted in four EU countries. All these countries have regions with high animal density, but diverse social-economic and environment conditions (Chapter 6).

In this chapter (Chapter 7), the main findings of my PhD thesis research are highlighted and discussed in a broader context. This chapter ends with a conclusion section, and a section in which future research needs are suggested.

7.2 Main findings

- In the EU-27, the total amounts of N and P in animal excretion produced annually are as large as or larger than the total annual amounts of fertilizer N and P used during last decades. However, there is a huge spatial variation in manure production. The methodologies used for estimating N (and P) excretion factors of animals differ between countries, and may also differ within countries between different policy reports. These differences hamper a sound comparison between countries and lead to uncertainties in total manure production and estimated emissions (Chapter 2). In an attempt to increase the accuracy of manure production estimates, I have developed a transparent and uniform methodology that couples feed availability in a country to N and P excretion rates at animal category and national levels, for all countries of the EU-27. This coupling allows to make checks of the plausibility of nutrient excretion values at the country level, and to arrive at a common basis for the estimation of the production of manure N and P, nutrient balances and emissions across the EU (Chapter 3).

- Increasing the effectiveness of mitigation measures for NH_3 emissions from animal manure requires a manure management chain approach (Chapters 4 and 5). Lowering the crude protein (CP) content in animal feed (by an absolute value of about 1% to 5%) significantly decreased NH_3 emissions from animal housing (by 24-65%, compared to the reference), and decreased emissions also from other management stages of the chain. External slurry storages with covers of straw and artificial films decreased NH_3 emissions from storages by on average 78 and 98%, respectively. This measure increases ammonium content of manure and, because of that, requires low-emission manure application measures to prevent increased downstream emissions. Low-emission manure application through incorporation and injection of manures in the soil may reduce NH_3 emissions by 70 to 80%, but tend to increase N_2O emissions. The manure N that is conserved by using mitigation measures can be used to fertilize crops and to replace equivalent amounts of fertilizer N. Measures taken to decrease NH_3 emissions have relatively small effects on emissions of CH_4 from the manure management chain, with the exception of slurry acidification. Slurry acidification significantly decreases emissions of NH_3 and CH_4 from slurry storages (83 and 87%, respectively), which decreases total GHG emissions from systems with acidified slurry. Proper combinations of mitigation measures at farm level are therefore important to minimize impacts of livestock production on emissions of NH_3 and GHG (Chapter 4).
- Effects of manure treatment on NH_3 and GHG emissions from animal manures were relatively small in the EU-27 in 2010. Manure treatment have contributed to deceases in GHG emissions from manure by 0-17% depending on countries (compared to situations without manure treatment in 2010), with the largest contributions from anaerobic digestion (especially in Germany, Denmark and Italy). Scenario analyses indicated that increasing the implementation of slurry acidification, thermal (forced) drying, incineration and pyrolysis (implemented at a level of 20% of total manure production from housings for each technique) may decrease both NH_3 (9-11%) and GHG (11-18%) emissions from the manure management chain in the EU-27. Nitrification-denitrification treatment decreased NH_3 emissions (8%), but increased GHG emission (6%) due to increased N_2O emissions. Solid-liquid separation (8-12%, depending on separators) and anaerobic digestion (19%) deceased GHG emissions, while the effects on whole-chain emissions of NH_3 were small. Combining acidification with separation or with anaerobic digestion (acidifying digested liquid

fraction) decrease both GHG (18-20%) and NH_3 (7-10%) emissions from the manure management chain. Composting marginally affected total NH_3 emissions; emissions increased during composting and decreased emissions from application of compost to land. Composting also not affected GHG emissions significantly; there is a large variability/uncertainty in GHG emission factors during composting (Chapter 5).

- The estimated amounts of N and P in manure applied to land, expressed as percent of N and P excreted in animal housings in EU-27 in 2010, were 57 and 98%, respectively. When increasing the implementation of treatment technologies (to an equivalent of 20% of total manure production in housings in all countries of the EU), the N recovery fraction would increase to 61% by acidification, but would decrease to 48% by incineration and to 52% by nitrification-denitrification treatment. Other technologies (solid-liquid separation, anaerobic digestion, drying, composting etc.) would only marginally affect the N recovery. Changes in the total P recovery due to all these technologies were relatively small, but there are likely differences between manure treatment products in P availability, which was not examined in my thesis. The N/P ratio of liquid manure products from treatment technologies varied from 3.4 to 10.7, compared to a mean of 3.5 in untreated slurry. For solid manure products, the N/P ratio ranged from 0.1 to 3.2, compared to 3.0 in raw solid manure in the reference. Production of manure products that vary in N/P ratio provides opportunities to better use manure nutrients and to better meet nutrient demands by crops (Chapter 5).
- Pressure from governmental regulations is a key factor that stimulates the development and adoption of manure treatment in practice in countries that have regions with high livestock density (Denmark, Italy, Spain, the Netherlands). The major barriers to technology adoption were related to economic factors, while there was relatively little concern regarding transport and noise burdens and health risks (Chapter 6).
- Slurry separation and anaerobic digestion were generally perceived to have the greatest potential for a common adoption in practice in all four countries. Other preferred technologies were more country-specific. Stakeholders had different views on the optimal scale of the manure treatment plant because of the differences in investment cost and complexity of treatment technologies. Manure treatment will remain a regional activity because of large differences between regions/nations in livestock densities and socio-economic, political and environmental conditions. To encourage the adoption of manure treatment, policies must be economically appealing

to attract new adopters (farmers and industries); long-term financial support schemes (e.g. subsidies) seem to be necessary. Outreach strategies are required to convey the knowledge to stakeholders from both the supply and the demand side, with respect to the economic, technical and environmental aspects of treatment technologies (Chapter 6).

7.3 The manure management chain approach

Measures and technologies taken to mitigate emissions from animal manures have to be optimized while taking the whole manure management chain into account (Petersen *et al.*, 2007). The 'chain approach' allows for the analysis of the consequences of a technology, implemented at one stage of manure management, on the emissions at other stages, and also on emissions of other pollutants. The manure management chain concept (Figure 7-1) includes all main sectors, namely animal feeding, animal grazing and housing, manure storage and treatment, and application of manure to land. There are complex nutrient transfers and transformations within the chain.

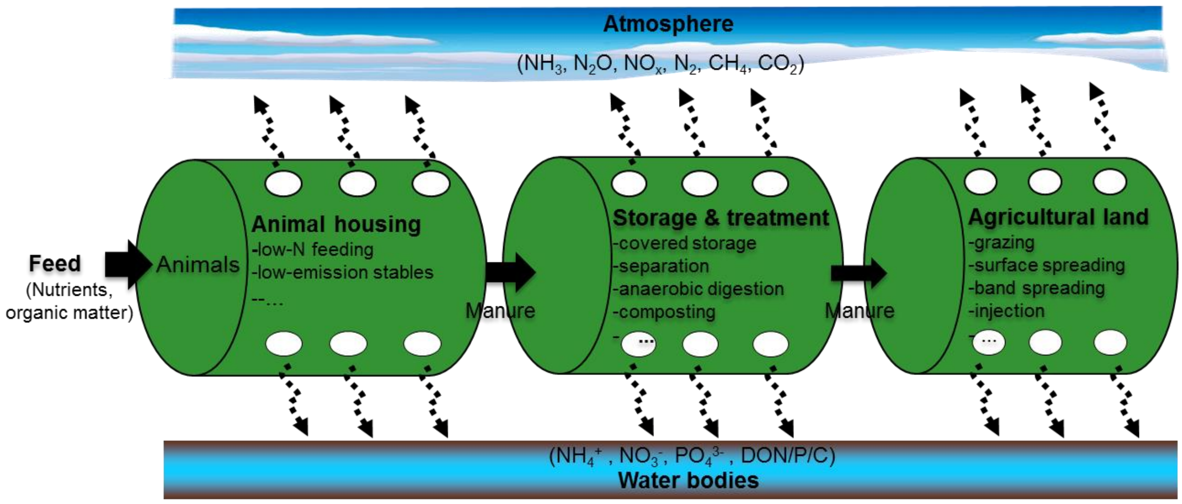

Figure 7-1. A simplified representation of different stages (including management practices) in the manure management chain. Dashed arrows show possible losses of nutrient elements to the atmosphere and to water bodies.

Several studies have been conducted to analyze gaseous emissions (NH_3, N_2O and/or CH_4) from animal agriculture in general and animal manure in particular in EU countries, including e.g., Denmark (Hutchings *et al.*, 2001), France (Gac *et al.*, 2007), Ireland (Hyde *et al.*, 2003), Switzerland (Reidy *et al.*, 2008; Kupper *et al.*, 2015), The Netherlands (Velthof *et al.*, 2012), UK (Webb & Misselbrook, 2004). These analyses were generally based on a mass flow approach. By using such approach, the consequences of abatement measures can be assessed at various stages of the manure management chain. However, it is difficult to make proper comparisons between countries, because these studies often used different system boundaries, emission factors and calculation methodologies.

Life cycle assessment (LCA) is an effective approach to analyze a wide range of environmental emissions along the entire life cycle of a product. LCA has been increasingly applied in environmental assessments of animal production (e.g. milk, pork and eggs) during the last decade. More recently, LCAs of treatment technologies have been conducted, such as for solid-liquid separation (ten Hoeve *et al.*, 2014), anaerobic digestion (Sandars *et al.*, 2003; Hamelin *et al.*, 2011; De Vries *et al.*, 2012; Mezzullo *et al.*, 2012) and pyrolysis and combustion processes (Fernandez-Lopez *et al.*, 2015). These LCAs were carried out in general at farm-scale, and the challenge therefore remains to up-scale the findings to regional/national levels. These LCAs often focus on a specific treatment technology or a specific type of manure only. The focus in my study was on the national and EU scale, considering the major manure types and the most used treatment technologies. Assessment was made with the MITERRA-Europe model.

The MITERRA-Europe is an integrated environmental assessment model, which quantifies all N and P flows as well as GHG emissions in agriculture for all member states of the EU at regional and national scales, using a uniform method and consistent dataset (Velthof *et al.*, 2009; Lesschen *et al.*, 2011). The model is partially based on information from the GAINS model (Asman *et al.*, 2011) and CAPRI model (Britz & Witzke, 2012). To fulfill the objectives of my PhD research, the model MITERRA-Europe was further improved. The main improvements are as follows: i) N and P excretion are quantified using the feed balance approach at animal category and country levels; ii) the estimation of possible side-effects of mitigation measures has been improved; iii) the environmental effects of a range of manure treatment technologies can be analyzed simultaneously at national scales now; iv) the recovery of N and P from animal manure can be quantified now; these recoveries are sensitive to changes in management along the chain.

There is scope for further improvement of the MITERRA-Europe model.

- Regional specific data on the implementation of mitigation measures are required to decrease the uncertainties of the model outcome. However, such data are not easy to obtain. The EU-wide 'Survey on agricultural production methods (SAPM)' was a one off survey in 2010 to collect farm level data on agri-environmental measures to support monitoring of the relevant European Union policies (http://ec.europa.eu/eurostat). The SAPM data have been assessed for using them in the scenario analyses in Chapter 5. However, some flaws and inconsistences were found in the database, which hamper its proper use. For example, i) animal housing regarding the category 'other' is not clearly defined; ii) housings are not distinguished between dairy cows and other cattle; iii) definitions of manure storage facilities are too complex to be properly interpreted; iv) data on application of manure are not detailed at animal category levels. Evidently, it is difficult to use this SAPM dataset for modeling purposes.
- In the MITERRA model, it was assumed that all manure treatment activities take place as soon as manures are produced by animals or after removal from the housings. However, this is not always the case in practice. Often, there is a period of storage prior to treatment, but there is currently no information available about possible interim storage. As a consequence, the effectiveness of manure treatment may have been overestimated in my study. Moreover, there is lack of activity data on manure treatment technologies at animal category, regional and country levels, which contributes also to uncertainties in the model outcome.
- There are uncertainties in model parameters, especially emission factors, which contribute to the overall uncertainty of the results (Chapter 5). Country specific parameters have been used in the MITERRA model when such data were available. However, monitoring data on emissions from animal manures are still insufficient (Chapter 4), especially regarding gaseous emissions from treated manures, and leaching of N and P from stored manures. Evidently, there is a need to improve the accuracy of regionally specific emissions factors for manure treatment activities.

7.4 A harmonized and transparent methodology for estimating feed use and nutrient excretion at national level.

Accurate estimating nutrient excretion by livestock is the first step towards improving nutrient cycling and mitigating gaseous emissions in the manure management chain. Variations in N excretion factors affect the calculated emissions of NH_3 and N_2O directly. The basis for the calculation of N excretion factors should be the input-output balance method, i.e. (N excretion) = (feed N intake) –(N retention in animal products, such as milk, eggs and live-weight gains). The input-output balance method is firmly grounded on the commonly accepted law of mass conservation. The input-output balance can be applied at animal level, farm level, as well as at national level. National studies that include all animal categories allow to check the feed balance (Eshel *et al.*, 2014a); do the total supplies of feed resources in a country indeed match well with the sum of the estimated national feed demand by all animal categories?

The national feed balance provides also a check of the accuracy of N excretion coefficients, and is a basis for the analysis of emission mitigation potentials of animal diet-related strategies. In addition, linking feed use to specific animal categories allows also to allocate feed production related resource use (e.g., land, water, fertilizers) and greenhouse gas emissions to individual animal categories and animal food products (Tilman & Clark, 2014). Ranking various animal products according to the associated environmental (e.g., N footprints) and human health impacts can facilitate the implementation of revised dietary recommendations (Eshel *et al.*, 2014b; Galloway *et al.*, 2014).

In Chapter 3, a transparent and uniform methodology for estimating N excretion rates of the main animal categories was developed and applied for all Member States of the EU, through linking statistical data and information on the availability (quantity and quality) of feed with animal numbers and the energy and protein requirements of the animals. The allocation of feed resources to animal categories was based on a set of constraints and an optimization routine. Results show that annual N excretion factors per animal category varied greatly between countries (80-140 kg N for dairy cows, 40-70 kg N for other cattle, 6-12 kg N for pigs), which is mainly caused by differences in feed use and animal productivity.

The methodology developed in Chapter 3 has also some limitations, and the methodology therefore needs to be improved and validated further. Firstly, the cross-border trade of live animals are not included in my calculations. Although the number of traded live animals for each country is recorded yearly in Eurostat, the live-weights of these animals are unknown. Since the live-weight is a crucial factor determining the energy requirements for animal maintenance and growth, such data are required to improve the accuracy of this methodology. Secondly, the constraints employed in the feed optimization routine were considered to be uniform for the majority of countries. These assumptions to some extent mask variations in the estimated feed use between countries. While this feed model is designed to enable the use of country-specific constraints, there is often insufficient information on these constraints, such as the ranges in the feed allocation to animal diets. The results from this model can be further validated by using regional and national statistical data on animal diets, such as those in the Netherlands (CBS, 2012). Furthermore, animal categories defined in my model are similar to those of the Farm Structure Survey, but the category ‘other cattle’ needs to be further refined to functional cattle categories (calves, beef cattle, suckling cows), which requires that the numbers, live-weight and feed requirements have to become available for these categories. Finally, national specific data on yields, mean energy and protein-N contents in grass and forages can be used to further improve the accuracy of estimating national total N excretion, as indicated in Chapter 3 (e.g. the sensitivity analyses). The suggested aforementioned improvements require additional information and data, which are not always easy to obtain, and this limits the applicability of the suggested improvements.

7.5 Impacts of mitigation measures and treatment technologies on emissions

The meta-analysis (Chapter 4) and the whole-chain analyses (Chapter 5) provide quantitative insights into the co-benefits and trade-offs of the main emission mitigation measures and treatment technologies.

Animal manures are responsible for about 80% of the total NH_3 emissions in EU-27 (EEA, 2014; Leip *et al.*, 2015). To comply with the UN-ECE Gothenburg Protocol, the EU National Emission Ceilings Directive (NECD, 2001/81/EC), and the EU Industrial Emissions Directive (IED, 2010/75/EU; the former IPPC Directive, 2008/1/EC), a variety of NH_3 mitigation measures for reducing emissions from animal manures have been developed and implemented in EU, but with large differences between countries. Estimating the effectiveness of mitigation measures can therefore have implications for the EU environmental policies. Scenario analyses in Chapter 4 indicate that implementation of single measures can decrease total NH_3 emissions from a slurry-based farming system by 9-29% (Table 7-1). At the EU-27 level, it was estimated that maximum implementation of single NH_3 mitigation measures may lead to 1-18% of reduction, compared to a situation without measures (Velthof *et al.*, 2009). The effects at the EU level also depend on the nature of livestock production systems; the mitigation measures are mostly applicable for manure produced from housed animals, and much less for manure from grazing animals. Application of manure with low emission approaches appears to have the largest abatement potential, followed by low-N feeding and covering manure storages (Table 7-1). The NH_3 abatement potential is strongly increased if measures are combined. The abatement potential in the EU-27 was relatively low compared to that in the developing countries, such as China, where the agri-environmental regulations are as yet not well-established and mitigation measures are not implemented yet (Ma *et al.*, 2013). A decrease of up to 45% of total NH_3 emissions from pig production systems in China may be achieved with adoption of low-emission animal houses and manure storages (Bai *et al.*, 2014).

Table 7-1. Summary of effects of the implementation of NH_3 emission mitigation measures on emissions at farm and EU-27 levels, compared to a situation without measures.

NH_3 emission mitigation measures	Changes in emissions, in %					
	At farm level [a]			At EU level [b]		
	NH_3	N_2O	CH_4	NH_3	N_2O	NO_3 leaching
Low-N feeding (Diet)	-18	-1.4	0	-4	-2	-2
Covering manure storages (Cover)	-9	2131	-48	-1	0	0
Low-emission manure application				-18	12	3
Band application	-23	0	0			
Slurry injection	-29	91	0			
Combinations of measures						
Straw (Cover) + slurry injection	-45	124	-3			
Plastic (Cover) + slurry injection	-49	119	-40			
Diet+ straw + injection	-66	95	-3			
Diet+ Plastic +injection	-69	90	-40			
Low-emission application + Balanced N application				-24	-15	-29

[a] refers to scenario analyses for the model liquid-based pig farm in Chapter 4
[b] refers to scenario analyses with maximum country-specific implementation of measures in EU-27 (Velthof *et al.*, 2009); effects on CH_4 emissions were not analyzed.

Implementation of single measures (except low-N feeding) may increase N_2O emissions (Chapter 4) and NO_3 leaching from soil (Velthof *et al.*, 2009). The possible increases in NO_3 leaching from soil take place when no supplemental measures are taken to correct for the increased soil N surplus. This follows in part from the "hole in the pipe" theory (Figure 7-1), i.e. blocking one of the holes in the pipe usually leads to increased leakages from other holes (i.e. pollution swapping), unless the N input is decreased and/or N output in useful products is increased proportionally (Oenema *et al.*, 2009). Therefore, it is recommended that NH_3 emission mitigation measures are combined with balanced N fertilization (Schröder *et al.*, 2007; Oenema *et al.*, 2009; Velthof *et al.*, 2009). Low-N feeding is an effective strategy to decrease N (e.g. NH_3, N_2O and N_2) emissions from the whole manure management chain, without unwanted side-effects. When the protein content of the animal feed decreases by a moderate range (up to 10%, relative to reference) in EU-27, NH_3 emissions from manures decrease by up to 6%. The scope of abatement by low-N feeding largely depends on the current protein content in animal feed and on the economics of low-N feeding (see Section 7.7).

Table 7-2. Effects of increased implementation of manure treatment technologies on NH_3 and GHG emissions and on nitrogen (N) recovery from manure in the EU-27, compared to a situation without treatment in 2010. It is assumed that 20% of the total manure production from housings is treated by each of the technologies. Technologies have been defined in Chapter 5.

Technologies	Change in emissions and N recovery, in %			
	NH_3	N_2O	CH_4	N recovery[a]
Screw press	-1	-1	-7	1
Decanter centrifuge	-1	-1	-11	1
Anaerobic digestion (AD)	0	-2	-17	1
Acidification (Acid)	-10	0	-18	7
Nitrification-denitrification	-8	24	-18	-9
Composting	-2	-2	-6	2
Thermal drying	-9	-5	-7	2
Incineration	-11	-4	-7	-16
Pyrolysis	-11	-5	-7	-9
Combination of techniques				
Acid-centrifuge	-10	0	-19	7
Centrifuge-Acid, pyrolysis	-7	-1	-19	0
AD-Acid-centrifuge	-7	0	-17	5

[a] Nitrogen recovery is defined here as the fraction of total N excreted in housing that is retained in manure and available for application to land..

The effects of manure treatment technologies on emissions from the manure management chain in EU-27 were assessed through scenario analyses in Chapter 5. The implementation level (20% of total manure production in housings) was assumed to be uniform for all scenarios; the level corresponds to the level currently achieved by forerunner countries for a specific technology. All treatment technologies have certain advantages and disadvantages. Slurry acidification appears to be the most promising technology for mitigating both NH_3 and GHG emissions, followed by thermal drying, incineration and pyrolysis (Table 7-2). Acidification is also relatively cheap (Foged *et al.*, 2011a; see Section 7.7). However, there are concerns related to the use of strong acids (e.g. H_2SO_4, HCl) (Fangueiro *et al.*, 2015). Acidifying slurry with H_2SO_4 may increase hydrogen sulfide (H_2S) emissions, which is highly toxic (Dai & Blanes-Vidal, 2013). Minimizing H_2S emissions requires the aeration (oxygenation) of the acidified slurry, which is energy-intensive (Jensen, 2002). Composting and nitrification-denitrification treatment technologies may enhance N_2O emissions significantly, depending on environmental conditions. Anaerobic digestion is effective for GHG mitigation, but does require gas-tight covering of storage facilities to minimize NH_3 emissions and the recovery of residual biogas from stored digestate (Gioelli *et al.*, 2011;

Battini *et al.*, 2014). Acidifying the digested slurry before storage can also decrease NH_3 and GHG emissions significantly (Regueiro *et al.*, 2016). Hence, combination of techniques may minimize unwanted side-effects. Biogas production can also reduce CO_2 emissions due to the replacement of heat and electricity produced from fossil fuels, which was not examined in my thesis. Based on an additional analysis, the avoided CO_2 emissions from coals may amount to 7.6-12.1 Tg CO_2-eq in the scenario with anaerobic digestion of 20% of total manure production. This is equivalent to 9-14% of non-CO_2 GHG emissions from the manure management chain in EU-27 in 2010 (see discussion in Chapter 5).

7.6 Nutrient recovery

The estimated total feed N and P intake by all animal categories in EU-27 in 2010 was 12.2 Tg N and 1.92 Tg P, respectively (Chapters 3 and 5). The fractions of N and P in animal feed retained in animal products (liveweight gains, eggs, milk) were on average 21 and 24%, respectively, and the fractions excreted via urine and faeces were on average 79% and 76%, respectively (Table 7-3). The N recovery, the amounts of N in manures that were applied to and deposited on pastures, expressed as percent of the total N excretion, varied from 53 to 78% between countries (Table 7-3). The remaining N fraction was lost during manure storage and application/deposition to agricultural land or was exported to other countries. Differences between countries in N recovery are related to differences in animal categories and productivity, manure management systems and implementation of mitigation measures.

Grazing systems have relatively high N recovery and low N losses when compared with landless, industrial animal systems. Urine excreted by grazing animals typically infiltrates into the soil before substantial NH_3 emissions can occur and overall NH_3 emissions per animal are therefore less for grazing animals than for housed animals. This explains why the total N recovery from animal manure is relatively high in Ireland and UK (78%, Table 7-3). Grazing animals account for 91% in Ireland and 78% in UK of the total number of animal (expressed in livestock units), while the average for the EU-27 is 57% (Eurostat, 2016). Increasing grazing time of cattle has been recommended as measure to mitigate NH_3 emissions and the recovery of N (Bittman *et al.*, 2014). The potential to increase grazing depends on soil type, topography, farm size and structure (distances), climatic conditions, etc. (Bittman *et al.*, 2014). Though the N recovery from the excreta from grazing animals is high, the effectiveness of N in dung and urine from grazing animals to fertilize pastures is low (Webb *et al.*, 2013). This is related to the spatially uneven distribution of the dung and urine patches and the poor synchrony of the deposition relative to herbage N demand in temperate climates. Hence, a significant fraction of the N from dung and urine is lost via NO_3 leaching and denitrification (e.g. Ryden *et al.*, 1984; De Klein & Van Logtestijn, 1994; Oenema *et al.*, 2008).

Table 7-3. The recovery of N and P at different stages of the manure management chain, from feed intake to manure applied to or deposited on land in EU-27 at country level in 2010 (derived from Chapters 3 and 5).

Country	N and P recoveries at different stages of the manure management chain (%)					
	Feed to excretion		Excretion to manure applied to land		Feed to manure applied to land	
	N	P	N	P	N	P
Austria	78	74	62	98	48	73
Belgium*	73	70	71	96	52	68
Bulgaria	81	82	64	98	52	80
Cyprus	76	76	56	97	43	74
Czech	77	76	60	98	46	74
Denmark	73	73	70	99	51	72
Estonia	79	75	66	98	52	74
Finland	74	73	67	99	50	73
France*	83	79	72	99	60	79
Germany*	73	70	66	102	48	72
Greece	82	82	78	99	64	81
Hungary	73	75	56	97	41	73
Ireland	88	82	78	99	68	82
Italy	80	76	60	98	48	74
Latvia	83	80	65	98	54	78
Lithuania	81	79	66	98	53	77
Luxembourg	84	80	73	99	62	80
Malta	76	82	53	97	40	80
Netherlands*	76	73	65	91	49	66
Poland	75	74	56	97	42	72
Portugal	81	77	73	98	59	76
Romania	84	83	70	98	59	82
Slovakia	81	80	59	97	48	78
Slovenia	79	76	61	98	48	75
Spain	79	77	71	98	56	76
Sweden	82	77	69	99	57	76
UK	84	80	78	99	65	79
EU-27	79	76	69	98	54	74

*The across-border transport of manure was considered for these countries. For importing countries, the amount of manure applied to land includes the imported manure, for exporting countries, the amount of manure applied to land excludes the exported manure.

Ammonia emission mitigation measures increase the N recovery fraction from the animal excreta collected in housing systems. The implementation of these measures in practice varies between countries. In 2010, nearly all animal farms in Belgium, Denmark and the Netherlands have covered manure storage facilities; therefore the potential to further increase the N recovery through covered storages is limited in these countries. The average recovery of N and P from animal manure in the EU-27 is relative high compared to those of for example China and United States. The recovery fraction was only 25% for N and 45% for P in China in 2010 (Ma *et al.*, 2010; Hou *et al.*, 2013); mitigation measures are not applied and massive amounts of manures are discharged into evaporation ponds and rivers (Wang *et al.*, 2010; Bai *et al.*, 2014). The recovery from manure was 30-40% for N and 70-95% for P on commercial dairy farms in United States (Spears *et al.*, 2003a, 2003b).

The estimated P recovery from manure is relatively high in EU-27 (Table 7-3). This is related to the obligatory leak-tight storage of manures (Nitrates Directive) and the estimated low P losses. Also manure discharges to surface waters are forbidden and grazing animals are not allowed to drop excreta in streams in EU-27. The P losses from manure storages via leaching and surface run-off range from 0 to 10% for EU countries, depending on storage conditions (Velthof *et al.*, 2009). The uncertainty in these P loss fractions is relatively high because very few experimental studies have been conducted to measure P losses from manure storages in practice (Radcliffe *et al.*, 2009).

7.7 Barriers to the implementation of mitigation measures and manure treatment

To increase the understanding related to the factors that affect the adoption of treatment technologies in practice, a stakeholder survey was conducted in Denmark, Italy, the Netherlands and Spain (Chapter 6). These four countries together shared about 55% of the total GHG mitigation achieved by manure treatment in the EU in 2010 (Chapter 5). Results from the survey indicate that the implementation of treatment technologies in practice is hindered by financial barriers such as high capital and operational cost.

Technologies such as solid-liquid mechanical separation, slurry acidification and composting of solid manure are generally considered to be less complex in operation and relatively low cost. Thus, these technologies are suitable for farm-scale operations (Chapter 6). Costs of treating slurry with a mechanical screen separator or with a decanting centrifuge were estimated at €0.15 to €0.8 per kg N in effluent, including the power costs, maintenance costs and the capital costs of the equipment (Møller *et al.*, 2000). Slurry acidification has a net cost of €0.14 per kg N in effluent (or €0.21, without subsides), according to a case study in Denmark (Foged *et al.*, 2011). In comparison, nitrification-denitrification treatment and anaerobic digestion have relatively high economic costs, and are most suitable for farm cooperatives or at industrial scale. Case studies indicate that the cost of nitrification-denitrification treatment was €1 to €1.2 per kg N in effluent for a plant in Spain, and about €2 to €2.5 per kg N in effluent for a plant in the Netherlands (Melse & Verdoes, 2005; Foged *et al.*, 2011). The net cost of anaerobic mono-digestion was estimated at €2.3 to €3 kg per N in the effluent for a biogas plant in Denmark. Co-digestion of manure with other substrates can increase the yield of biogas, and thus may lead to a decrease in cost (Foged *et al.*, 2011).

Several assessments have indicated that subsidies are necessary for the successful adoption of anaerobic digestion, when the price of fossil energy remains as low as during the period 2010-2016 (Chinese *et al.*, 2014; CDM, 2015; Chapter 6). Installation of an incineration plant requires a vast investment and also requires subsidies in order to make it economically feasible. The poultry incineration plant in the Netherlands relies on the MEP (Environmental Quality of Electricity Production) subsidy of more than €20 million per year. In addition, there are economic returns from sales of electricity, ash (PK-fertilizer), and from a gate fee for the poultry manure. This gate fee ranges from €4 to €21 per ton of poultry manure, depending on the quality of the manure, the distance to the plant and the duration of the contract.

Table 7-4. Ammonia emission abatement measures and estimated costs for farmers in Europe and North America (summarized from the UNECE guidance document; Bittman *et al.*, 2014).

Abatement measures	Management stages	Cost (€/kg NH_3-N reduced) [b]
Low-protein feeding[a]	whole chain	-2.0-2.0
Covered storages		
-tight lid, tent structure	slurry storage	1.0-2.5
-plastic cover	slurry storage	0.5-1.3
-floating cover	slurry storage	0.3-5.0
-straw cover	slurry storage	-
Low-emission application	application to land	
-trailing hose		-0.5-1.5
-trailing shoe	arable land/grassland	-0.5-1.5
-injection, open slot	grassland	-0.5-1.5
-injection, closed slot	arable land/grassland	-0.5-1.2
-Incorporation of surface applied slurry	arable land	
(immediately by ploughing)		-0.5-1.0
(within 4 hours)		-0.5-1.0
(within 24 hours)		-0.5-2.0
-Incorporation of surface applied solid manure	arable land	
(immediately by ploughing)		-0.5-1.0
(within 4 hours)		0.5-1.5
(within 12-24 hours)		0.5-2.0

[a] a decrease of 1% in protein content in feed (absolute value).
[b] negative costs indicate net gains, mainly through feed savings (low-protein feeding measures) or fertilizer N savings (low-emission application measures).

Implementation of NH_3 emission abatement measures and manure treatment increases the cost of livestock farming, and may have significant effects on farmers' income (Oenema *et al.*, 2009). Because of the perceived high costs, farmers may be reluctant to implement these measures, which in part contributes to the variable and slow responses of Member States to environmental policies in agriculture (Oenema *et al.*, 2011). Economic costs and cost-effectiveness of abatement measures are the key factor affecting farmers' decisions. Costs of

NH_3 emission abatement measures at different stages of the manure management chain are summarized in Table 7-4.

The cost of diet manipulations depend on the initial animal feed composition and on the market prices of feed ingredients and synthetic amino acids (Bittman *et al.*, 2014; Reis *et al.*, 2015). Low-N animal feeding is mostly applicable to housed animals and less to grassland-based systems with grazing animals. In EU-27, about 65% of N excretion is currently produced in animal houses, and 35% is from grazing animals in pastures (Chapter 5). The costs of low-emission manure application technologies are related to investment, depreciation and maintenance costs (spreader, use of heavier tractor), labor costs (increased labor time), and the size of the farm area. The additional costs of low-emission application are partially outweighed by the financial benefits of higher yields and yield consistency, reduced mineral fertilizer requirements, and by a reduction of odor and crop contamination (Webb *et al.*, 2006; Bittman *et al.*, 2014).

7.8 Some concluding remarks

Scientific research on manure and manure treatment started more than a century ago (e.g. Kolenbrander and De la Lande Cremer, 1967; Russell, 1978). For a long time, the research was focused on its fertilizing and soil quality improvement values. Bussink and Oenema (1998) reviewed that possible NH_3 losses from manures have been known since early 19^{th} century, while the full environmental impacts of N losses from animal manure have been examined seriously only from the 1980s. Emission mitigation measures have been implemented in practice from the 1990s, while the idea of a manure management chain approach was developed from the 2000s (e.g. Velthof et al., 2009).

My PhD thesis research has been built on a long history of research. The most novel aspects of my PhD thesis research are

- The consistent application of a manure management chain approach, which allows a coherent analysis and estimation of the effects (including side-effects) introduced through interventions in this chain,
- A systematic comparison of methodologies used for the estimation of manure N production per animal category in EU-countries,
- The development of a harmonized and transparent methodology for the estimation of manure N and P production at country and EU levels,

- A systematic analysis of the effects of the main NH_3 emission mitigation measures on NH_3 and GHG emissions,
- A systematic analysis of the effects of the main manure treatment technologies on nutrient recovery and gaseous emissions,
- A survey on stakeholder perceptions related to manure treatment technologies,
- An integrated assessment of emission mitigation measures and manure treatment techniques at national level and EU level.

The findings reported in this thesis are important for scientists, policy makers, industry, extension services, as well as farmers and farmers' organizations. The amounts of N and P in animal manures produced annually are large in the EU-27 (similar to the amounts of fertilizer N and P used annually), while emissions of NH_3, N_2O and CH_4, and N and P losses from the whole manure management chain have significant environmental impacts. Various emission mitigation measures and manure treatment techniques exist, but the effectiveness and costs of these measures and techniques vary greatly. The implementation of measures and techniques will not happen overnight; they require incentives, guidance and cooperation.

7.9 Recommendations for future research

My research has identified also a number of topics that would benefit from further studies.

- Country and regional specific NH_3 and GHG emission factors for manure treatment techniques, considering also possible temporal (seasonal) variations in emissions.
- Quantification of the N and P fertilizer replacement values of manure treatment products for different crops.
- Quantification of P losses from manure storages and manure treatment techniques across EU-27.
- Further investigating barriers to the implementation of emission mitigation measures and manure treatment technologies in different countries, including those that have (e.g. Germany, Belgium, Netherlands, and UK) and do not have much experiences with these measures and technologies.
- Further development of treatment technologies that minimize unwanted side-effects and lower the cost (through increases in energy efficiency, and maximizing the mitigation potential). In the ReUseWaste research project, improved approaches for

manure treatment have been developed for manure separation, energy recovery, emission mitigation during manure storage and field application, which may increase the fertilizer value of manure products (see http://www.reusewaste.eu/). Further validation of these improved technologies and products in pilot and commercial scales are needed. The results of these additional tests will contribute also to improving the MITERRA-Europe model used for integrated assessments at national and EU levels.

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SUMMARY

Summary

Animal manures are major sources of nutrients and organic matter, to be used to fertilize crops and improve soil quality. However, when not properly managed, these manures release considerable amounts of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) into the air, and nitrogen (N) and phosphorus (P) to water bodies, which create a range of unwanted environment impacts. Emissions of NH_3 contribute to the acidification and eutrophication of nitrogen-limited ecosystems, and can have negative effects on human health. Emissions of N_2O and CH_4 contribute considerably to the radiative forcing; the global warming potentials of N_2O and CH_4 are 298 and 25 times higher, respectively, than that of CO_2 per kg. Approximately 40% of the global anthropogenic NH_3 and N_2O emissions and about 10% of the global anthropogenic CH_4 emissions are associated with animal manures. N and P losses from manure decrease also its fertilization value.

Emissions of NH_3, CH_4 and N_2O to air and leaching losses of N and P to water bodies from manure depend on the management activities and techniques used at different stages of the whole manure management chain, from animal feeding up to manure application to land. A large number of emission mitigation measures and manure treatment technologies have been developed during the last decades, and part of these have been implemented in practice in Europe. Introducing a measure or technology to mitigate emissions from one source may affect emissions downstream in the chain (so-called ‘pollution swapping’), or emissions of other pollutants. The trade-offs and co-benefits of emission mitigation measures and manure treatment technologies are as yet poorly understood, especially when taking the whole manure management chain into account. Moreover, the effects of combinations of measures and technologies have not been well analyzed, and analyses at national scales are lacking.

The overall objective of this PhD thesis research is (i) to enhance the quantitative insight into the effects of emission mitigation measures and treatment technologies on emissions of NH_3, N_2O and CH_4, and the recovery of N and P from animal manure in the whole chain from animal feeding to manure application to land, and (ii) to explore the effects of combinations of measures and technologies to mitigate these emissions and to increase the N and P recovery. The research was part of the Marie Curie Training Program ReUseWaste (Recovery and Use of Nutrients, Energy and Organic Matter from Animal Waste). In total eleven PhD students and two post docs have been working on various specific mitigation measures and treatment technologies. The research reported in this thesis focused on the integral analysis of

the whole manure management chain, and on the up-scaling and synthesis of research results, including those from literature. This thesis has 5 research chapters, next to a general introduction (Chapter 1) and a general discussion (Chapter 7)

In Chapter 2, methodologies for estimating N excretion factors for the main animal categories in member states of the European Union (EU) were reviewed. The review included the guidelines and methodologies suggested by the EU Nitrates Directive, the OECD/Eurostat gross N balance guidebook, the EMEP/EEA Guidebook and the IPCC Guidelines. In addition, approaches used in modeling studies were reviewed. Nitrogen excretion factors (or coefficients) are defined as the total amount of N excreted by a well-defined livestock category per year via urine and faeces. Results show that N excretion factors for dairy cattle, other cattle, pigs, laying hens, broilers, sheep, and goats differ significantly between countries and also within countries between policy reports. Part of these differences may be related to differences in animal breeds and animal productivity, size/weight of the animals, and feed composition. Another part of the differences in N excretion factors is related to differences in methodologies and the aggregation of livestock categories. The methodologies and data used by member states are often not well described. It is concluded that there is a need for a common, harmonized methodology and procedure for the estimation of N excretion factors, to arrive at a common basis for the estimation of the production of manure N, and for the estimation of N balances and emissions of NH_3 and N_2O across the EU.

In Chapter 3, a transparent and uniform methodology for estimating annual feed use and N excretion per animal category for all countries of the EU-27 was developed, based on the energy and protein requirements of the animals and statistics of feed use and composition, animal number and productivity. The calculated total feed use in the EU-27 was 506 Tg dry mass in 2010. Dairy cows used 29%, other cattle 34%, pigs 17%, chicken 9%, sheep and goats 8%, and other animal categories 3% of the total feed use. Grass and annual forages were mainly used by dairy cows (30 and 49%, respectively) and other cattle (55 and 44%); pigs used most of the feed cereals (53%); protein-rich feed (e.g., soybean meal) were mostly used by pigs (34%) and chicken (24%). Differences between countries in feed use were large, which are mainly related to variations in national feed supply and animal productivity. The calculated total N excretion of the animals amounted to 9.7 Tg in 2010, and varied between countries from 14 in Bulgaria to 291 in Netherlands kg N ha^{-1} of utilized agricultural land. The method developed in this Chapter addresses various key livestock categories simultaneously, using a uniform methodology and common national statistics, and provides national averages, which allow direct comparison of feed use and N excretion coefficients among countries.

In Chapter 4, firstly the impacts of a suite of NH_3 mitigation measures on emissions of NH_3, N_2O and CH_4 at individual stages of the manure management chain were analyzed by means of a meta-analysis of published data (derived from 126 published studies). Secondly, the overall impacts of alternative combinations of mitigation measures on NH_3, CH_4, and N_2O emissions from the whole chain were evaluated through scenario analysis. Significant NH_3 emission reduction was estimated for i) housing via lowering the dietary crude protein (CP) content (emission reduction of 24-65%, depending on the reduction rate of CP), for ii) external slurry storages via acidification (83%) and covers of straw (78%) or artificial films (98%), for iii) solid manure storages via compaction and covering (61%, compared to composting), and for iv) manure application to land via band spreading (55%, compared to surface application), via incorporation into the soil (70%) and via injection into the soil (80%). Acidification decreased CH_4 emissions from stored slurry by 87%. Significant increases in N_2O emissions were found for straw-covered slurry storages (by two orders of magnitude) and manure injection (by 26-199%). Compaction, static stockpiling and covering of solid manure tend to decrease NH_3 emissions and increase CH_4 and N_2O emissions compared to manure heaps subjected to composting. However, the number of observations underlying these latter effects is low. Lowering the protein content of feed and acidifying slurry are strategies that consistently reduce NH_3 and GHG emissions in the whole chain. Other strategies may reduce emissions of a specific gas or emissions source, by which there is a risk of unwanted trade-offs in the manure management chain. Proper combinations of mitigation measures at farm level are therefore important to minimize emissions of NH_3 and GHG from animal manure.

Chapter 5 reports on an integrated assessment of the effects of manure treatment on NH_3, N_2O and CH_4 emissions from manure management chains in EU-27 at the national level for the year 2010, using the improved model MITERRA-Europe. Whole-chain effects of implementing twelve treatment technologies in EU-27 on emissions and nutrient (N and P) recovery were further explored through scenario analyses; the level of implementation corresponded to levels currently achieved by forerunner countries (i.e. 20% of total manure production from housings for each technique). The scenario analyses included an uncertainty analysis. Results show that total NH_3 emissions from the manure management chain in EU-27 were 2.5 Tg N and GHG emissions were 86.9 Tg CO_2-eq in 2010, with a relative uncertainty (coefficient of variation) of 16% and 20%, respectively. Manure treatment decreased GHG emissions from 0 to 17% depending on country in 2010, with the largest reduction from anaerobic digestion (especially in Germany, Denmark and Italy). Manure treatment effects on NH_3 emissions were small in 2010. Scenario analyses indicate that acidification, thermal drying, incineration and pyrolysis can consistently decrease both NH_3 (9-11%) and GHG (11-18%) emissions from the management chain of EU-27 (compared to the reference without manure treatment). Nitrification-denitrification treatment decreased NH_3 emissions, but increased GHG emissions, due to increased N_2O emissions. Composting had no significant effects on total emissions of NH_3 and GHG from the chain. Anaerobic digestion decreased GHG emissions (19%), but only marginally affected overall NH_3 emissions. Combining anaerobic digestion with acidification (i.e. acidifying digested slurry) resulted in significant reductions in both NH_3 and GHG emissions. The N recovery (% of nitrogen excreted in animal housings that is applied to land) in EU-27 would increase from a mean of 57% (in 2010) to 61% in the acidification scenario, but would decrease to 48% in the incineration scenario and to 52% in the nitrification and denitrification treatment scenario. Other technologies (solid-liquid separation, anaerobic digestion, drying, composting etc.) only marginally affect the N recovery. The P recovery was estimated at 98% in 2010, and was not significantly affected by the manure treatment scenarios.

Chapter 6 reports on a survey conducted under various stakeholder groups with expertise in the domain of manure treatment in four European countries (Denmark, Italy, the Netherlands and Spain) that have regions of high animal density. The survey addressed questions related to i) which factors facilitate and hinder the implementation of treatment technologies in practice, ii) which technologies have the most potential for successful adoption, and iii) how farm characteristics and the scale of the treatment operation affect priorities for adoption. Results show that pressure from governmental regulations was perceived as a key factor that stimulated manure treatment in all four countries (70% of respondents). Processing manure to produce bioenergy was considered important in Denmark and Italy, but less important in Spain and the Netherlands. The major barriers to technology adoption were related to economic factors (lack of investment capital, high processing cost and a long payback time; 45-60% of respondents), while there was relatively little concern regarding transport, noise burden and health risk. Slurry separation and anaerobic digestion were perceived to have the

greatest potential for a common adoption in practice in all four countries. Other preferred technologies were more country-specific (e.g. acidification in Denmark, composting in Spain, and drying and reverse osmosis in Netherlands), which is related to the differences between regions/nations in livestock densities and socio-economic, political and environmental conditions. Farm characteristics and scale of operation were identified as important factors that can influence the decision of farmers and investors for technology adoption.

The main conclusions of this PhD thesis are as follows:

- In EU-27, the amounts of N and P in manure are as large as or larger than the total amounts of fertilizer N and P used annually. However, there is a huge spatial variation in manure production. Nutrient excretion factors per animal category also vary between countries, as a result of variations in feed use and animal productivity. Clearly, for accurate inventories of national emission there is a need for estimating nutrient excretion using country-specific feed use data. There is a need for a common methodology and protocol for the estimation of N and P excretion factors per animal category, because some differences between countries in reported excretion factors were related to differences in methodology and aggregation/upscaling.
- Increasing the effectiveness of measures to mitigate NH_3 and GHG emissions from animal manure requires proper combination of measures in the manure management chain. Lowering the dietary protein content in animal feed is an effective measure to reduce NH_3 emissions and other N emissions at all stages of the manure management chain. Other measures may reduce emissions of a specific gas or emissions source, by which there is a risk of unwanted trade-offs in the manure management chain. Joint adoption of these measures with low-N feeding strategies and slurry acidification can greatly decrease the risk of pollution swapping.
- Implementation of manure treatment is on average still limited in EU-27. Effects of manure treatment on NH_3 and GHG emissions are therefore relatively small at EU level. Increasing the implementation of treatment technologies, including acidification, incineration and thermal drying, or optimized combinations of treatment technologies, can significantly contribute to achieving NH_3 and GHG emission targets of EU environmental policies.
- Implementation of manure treatment technologies provides opportunities to improve the use of plant nutrients in manures, because of the release of manure products with

different N/P ratios. Applying acidification technology and optimized combination of NH_3 emission mitigation measures increase the N recovery from animal manure, and can decrease the demand of mineral fertilizers. However, some technologies decrease the N and P recovery and/or decrease the availability of the N and P in manure products to plants.

- Implementation of manure treatment in practice is forced by the pressure from EU environmental regulations, and is hindered by financial barriers. To encourage the adoption of manure treatment, policies must be economically appealing to attract new adopters (farmers and industries). Long-term financial support schemes (e.g. subsidies) seem to be necessary, especially with the current low prices for fossil fuels. Outreach strategies are required to convey the knowledge to stakeholders from both the supply and the demand side, with respect to the economic, technical and environmental aspects of manure treatment technologies.

SAMENVATTING

Dierlijk mest is een belangrijk bron van nutriënten en organisch materiaal. De mest dient om gewassen te bemesten en om de bodemkwaliteit te verbeteren. Echter, als mest niet goed wordt opgeslagen en/of behandeld, dan kunnen nutriënten en broeikasgassen naar het milieu lekken. Het gaat daarbij om de gasvormige verbindingen ammoniak (NH_3), lachgas (N_2O) en methaan (CH_4) die naar de lucht kunnen ontsnappen, en om nutriënten die uit mestopslagen en na toediening op het land naar grondwater en oppervlaktewater kunnen uitspoelen. Deze emissies hebben een reeks van milieukundige effecten.

Emissies van NH_3 leiden tot verzuring en eutrofiëring van stikstof-gelimiteerde ecosystemen. In de lucht heeft NH_3 een negatief effect op de menselijke gezondheid. Lachgas (N_2O) en methaan (CH_4) zijn broeikasgassen. Het broeikasgaseffect van N_2O en CH_4 is respectievelijk 298 en 25 keer sterker dan dat van CO_2 per kg. Op wereldschaal is circa 40% van de totale antropogene NH_3- en N_2O-emissies en bijna 10% van de antropogene CH_4-emissies afkomstig van dierlijk mest. Het verlies van stikstof (N) en fosfor (P) uit mest leidt bovendien tot een vermindering van de bemestende waarde van mest.

De risico's van emissies van NH_3, CH_4 en N_2O uit mest naar de lucht en van de uitspoeling van N en P uit mest naar grondwater en oppervlaktewater hangt af van de opslag en behandeling van mest in de gehele mestketen. De mestketen begint bij het voeren van de dieren en eindigt bij de toediening van de mest op het land.

Er zijn verschillende maatregelen en technieken beschikbaar om mest te bewerken en emissies uit mest te reduceren. De introductie van een maatregel in het begin van de mestketen kan effect hebben op de emissies in een later deel van de mestketen, en het kan de emissie van andere vervuilende stoffen veroorzaken. De effecten van en interacties tussen maatregelen op emissies uit mest in de mestketen zijn slechts in beperkte mate bekend. Bovendien zijn de gevolgen van combinaties van maatregelen en technieken op emissies nauwelijks onderzocht.

Het onderzoek, dat in dit proefschrift is beschreven, had tot doel (i) het inzicht te verbeteren in de effecten van emissiebeperkende maatregelen en mestbewerkingstechnieken op de emissies van NH_3, N_2O en CH_4 naar de lucht en op de resterende hoeveelheden N en P in de mest in de mestketen, en (ii) om de effecten te verkennen van combinaties van maatregelen en technieken om de emissies verder te verminderen en de resterende hoeveelheden N en P in de mest te verhogen.

Het onderzoek maakte deel uit van het Marie Curie Trainings Programma “ReUseWaste” (Winning en gebruik van nutriënten, energie en organisch materiaal uit dierlijke mest). In totaal waren elf promovendi en twee postdocs betrokken bij dit project; er zijn diverse specifieke emissiebeperkende maatregelen en mestbewerkingstechnieken onderzocht en verbeterd. Het onderzoek beschreven in dit proefschrift richtte zich op de integrale analyse van mogelijke maatregelen op de emissies in de gehele mestketen en op de opschaling en de synthese van de onderzoeksresultaten uit ReUseWaste en de literatuur. Het proefschrift heeft 5 hoofdstukken met specifieke resultaten van het onderzoek, een algemene inleiding (hoofdstuk 1) en een algemene discussie (hoofdstuk 7).

In hoofdstuk 2 is een studie beschreven waarin verschillende methodes voor de berekening van de N-excretie door vee werden beoordeeld. De stikstofexcretiefactor is gedefinieerd als de totale hoeveelheid N die gemiddeld per diersoort per jaar via de urine en feces wordt uitgescheiden. In de studie zijn de methoden en richtlijnen, die door de Nitraatrichtlijn, OECD/Eurostat, EMEP/EEA, IPCC en lidstaten worden voorgeschreven, geanalyseerd en met elkaar vergeleken. De resultaten tonen aan dat N-excretiefactoren voor melkvee, varkens, leghennen, vleeskuikens, schapen en geiten aanzienlijk verschilden tussen lidstaten, en ook binnen landen tussen verschillende rapportages ten behoeve van nitraat-, ammoniak- en klimaatbeleid. Een deel van deze verschillen worden veroorzaakt door verschillende in veerassen (grootte/gewicht van de dieren), productiviteit, en in voersamenstelling. Een ander deel van de verschillen in N-excretiefactoren wordt veroorzaakt door verschillen in rekenmethoden en in de methode van aggregatie van diercategorieën. De methoden en gegevens die worden gebruikt door de lidstaten in de EU zijn vaak niet goed beschreven. Er is behoefte aan geharmoniseerde en goed gedocumenteerde rekenmethoden en procedures voor de vaststelling van N-excretiefactoren. Uniforme methoden en procedures vormen de basis voor een nauwkeurige schatting van de productie van N in mest, van N-balansen in de landbouw en de uitstoot van NH_3, N_2O en CH_4 uit mest in de EU.

Hoofdstuk 3 beschrijft een studie waarin een robuuste en uniform toepasbare methode is ontwikkeld om de N-excretie per diercategorie per jaar voor alle landen van de EU-27 te berekenen. Deze methode is gebaseerd op de energie- en eiwitbehoefte van de dieren en op statistische data en informatie over voersamenstelling, dieraantallen en productiviteit. In de EU-27 werd in 2010 in totaal 506 Tg voer gebruikt (1 Tg = 10^{12} = 1 miljoen ton). Melkkoeien gebruikten 29%, jongvee en mestvee 34%, varkens 17%, kippen 9%, schapen en geiten 8%, en andere diercategorieën 3% van het voerverbruik in de EU. Gras en andere ruwvoeders werden voornamelijk gebruikt door melkkoeien (30 en 49%, respectievelijk) en ander rundvee (55 en 44%). Varkens gebruikten vooral voergranen (53%). Eiwitrijke diervoeders (bijvoorbeeld sojameel) werden vooral door varkens (34%) en kippen (24%) gegeten. Er waren grote verschillen tussen de landen in diervoeding, voornamelijk veroorzaakt door de variaties tussen landen in voerbeschikbaarheid en in de productiviteit van de dieren. De berekende N-excretie door alle landbouwdieren bedroeg 9,7 Tg in EU-27 in 2010. De N-excretie varieerde van 14 kg N ha^{-1} landbouwgrond in Bulgarije tot 291 kg N ha^{-1} in Nederland. De ontwikkelde methode maakt het mogelijk om N-excretiefactoren van landen te valideren en om verschillen tussen landen te duiden.

In hoofdstuk 4 wordt een studie beschreven waarin de effecten zijn geanalyseerd van NH_3-emissie beperkende maatregelen op de emissies van NH_3, N_2O en CH_4 uit mest. De analyse werd uitgevoerd via een meta-analyse van gepubliceerde gegevens (afkomstig van 126 gepubliceerde studies). Ook zijn de effecten van alternatieve combinaties van emissiebeperkende maatregelen op NH_3-, CH_4,- en N_2O-emissies in de mestketen geëvalueerd via scenario-analyses. De volgende maatregelen leidden tot een significante reductie van NH_3-emissies i) het verlagen van het ruw-eiwitgehalte in het voer (emissiereductie 24-65%, afhankelijk van de reductie in ruw-eiwitgehalte), ii) het aanzuren van dunne mest in de mestopslag (83%), het afdekken van dunne mest in de opslag met stro (78%) of met een zeil (98%), iii) het verdichten en afdekken van vaste mest in de opslag (61%), en iv) het emissiearm toedienen van mest op het land via een sleepvoet-machine (55%), via het inwerken van de mest in de grond (70%) of via injectie van de mest in de grond (80%). Door het aanzuren van de mest daalt ook de CH_4-emissie uit mest in opslag (met 87%). Het bedekken van dunne mest met stro leidt tot een forse stijging van de N_2O-emissie (met een factor honderd). Ook mestinjectie leidt tot hogere N_2O-emissies (26-199%). Bij opslag van vaste (stapelbare) mest leidt het verdichten van de mest (via het berijden met een zwaar voertuig) en het afdekken met een tentzeil gemiddeld tot een verlaging van NH_3-emissies, maar nemen de CH_4- en N_2O-emissies toe. Het aantal waarnemingen is echter gering dat ten grondslag ligt aan de laatst genoemde bevindingen. Het verlagen van het eiwitgehalte van rantsoenen en het aanzuren van dunne mest zijn strategieën die alle emissies (NH_3 , N_2O en CH_4) in de mestketen verminderen. Verschillende andere strategieën beperken de emissies van een specifiek gas of emissiebron, maar hebben ongewenste neveneffecten elders in de mestketen. Door te kiezen voor een juiste combinaties van maatregelen op bedrijfsniveau kan de uitstoot van NH_3 en broeikasgassen uit dierlijke mest tot een minimum worden beperkt.

Hoofdstuk 5 beschrijft een integrale analyse van maatregelen om de NH_3-, N_2O- en CH_4-emissies uit de mestketens te verminderen op nationaal niveau in EU-27 voor 2010. Hierbij is gebruik gemaakt van een aangepaste versie van het model MITERRA-EUROPE. De effecten van 12 mestbehandelingstechnieken op de emissies van NH_3, N_2O en CH_4 en de resterende hoeveelheden N en P in de mest werden verkend door middel van scenario-analyses. De effecten van maatregelen werden berekend bij een uniforme implementatiegraad van 20% in alle landen. Deze implementatiegraad werd in 2015 voor enkele maatregelen gerealiseerd in landen die voorop lopen met mestbehandeling. Er is ook een onzekerheidsanalyse uitgevoerd. De totale NH_3-emissie uit de mestketen in EU-27 in 2010 was 2.5 Tg N en de totale broeikasgasemissie was 86.9 Tg CO_2-equivalenten, met een relatieve onzekerheid (variatiecoëfficiënt) van respectievelijk 16% en 20%. De behandeling van mest reduceerde de broeikasgasemissies van 0 tot 17% in 2010, afhankelijk van het land. De grootste reductie werd verkregen door mestvergisting (met name in Duitsland, Denemarken en Italië). De effecten van mestbehandeling op NH_3-emissies waren klein in 2010. De scenario analyses laten zien dat aanzuren, thermisch drogen, verbranden en pyrolyse tot een reductie van zowel NH_3-emissies (9-11%) als broeikasgasemissies (11-18%) leiden in de EU-27 (in vergelijking tot een referentie zonder mestbehandeling). Mestbehandeling via nitrificatie-denitrificatie reduceerde de NH_3-emissies, maar verhoogde de broeikasgasemissies (omdat N_2O emissie toeneemt bij deze techniek). Composteren had geen significant effect op de totale emissies van NH_3 en broeikasgassen. Mestvergisting reduceerde de broeikasgasemissie met 19%, maar had amper een effect op de totale NH_3-emissie. De combinatie van mestvergisting en het aanzuren van mest resulteerde in een significante vermindering in zowel NH_3– als broeikasgasemissies. De hoeveelheid N in de mest die uiteindelijk wordt toegediend op het land (in % van de N-excretie in de stal) in EU-27 neemt toe van gemiddeld 57% in 2010 tot 61% bij het scenario ‘aanzuren’, maar neemt af tot 48% in het scenario ‘mestverbranding’ en tot 52% bij behandeling via nitrificatie-denitrificatie. Andere technieken, zoals het scheiden van mest in een dikke en dunne fractie, mestvergisting, drogen en composteren, hadden slechts een beperkt effect op de resterende hoeveelheid N in de mest. De hoeveelheid P die uiteindelijk via de mest op het land wordt toegediend (in % van de P-excretie in de stal) is geschat op 98% in 2010; deze hoeveelheid werd niet significant beïnvloed door mestbehandeling.

Hoofdstuk 6 beschrijft de resultaten van een enquête onder verschillende groepen belanghebbenden en actoren op het gebied van mestbehandeling in vier landen met een hoge veedichtheid (Denemarken, Italië, Nederland en Spanje). De enquête was gericht op vragen betreffende i) succes- en faalfactoren bij de toepassing van mestbehandeling in de praktijk, ii) technieken die de meeste perspectieven bieden voor toepassing in de praktijk en iii) bedrijfskarakteristieken en de schaalgrootte van mestbehandeling die de toepassing van mestbehandeling beïnvloeden. Regelgeving van overheden wordt gezien als de belangrijkste factor die mestbehandeling in de praktijk stimuleert in de vier landen (70% van de respondenten). De grootste obstakels voor de toepassing van technieken zijn gerelateerd aan economische factoren; gebrek aan investeringskapitaal, hoge kosten voor mestbehandeling en een lange terugverdientijd (45-60% van de respondenten). Er waren relatief weinig zorgen over transport, geluidsoverlast en gezondheidsrisico's bij mestbehandeling. Mestscheiding en mestvergisting bieden het meeste perspectief voor toepassing in alle vier landen. De perspectieven van andere technieken waren meer land-specifiek, zoals het aanzuren in Denemarken, composteren in Spanje en het drogen en omgekeerde osmose in Nederland. Deze verschillen worden veroorzaakt door verschillen tussen landen en regio's in veedichtheid en in sociale, economische, politieke en milieukundige factoren. Bedrijfskarakteristieken en de schaal van toepassing van mestbehandeling werden genoemd als de belangrijke factoren bij het kiezen van een bepaalde techniek.

De belangrijkste conclusies van dit proefschrift zijn:

- De hoeveelheden N en P in mest in EU-27 zijn vergelijkbaar met of groter dan de hoeveelheden N en P die via kunstmest worden gebruikt. Er is echter een grote ruimtelijke variatie in mestproductie binnen EU-27. Ook de omvang van de excretie van N en P per landbouwdier varieert sterk tussen landen. Dit wordt vooral veroorzaakt door verschillen in rantsoensamenstelling en productiviteit van de dieren. Voor een nauwkeurige bepaling van de emissies van stikstof en broeikasgassen op nationaal niveau is het noodzakelijk om de excreties van N en P nauwkeurig te berekenen op basis van landen-specifieke data, een algemeen toepasbare rekenmethode en een transparent protocol. Een deel van de huidige verschillen tussen landen in gerapporteerde excretiefactoren en emissies wordt veroorzaakt door verschillen in rekenmethoden en aggregatiemethoden.
- Combinaties van effective maatregelen in de mestketen zijn nodig en mogelijk om de bemestende waarde van mest te verhogen en om de emissies van NH_3 en broeikasgassen uit dierlijke mest te verlagen. Het verlagen van het eiwitgehalte in rantsoenen tot aanbevolen niveaus is een effectieve maatregel om de emissies van NH_3

en andere N-verbindingen in alle delen van de mestketen te verminderen. Andere maatregelen leiden vaak tot vermindering van de emissie van een specifiek gas of van een specifieke emissiebron, maar hebben een risico op afwenteling naar emissies van andere gassen of veroorzaken een toename van de emissie elders in de mestketen. Een combinatie van het verlagen van het eiwitgehalte in het voer en het aanzuren van mest kunnen het risco op afwenteling van emissies sterk beperken.

- De implementatie van mestbehandeling in EU-landen is divers en in het algemeen nog beperkt. De effecten van mestbehandeling op NH_3- en broeikasgasemissies zijn daardoor relatief klein op EU-schaal. Een grotere implementatie van de mestbehandelingstechnieken, inclusief aanzuren, verbranden, thermisch drogen, en vooral een geoptimaliseerde combinatie van technieken kan significant bijdragen tot het realiseren van de beleidsdoelstellingen met betrekking tot de reductie van NH_3- en broeikasemissies in de EU.
- De implementatie van mestbehandelingstechnieken geeft mogelijkheden om de benutting van N en P uit mest voor bemesting te verbeteren, omdat er producten vrijkomen met een andere N/P-verhouding. Het toepassen van een geoptimaliseerde combinatie van technieken om NH_3-emissie te reduceren, leidt er toe dat meer N in de mest resteert, waardoor minder kunstmest N nodig is. Sommige technieken leiden echter tot minder N en P in mest en/of verminderen de beschikbaarheid van N en P in de mest voor gewassen.
- De implementation van mestbehandelingstechnieken in de EU wordt indirect gestimuleerd door EU-milieurichtlijnen en wordt beperkt door economische barrières. Technieken moeten economisch attractief zijn om toegepast te kunnen worden. Mestvergisting biedt inkomsten uit de productie van bio-energie, maar is zonder subsidies momenteel nauwelijks attractief vanwege de huidige lage prijzen voor fossiele brandstoffen. Er zijn slimme strategieën nodig om de kennis over mestbehandeling met betrekking tot de economische, technische en milieukundige aspecten te verbeteren en te verspreiden onder de belanghebbenden.

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Shijiazhuang

Publication list

(in this thesis)

Hou, Y., Velthof, G.L., Oenema, O., 2015. Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: a meta-analysis and integrated assessment. *Global Change Biology*. 21, 1293–1312.

Hou, Y., Bai, Z., Lesschen, J.P., Staritsky, I.G., Sikirica, N., Ma, L., Velthof, G.L., Oenema, O., 2016. Feed use and nitrogen excretion of livestock in EU-27. *Agriculture, Ecosystems and Environment.* 218, 232–244.

Hou, Y., Velthof, G.L., Case, S.D.C., Oelofse, M., Grignani, C., Balsari, P., Zavattaro, L., Gioelli, F., Bernal, M.P., Fangueiro, D., Trindade, H., Jensen, L.S., Oenema, O., 2016. Stakeholder perceptions of manure treatment technologies in Denmark, Italy, the Netherlands and Spain. *Journal of Cleaner Production.* (*In press*)

Hou, Y., Velthof, G.L., Lesschen, J.P., Staritsky, I.G., Oenema, O., Nutrient recovery and emissions of ammonia, nitrous oxide and methane from animal manure in Europe: effects of manure treatment technologies. *Environmental science & technology*. (*Accepted subject to revision*)

Velthof, G.L., **Hou, Y.**, Oenema, O., 2015. Nitrogen excretion factors of livestock in the European Union: a review. *Journal of the Science of Food and Agriculture*. 95, 3004–3014.

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Hou, Y., Gao, Z.L., Heimann, L., Roelcke, M., Ma, W.Q, Nieder, R., 2012. Nitrogen balances of smallholder farms in major cropping systems in a peri-urban area of Beijing, China. *Nutrient Cycling in Agroecosystems.* 92, 347–361.

Hou, Y., Ma, L., Gao, Z.L., Wang, F.H., Sims, J.T., Ma, W.Q., Zhang, F.S., 2013. The Driving Forces for Nitrogen and Phosphorus Flows in the Food Chain of China, 1980 to 2010. *Journal of Environmental Quality*. 42, 962-971.

Hou, Y., Ma, L., Sárdi, K., Sisák, I., Ma, W.Q., 2015. Nitrogen flows in the food production chain of Hungary over the period 1961–2010. *Nutrient Cycling in Agroecosystems.* 102, 335–346.

Heimann, L., Roelcke, M., **Hou, Y.**, Ostermann, A., Ma, W.Q., Nieder, R., 2015. Nutrients and pollutants in agricultural soils in the peri-urban region of Beijing: Status and recommendations. *Agriculture, Ecosystems and Environment.* 209, 74–88.

Curriculum vitae

Yong Hou was born on 09 October, 1985 in Lincheng County, Hebei Province, P.R. China. In 2003, he graduated from the Lincheng senior high school. After that, he studied in the major of Agricultural Resource and Environmental Science in the Department of Resource and Environmental Science, Hebei Agricultural University, China. After he completed the Bachelor degree in 2008, he continued his research as a master student in the field of nutrient and resource management, in the Plant Nutrition Group, Hebei Agricultural University. He successfully completed the Master of Science degree in Agronomy in 2011, with the master dissertation entitled ‘Characteristics and regulatory approaches of nutrient flows in intensive crop-livestock production systems of the Peri-urban area of China’, financially supported by the Sino-German project “Recycling of Organic Residues from Agricultural and Municipal Residues in China”. During the master period, he was supervised by Prof. Wenqi Ma in Hebei Agricultural University, and jointly by Dr. Marco Roelcke and Prof. Rolf Nieder in Technical University of Braunschweig, Germany. During Sep. 2011- Sep. 2012, he was employed as an research assistant in Hebei Agricultural University.

In September 2012, he became a PhD candidate in Soil Quality Group, Wageningen University, the Netherlands, and was awarded as a Marie Curie fellowship as part of the EU-funded project “ReUseWaste: Recovery and Use of Nutrients, Energy and Organic Matter from Animal Waste”. He was supervised by Prof. Oene Oenema and Senior scientist Dr. Gerard L. Velthof in Wageningen University and Research. His PhD dissertation is entitled “Towards Improving the Manure Management Chain”. His PhD research was funded by the People Programme (Marie Curie Actions) of the European Union’s Seventh Framework Programme FP7/2007-2013/under REA grant agreement no 289887.

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)

Review of literature (6 ECTS)

- Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: a meta-analysis and integrated assessment

Writing of project proposal (4.5 ECTS)

- Integrated assessment of manure management chains in EU-27

Post-graduate courses (3.8 ECTS)

- Introduction to R for statistical analysis; PE&RC (2013)
- Material flow analysis with STAN 2.5; TU Vienna (2013)
- Governmental policies dealing with nutrient management; Wageningen University (2013)
- Mata-analysis; PE&RC (2014)

Laboratory training and working visits (4.5 ECTS)

- Data analysis and joint scientific paper writing; University of Copenhagen (2014)

Invited review of (unpublished) journal manuscript (2 ECTS)

- Agriculture, Ecosystems and Environment: band application of treated cattle slurry as an alternative to slurry injection: the need for an integrated evaluation (2015)
- Agriculture, Ecosystems and Environment: field evaluation combined with modelling analysis to study fertilizer and tillage as factors affecting N_2O emissions: a case study in the Po valley, Northern Italy (2016)

Competence strengthening / skills courses (6.9 ECTS)

- PhD Introduction course; University of Copenhagen (2012)
- Project management workshop; University of Turin (2013)
- Scientific writing-A, fundamentals; Instituto Superior de Agronomia, University of Lisbon (2015)
- How to write grant proposals; University of Copenhagen (2015)

PE&RC Annual meetings, seminars and the PE&RC weekend (1.2 ECTS)

- PE&RC Weekend (2012)
- PE&RC Day (2013)

Discussion groups / local seminars / other scientific meetings (5.2 ECTS)

- Sustainable intensification of agricultural systems, Wageningen (2014-2015)
- Joint scientific workshop: innovative strategies to improve the recycling of energy, nutrients and organic matter from waste materials; Erfurt, Germany (2015)
- R-Users discussion group; Wageningen (2015-2016)

International symposia, workshops and conferences (4.7 ECTS)

- 15th International conference: RAMIRAN; Versailles, France (2013)
- ManuResource conference; Bruges, Belgium (2013)
- 18th Nitrogen workshop; Lisbon, Portugal (2014)