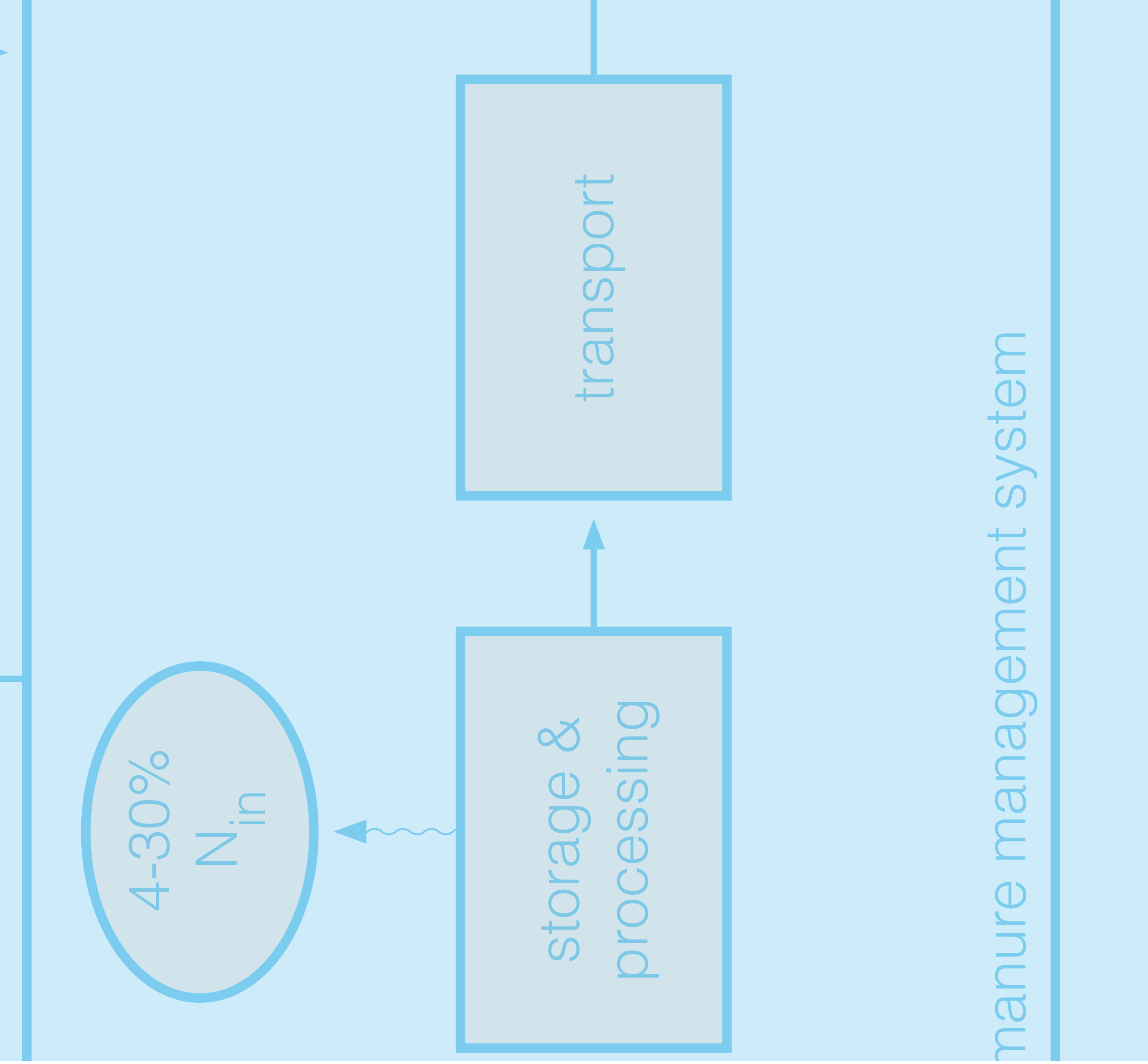

FROM ANIMALS TO CROPS

// ENVIRONMENTAL CONSEQUENCES OF CURRENT AND FUTURE STRATEGIES FOR MANURE MANAGEMENT

Jerke W. de Vries

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Therefore do not seek to understand in order to believe,
but believe that thou mayest understand.

Saint Augustine of Hippo

Any fool can know; the point is to understand.

Albert Einstein

/ ABSTRACT

De Vries, J.W. (2014). From animals to crops - Environmental consequences of current and future strategies for manure management. PhD thesis, Wageningen University, the Netherlands.

Animal manure is a key component that links crop and livestock production as it contains valuable nutrients for the soil and crop. Manure is also a source of environmental pollution through losses of nutrients, such as nitrogen (N) and phosphorus (P), and losses of carbon (C). These losses are largely determined by the way manure is managed. Technologies to reduce nutrient and C losses from manure mainly focused on reducing a single emission while unwillingly increasing another emission at the same time; a phenomenon called pollution swapping. To prevent pollution swapping, we need to gain insight into the integral environmental consequences of technologies and use these insights to (re)design the manure management chain. The aim of this thesis, therefore, was to provide knowledge and insight into the environmental consequences of current and future strategies for manure management. The environmental consequences of the following technologies were assessed: mono- and co-digestion of liquid manure; high-tech separation of liquid manure with further dewatering of the liquid fraction; and segregating fattening pig urine and feces inside the housing system. Following, we designed new strategies for integrated manure management that prevent pollution swapping, and assessed the environmental consequences of these strategies. Life cycle assessment was used to calculate the environmental impacts of current and future strategies. For the design, we adapted and used a structured approach to engineering design to create new strategies for integrated manure management. It was concluded that mono-digestion of liquid manure reduced the environmental impact compared to conventional manure management, but has a low potential to produce bio-energy. Co-digestion with waste and residues, such as roadside grass, increased bio-energy production and further reduced the environmental impact. Co-digestion with substrates that compete with animal feed increased bio-energy production, but also the overall environmental impact from producing a substitute for the used co-substrate. Separating liquid manure into liquid and solid fractions with further de-watering of the liquid fraction increased the environmental impact compared to manure management without processing. A combination of separation and anaerobic mono-digestion of the solid faction reduced climate change and fossil fuel depletion. Segregating fattening pig urine and feces in the housing system reduced climate change, terrestrial acidification, and particulate matter formation and provided a sound basis for environmentally friendly manure management. Applying a structured design approach enabled the design of new strategies for integrated manure management that prevented pollution swapping. The approach proved to be successful because the environmental impact reduced throughout the manure management chain by at least 57% and more than doubled the nitrogen use efficiency compared to current North Western European manure management practices.

Table of contents

Chapter 1

// GENERAL INTRODUCTION

J.W. De Vries

1.1 BACKGROUND

Animal manure is the key component that links crop and livestock production. It contains valuable nutrients for the soil and crop, such as nitrogen (N), phosphorus (P), and potassium (K), and carbon (C). Efficient cycling of nutrients and C among soils, crops and animals is essential to sustain soil quality and crop growth, and hence to produce food, feed, fiber, and biofuels. In the past 50 to 60 years, these cycles have been disrupted, mainly because of: development of mixed crop-livestock farms into specialized farms that produce feed or food crops or keep livestock that produce animal-source food; geographical relocation of farms into centralized production regions; and intensification of farming (Steinfeld et al., 2006; Tilman et al., 2002; Wilkins, 2008). Forecasts show that global manure production is expected to increase as a result of increased consumption, and thus production of animal-source food products (Steinfeld et al., 2006). Overall, above described trends lead to areas with high livestock densities and a surplus of manure, such as the Netherlands, and other areas with little manure for crop production, inducing the need to use nutrients from mineral fertilizer (Wilkins, 2008).

In areas with high livestock densities, such as the European Union (EU), field application of manure causes losses of nutrients and C into air, water, and soil, which cause environmental pollution. Major pathways of nutrient losses include: leaching and run-off of nitrate (NO_3^-) and phosphate (PO_4^{3-}) to ground and surface waters, resulting in eutrophication and human health problems; emission of greenhouse gases (GHGs), such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), resulting in climate change; and emissions of ammonia (NH_3) resulting in acidification, eutrophication, and particulate matter formation (Steinfeld et al., 2006). Gaseous N losses also contribute to the formation of particulate matter, which results in human health problems. Additionally, intensive livestock production causes odor nuisance and fossil fuel depletion. During the processing and transport of surplus manure to other regions, energy is used. N losses indirectly also lead to energy use, as mineral N fertilizer is required for crops requiring energy to produce (Berglund & Börjesson, 2006).

Previous environmental pollution related to livestock production lead to international and national regulations, e.g. Kyoto Protocol, Gothenburg Protocol, National Emission Ceilings (NEC), and Nitrates Directive. The Kyoto protocol for the EU-15, for example, aimed at an 8% reduction in GHG emissions in 2008 - 2012 compared to 1990; this was 15% in 2011 and is aimed at 20% reduction for 2020 (EC, 2013). The NEC for NH_3 emission was achieved by most countries in 2010 (i.e. 128 kilotons for the Netherlands), but will be lowered further (EC, 2012; EU, 2001). Further sharpening of the directives is necessary, especially in regions that, for example, are sensitive to NH_3 deposition, such as Natura2000 zones. This means that such regulations will stimulate the development of new strategies for manure management to comply with the regulations and ultimately the environment (Burton & Turner, 2003).

1.2 CATTLE AND PIG MANURE MANAGEMENT IN NORTH WEST EUROPE AND THE NETHERLANDS

Main types of manure produced in the EU are: liquid pig (~149 million tons), and liquid and solid cattle manure (~448 and 295 million tons, respectively) (Henning Lyngsø et al., 2011). In 2011, the Netherlands produced about 11 million tons of liquid pig manure, 46 million tons of liquid cattle manure, and only 0.1 million tons of solid cattle manure (CBS, 2011).

In this thesis, the management of this manure includes: the collection and storage in the animal house, outside storage, processing, transport and field application (Fig. 1.1). Manure management also can affect other so-called external processes, such as electricity and fuel production, production of mineral fertilizer, and production of (by-)products needed for manure processing (flocculating additives, acids), or anaerobic digestion (co-substrates).

Common systems for liquid manure management in the EU are: in-house storage; outside storage in an open or covered tank; and field application by means of broadcast spreading and sometimes injection, whereas common systems for solid management are: stockpiling of excreted manure, inside or outside the animal house; and field application by means of spreading, sometimes in combination with incorporation into the soil. Liquid and solid manure are typically stored about 4 - 6 months during winter, as application of manure is prohibited during this time (Burton & Turner, 2003).

During manure management, losses of N, P, K, and C occur. In this thesis we focus on losses of N, P, and C, because of their potential for environmental pollution. Losses of N and C occur from three main emission processes, including conversion or production and volatilization of compounds: conversion of urea in urine to NH_3 and subsequent volatilization of NH_3;

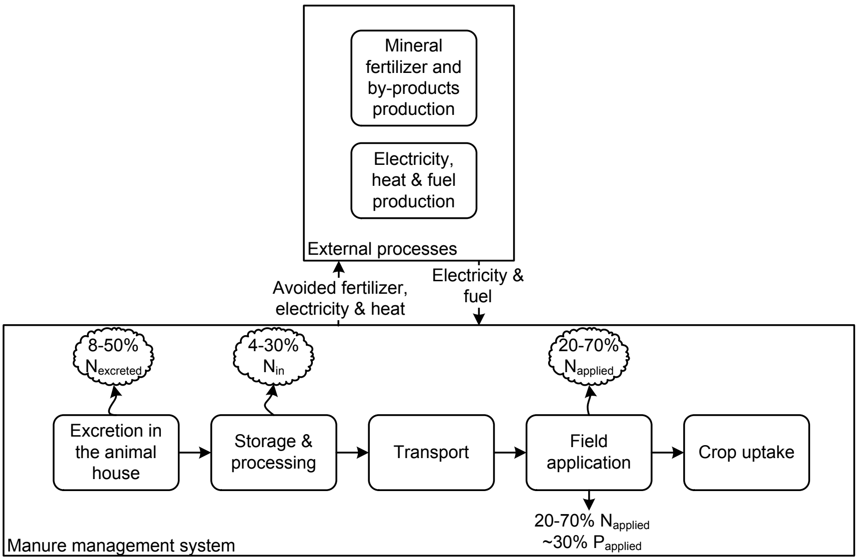

Fig. 1.1 Manure management system, range of N and P losses during management (gaseous losses upwards, leaching losses downwards), and external processes that are affected by changes in manure management.

nitrification of ammonium (NH_4^+) to NO_3^-, and subsequent denitrification of NO_3^- to nitrogen gas (N_2), with N_2O and NO as intermediate products; and methanogenisis of organic matter to CH_4 and CO_2 (Chadwick et al., 2011). Losses of C as CO_2 also result from burning of fossil fuels and production of electricity needed for management. Losses of N in the separate stages of the manure management system are about: 8 - 50% of the excreted N from in-house storage, 4 - 30% of the N entering storage and processing, and 20 - 70% of the applied N from field application, leading to an overall N use efficiency of roughly 50% in the EU (Fig. 1.1) (Oenema et al., 2007; Oenema & Tamminga, 2005; Rotz, 2004). Increasing the NUE was indicated as critical for improving the environmental performance of food production (Cassman et al., 2002; Spiertz, 2010). Losses of P occur mainly from run-off or leaching of PO_4^{3-} into ground and surface water. The soil P surplus or P use efficiency (PUE) is often used as an indicator for risk of P loss. The average PUE in the EU is about ~70% in 2000; meaning that ~30% of the applied P is not taken up by the crop, but accumulates in the soil, where it is susceptible to leaching over time (Fig. 1.1) (Bouwman et al., 2009).

The previous losses contribute to different environmental impacts as considered in this thesis (Table 1.1). Specifically, in this thesis we considered the changes in environmental impact, called environmental consequences, from adjusting the manure management system and external processes.

Emissions of NH_3, N_2O and CH_4 from manure management in the EU caused about 17% of the agricultural GHG emissions and 72% of NH_3 emission in 2010 (EEA, 2012), whereas in the Netherlands this was about 23% and 77%, respectively (CBS et al., 2012a; RIVM, 2012). GHG emission in the whole system expressed per ton of manure have been reported to be roughly 105 - 250 kg CO_2-equivalents (eq.) per ton of pig manure, and 300 kg CO_2-eq per ton of dairy cattle manure (Hamelin et al., 2011; Lopez-Ridaura et al.,

Table 1.1 Environmental impact categories, contributing compounds and resources and their respective units as considered in this thesis

Environmental impact categories and indicators	Main contributing compounds or resources	Unit
Climate change (CC)	CH_4, N_2O, and CO_2	kg CO_2-eq[a]
Terrestrial acidification (TA)	NH_3, NO_x, and SO_2	kg SO_2-eq
Marine Eutrophication (ME)	NH_3, NO_x, and NO_3^-	kg N-eq
Freshwater eutrophication (FE)	PO_4^{3-}	kg P-eq
Particulate matter formation (PMF)	NH_3, NO_x, and SO_2 as precursors	kg PM_{10}-eq
P_2O_5 over application/ soil surplus[c] (POA)	P_2O_5 application	kg P_2O_5
Fossil fuel depletion (FFD)	Non-renewable sources, such as oil, natural gas, and uranium.	kg oil-eq[b] or MJ
Land use	Crop production	m^2
N use efficiency (NUE)	N losses	%

[a]eq = equivalent, [b]1 kg oil-eq = 42 MJ (Goedkoop et al., 2009), [c]In Chapter 6 we used the P over application relative to crop demand as an indicator for the P soil surplus and eventual P loss.

2009; Prapaspongsa et al., 2010). Losses of P contribute to freshwater eutrophication and occur mainly from 'over application' of manure to agricultural soils. In the Netherlands, for example, 37% of the total applied P is accumulated in soils. Of the applied P, about 77% is added by animal manure (CBS et al., 2012b).

Summing up from the previous, manure management results in significant environmental pollution and, therefore, requires strategies to reduce the environmental impact of future manure management systems.

/ 1.3 MANURE MANAGEMENT AND NEED FOR (RE-)DESIGN AND INTEGRATED ASSESSMENT

1.3.1 Manure management and pollution swapping

Main factors that determine the environmental impact of manure management include: composition of the manure product, such as N/P_2O_5 ratio, and the suitability for soil and crop requirements; the type of housing, storage and processing method; and timing, method and location of manure product application (Rasmussen et al., 1998; Schröder, 2005). In parallel to manure management, soil management is important for nutrient and C dynamics and thus interacts with manure application (Venterea et al., 2005). Soil management includes the type of tillage, such as inversion or no tillage, and the type of field traffic applied.

So far, technologies to reduce the losses of N, P, and C and the environmental impact of manure management mainly focused on reducing the loss of a single compound, such as NH_3, N_2O, or CH_4, or focused on a single management stage only, such as storage or field application. Examples are: separation of urine and feces directly after excretion in the housing system (referred to as segregation), covering manure storages, anaerobic digestion for bio-energy production, or new field application techniques, such as injection. As a result of using these technologies, the targeted loss was decreased, but other losses were increased or were swapped to another management stage that was not assessed, causing so-called environmental trade-offs. These trade-offs are also referred to as 'pollution swapping'. Pollution swapping is caused by complex interactions between process variables of underlying emission processes and seems difficult or impossible to solve, due to conflicting requirements of the these processes (Jarivs & Menzi, 2004). Compared to conventional pig manure management, for example, segregating pig urine and feces inside the animal house reduced CH_4 up to 80%, NH_3 emission up to 75%, and odor emission up to 74% (Aarnink et al., 2007; Lachance et al., 2005). Assessments, however, excluded changes in emissions during storage and field application of separate urine and feces. Covering manure storages reduced both odor and NH_3 emission up to 95% (Bicudo et al., 2004), but increased N_2O emission by more than 4.5 fold (Berg et al., 2006). Injecting and incorporating liquid and solid manure reduced NH_3 emission up to 90% compared to broadcast spreading (Sommer & Hutchings, 2001), but increased N_2O emission by more than 3 fold (Velthof & Mosquera, 2011). More integrated assessments, using for example life cycle assessment (LCA), also showed that technologies for manure management cause pollution swapping. Lopez-Ridaura et al. (2009) showed that transferring manure to another region or processing the manure with biological treatment and composting did not affect CC. Other impacts, like eutrophication and acidification, however, increased at least by 50% and energy use was more than tripled. This means that available technologies for manure management hardly prevent pollution swapping, because they often do not consider process variables of underlying emission processes. To prevent pollutions swapping, therefore, a structured approach and (re-)design of the manure management system is required that addresses all functions and underlying processes that lead to losses of N, P, and C. The consequences of changes in external processes need to be accounted for also to prevent swapping to outside the manure management system (Fig. 1.1). Finally, such designed management approaches require integrated assessment to reveal their potential to reduce environmental impact.

1.3.2 Need for integrated assessment of current manure processing technologies

Life cycle assessment (LCA) is an internationally recognized method for calculating and comparing the environmental impact of a product or service provided by a production system. LCA helps to assess the environmental impact in an integrated way and can unveil pollution swapping, because various emissions and environmental impact categories are included (Table 1.1). The method comprises four phases: 1. goal and scope definition, 2. life cycle inventory analysis, 3. life cycle impact assessment, and 4. interpretation of results (ISO-14040, 2006).

LCA studies that assessed the environmental performance of manure processing technologies are few (Lopez-Ridaura et al., 2009; Prapaspongsa et al., 2010; Sandars et al., 2003). Currently, technologies for manure management, such as separation of liquid manure and de-watering of the liquid fraction; anaerobic co-digestion with various co-substrates, such as silage maize and by-products from processing industries; and other separation methods are developed and implemented in the EU and the Netherlands. They, however, do require integrated assessment to consider their environmental consequences. Anaerobic co-digestion of manure with energy crops, for example, requires land for production of these crops. This expansion of land may induce land use change (LUC) and is related to increased C emissions that may off-set reductions in C emission from using biofuels (Plevin et al., 2010; Tonini et al., 2012). Assessing the use of other co-substrates, such as glycerin, roadside grass and beet tails from sugar processing, therefore, is essential, because these co-substrates compete with other markets and may require the production of a substitute in these markets. The production impact of substitutes should be accounted for.

Concluding, there remains an overall need to assess and understand the environmental consequences of current and future strategies for manure management. Current developments, such as anaerobic co-digestion require integrated assessment to consider their environmental consequences. Knowledge obtained from these assessments in turn can feed into the (re-)design of future strategies for manure management.

1.4 OBJECTIVES

The overall objective of this thesis was to provide knowledge and insight into the environmental consequences of current and future strategies for manure management. We did this by setting five sub-objectives:

1. assess the environmental consequences of bio-energy production by means of anaerobic mono and co-digestion of pig manure, while accounting for the production of substitutes for used co-substrates,

2. assess the environmental consequences of high-tech separation of liquid pig and dairy cattle manure,

3. assess the environmental consequences of segregating fattening pig urine and feces inside the housing system,

4. design strategies for integrated manure management that prevent pollution swapping and reduce the environmental impact throughout the manure management system,

5. assess the environmental consequences of the designed strategies for integrated manure management.

Objectives 1 to 3 were directed at providing knowledge and insight into environmental consequences of current technologies for manure management. Related studies focused on liquid pig and cattle manure, as these are the main types of manure produced and used in crop production. Objective 4 did build on obtained insights and knowledge and was directed at designing new strategies for integrated manure management to prevent pollution

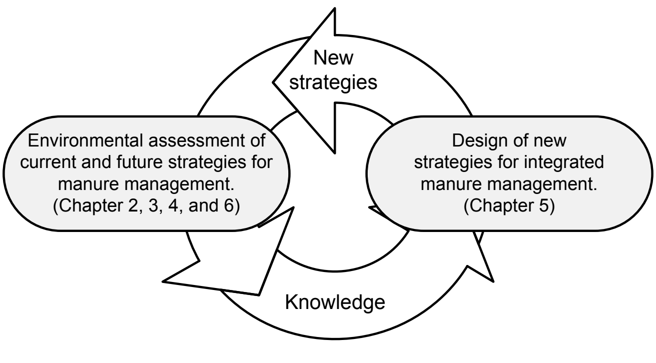

Fig. 1.2 Outline of the chapters in the thesis. Knowledge from Chapter 2, 3, and 4 (corresponding with sub-objectives 1, 2, and 3) is fed into the design of new strategies for integrated manure management in Chapter 5 (corresponding with sub-objective 4). The new strategies were assessed for their environmental impact in Chapter 6 (corresponding with sub-objective 5).

swapping and reduce environmental impact (Fig. 1.2). Objective 5 was directed at assessing the environmental consequences of the designed strategies. Objective 4 and 5 also focused on solid dairy cattle manure as another important type of manure being produced in the EU.

1.5 THESIS OUTLINE

The thesis consists of the following main chapters, next to the general introduction (Chapter 1), that answer the respective objectives (Fig. 1.2):

- In Chapter 2, we compared the environmental consequences of anaerobic mono-digestion and co-digestion of pig manure to produce bio-energy by LCA. This included several important by-products used as co-substrate for anaerobic digestion. The environmental impact of producing a substitute for the used by-products was included together with LUC emissions.

- In Chapter 3, we compared the environmental consequences of high-tech processing of manure to produce a mineral fertilizer replacement, called mineral concentrate, and to produce bio-energy from the solid fraction.

- Chapter 4 consists of an LCA study of segregating fattening pig urine and feces compared to conventional liquid manure management. All stages in the manure management system were included and results were assessed for their uncertainty by Monte Carlo analysis.

- Chapter 5 includes the design of new strategies for integrated pig and dairy cattle manure management to avoid pollution swapping and reduce environmental impact. We applied a structured design approach based on engineering design to fathom the system and address all underlying emission processes and resource use, functions, and their interactions.

- Chapter 6 presents the quantification of the environmental consequences of the newly designed strategies for manure management. Environmental impact was quantified and compared to a reference of North Western Europe and the Dutch situation of manure management.

- Finally, Chapter 7 includes the general discussion and conclusions based on the previous chapters and sums up the essence of the research outcomes.

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Chapter 2

// COMPARING ENVIRONMENTAL CONSEQUENCES OF ANAEROBIC MONO- AND CO-DIGESTION OF PIG MANURE TO PRODUCE BIO-ENERGY - A LIFE CYCLE PERSPECTIVE

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ABSTRACT

The aim of this work was to assess the environmental consequences of anaerobic mono- and co-digestion of pig manure to produce bio-energy, from a life cycle perspective. This included assessing environmental impacts and land use change emissions (LUC) required to replace used co-substrates for anaerobic digestion. Environmental impact categories considered were climate change, terrestrial acidification, marine and freshwater eutrophication, particulate matter formation, land use, and fossil fuel depletion. Six scenarios were evaluated: mono-digestion of manure, co-digestion with: maize silage, maize silage and glycerin, beet tails, wheat yeast concentrate (WYC), and roadside grass. Mono-digestion reduced most impacts, but represented a limited source for bio-energy. Co-digestion with maize silage, beet tails, and WYC (competing with animal feed), and glycerin increased bio-energy production (up to 568%), but at expense of increasing climate change (through LUC), marine eutrophication, and land use. Co-digestion with wastes or residues like roadside grass gave the best environmental performance.

2.1 INTRODUCTION

The demand for renewable energy is rising because of increasing social awareness of consequences related to non-renewable energy use, e.g. fossil fuel depletion, energy security, and climate change (CC). Renewable energy production in the European Union, for example, is targeted to reach 20% of total energy production by 2020 (EU, 2009). This transition requires insight into environmental consequences of producing renewable energy, including CC, fossil fuel depletion, and land use changes. Life cycle assessment (LCA) is an internationally accepted method to gain insight into the environmental consequences of a product or system (ISO-14040, 2006).

Bio-energy is a form of renewable energy and is produced from biomass. Biomass can be converted by anaerobic digestion (AD) into biogas, composed of methane (CH_4), carbon dioxide (CO_2) and some trace gases (e.g., hydrogen gas), which can then be used to produce bio-energy in the form of electricity, heat, or transport fuel (De Vries et al., 2012, Chapter 3; Hamelin et al., 2011). The remaining product after AD, i.e. digestate, can be recycled as organic fertilizer for crop cultivation to substitute mineral fertilizer (Börjesson & Berglund, 2007). Main substrates for AD include agricultural biomass in the form of animal manures and energy crops (e.g. maize), organic residues from the processing industry (e.g. glycerin, beet tails, and gut and intestines from slaughtering houses), and other residues such as, roadside grass or forest residues (Cherubini & Strømman, 2011).

Environmental LCA studies of AD of pig and cattle manure (raw or separated fraction) and energy crops, such as maize and rye grass focused on bio-energy production, greenhouse gas (GHG) emission reduction potentials, and various biogas end applications (Börjesson & Berglund, 2007; De Vries et al., 2012, Chapter 3; Hamelin et al., 2011; Thyø & Wenzel, 2007). These studies highlighted that AD of solely, or fractions of, animal manure (mono-digestion) reduced GHG emissions and fossil fuel depletion due to bio-energy production compared to a reference without digestion. To boost bio-energy production and economic profitability of mono-digestion, co-substrates are added, including energy crops and wastes (co-digestion) (Hamelin et al., 2011; Thyø & Wenzel, 2007). This use of co-substrates competes with other applications, such as animal feed or the production of heat or compost and, therefore, will induce the need of a substitute for their initial use. The environmental impact of producing these substitutes, however, has so far not been considered in LCAs of AD. To further improve the insight into the consequences of such a change, LCAs of bio-energy should include other environmental impacts, such as acidification and eutrophication (Cherubini & Strømman, 2011). Additionally, LCAs of bio-energy production should account for the impact of land use change (LUC) and its related carbon (C) emissions from using various substrates. Generally direct (DLUC) and indirect land use change (ILUC) are distinguished, both included in LUC. While DLUC represents the land use changes in a given country or region associated with the expansion of a specific crop in that area, ILUC refers to global market reactions to feedstock displacement and the resultant land use changes. Accounting for LUC is important as it has the potential to undermine reductions in GHG emissions obtained by bio-energy production (Plevin et al., 2010). However, LUC is most often not addressed in LCAs of AD.

The aim here was to assess and compare environmental consequences of anaerobic mono-digestion and co-digestion of pig manure to produce bio-energy. Environmental impacts of producing a substitute for the initial use of the substrates, including the induced LUC, were accounted for. For co-digestion, five co-substrates were evaluated: ensiled whole crop maize, glycerin, beet tails, wheat yeast concentrate (WYC) and roadside grass. These co-substrates represent various product groups that are, or will be, used in agricultural digesters, i.e. energy crops, by-products from food or feed industry, animal feed products, and residual or waste products.

2.2 MATERIALS AND METHODS

2.2.1 LCA approach and functional unit

The ISO-14040 standard provides the general framework for LCA, which was followed in this study (ISO-14040, 2006). A consequential approach to LCA was used to compare the environmental consequences of mono-digestion with co-digestion using alternative substrates. This implied that all processes affected by the mono- or co-digestion systems studied were included in the model (i.e. system expansion). For the affected processes the marginal suppliers were included (e.g. for electricity, heat, and mineral fertilizers) (Weidema, 2003).

To enable a comparison of scenarios, environmental impacts were related to a functional unit (FU), i.e. the main function of the system expressed in quantitative terms. As the study is focused on the use of various substrates and the substitution of their initial use, an input-related FU of one ton substrate (fresh matter) added to the digester was used. This was either pig manure or a mixture of pig manure and co-substrate(s). Studies addressing different applications of substrates, in this case bio-energy production, are recommended to use input-related FUs (Cherubini & Strømman, 2011).

2.2.2 System boundaries and definition of scenarios

2.2.2.1 System boundaries common to all scenarios

The general scope of this research was North-Western Europe. The context of the Netherlands was used to identify the involved marginal suppliers for electricity, heat, and mineral fertilizer, when establishing the composition of manure and co-substrates, and when legislation had to be taken into account (e.g. for co-digestion).

The system, the included processes, and the system boundary are illustrated in Figure 1. It was considered that digesting manure avoided the conventional management of raw manure without further processing, i.e. outside storage in a concrete-covered tank, transport, and field application. Hence, manure was stored solely inside the animal house prior to digestion where after it was transported to the digestion facility. Processes included in the system boundary, therefore, were: manure storage in the animal house; (avoided) outside raw manure storage and application (avoided); anaerobic digestion; storage and field application of digestate; produced electricity and heat from biogas; avoided production of fossil-based electricity, heat, and mineral fertilizer; production of substitutes for initial use of the co-substrates; transport between several life cycle stages; cultivation of silage maize (as a co-substrate) and the displaced barley, and impacts related to production of capital goods. Pig production was excluded from the system boundary because it was assumed to be unaffected by a change in demand for manure as a substrate for AD (i.e. pig production as main production process with manure as by-product). Similarly, the main production processes of other used by-products (i.e. glycerin, beet tails, and WYC) were excluded, as these were not affected by a change in demand for these products as co-substrates.

A centralized and average scale agricultural digestion plant was considered. The biogas it produced was used in a combined heat and power unit (CHP) for the production of electricity and heat. Produced electricity was assumed to substitute marginal Dutch electricity, i.e. 28% coal-based, 67% natural gas-based, and 5% wind-based electricity (Chapter 3). Produced heat was partly used for the digestion process and partly for substitution of marginal heat, i.e. 79% natural gas-based and 21% light fuel oil-based in the Netherlands (Chapter 3). The digestate was transported and applied to the field as fertilizer, where it substituted marginal mineral N, P, and K fertilizer. Marginal production of mineral fertilizer was assumed to be calcium ammonium nitrate for N, triple superphosphate for P_2O_5, and potassium chloride for K_2O (Chapter 3).

2.2.2.2 Definition of scenarios and substituting products

Mono-digestion of pig manure comprised the digestion of solely manure (1 ton wet weight). Afterwards, digestate was stored and applied to the field as fertilizer. In scenarios with co-digestion, a mixture of 50% manure and 50% co-substrate (on a wet weight basis) was assumed. The Dutch law requires a minimum input of 50% manure in AD in order to allow application of digestate to the field (DR, 2012).

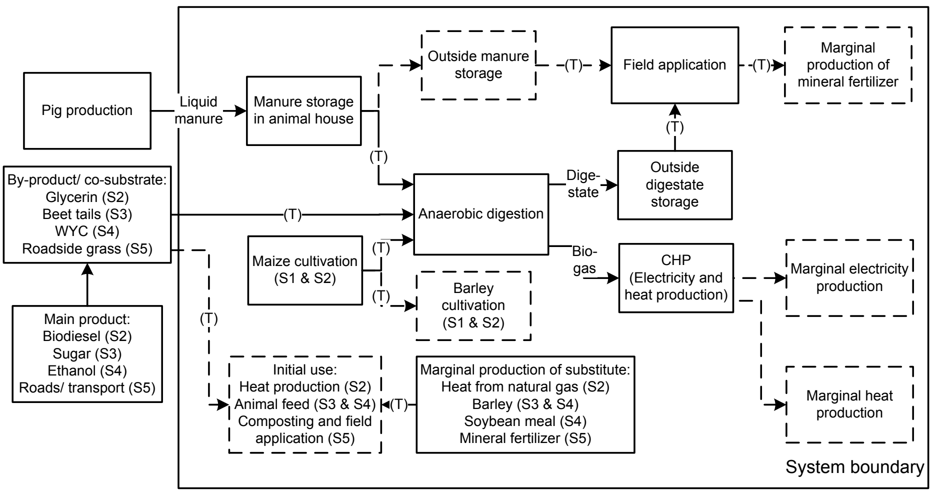

Fig. 2.1. Processes considered within the system boundary. Dashed boxes represent avoided processes. Black arrows indicate induced flows whereas dashed arrows indicate avoided flows. (T) represents transportation. S1 - S5 are the considered scenarios.

Scenario 1 (S1) comprised co-digestion of manure with maize silage. Maize is the most commonly used energy crop for biogas production in Europe. It is attractive due to its high dry matter (DM) yield per ha and high CH_4 production potential (Amon et al., 2007). The maize silage was produced in the Netherlands, specifically for AD (Fig. 2.1), and displaced the production of spring barley (i.e. the marginal crop) (Weidema, 2003). Such displacement induced LUC (i.e. including DLUC as well as ILUC). As the production of maize, instead of spring barley, was assumed to induce only minor changes in emissions (i.e. in fertilization, tillage, etc.), DLUC was excluded from the model. ILUC, however, was included to reflect the production of the displaced spring barley in locations outside the Netherlands, as further detailed in section 2.3.7.

Scenario 2 (S2) comprised co-digestion of manure with crude glycerin (10%) and maize silage (40%). Glycerin is a by-product of biodiesel production and significantly increases CH_4 production when added to AD, as it contains large quantities of labile organic matter. Glycerin, however, is known to inhibit the digestion process when added in high quantities (>10 - 15%), as the concentration of volatile fatty acids becomes inhibiting for methanogenic bacteria. Addition of glycerin to the substrate, therefore, was considered to be 10% with 40% of maize silage. Glycerin was assumed to be initially used for heat production in an industrial gas-fired boiler (i.e. marginal use), where it would have replaced 0.94 GJ of natural gas per GJ of glycerin (Thyø & Wenzel, 2007). Use of glycerin in other markets, like cosmetics or the drug industry, was not considered both because these are currently saturated and given the costs for purification. As in S1, S2 included LUC for the maize silage portion digested.

Scenario 3 (S3) comprised co-digestion of manure with beet tails. Beet tails are cut off at the first cleaning of sugar beets during sugar processing, and represents about 5% of the fresh mass of beets. Beet tails are available all year long for digestion when preserved as silage. Beet tails, now used for AD, were assumed to be initially used in dairy cattle feed for their carbohydrate value, which induces the need for another carbohydrate fodder. The marginal carbohydrate fodder was assumed to be spring barley produced in the Netherlands. For one ton of beet tails, 0.11 ton of barley was required, based on the Dutch energy value for animal feed (SI). The extra production of barley on Dutch agricultural soils induced LUC that was included in the analysis.

Scenario 4 (S4) comprised co-digestion of manure with wheat yeast concentrate (WYC). WYC is a protein-rich by-product from bio-ethanol production. WYC was assumed to be initially used in animal feed for its protein value and, therefore, soy bean meal from Brazil was assumed to replace WYC in feed (Weidema, 2003). For 0.50 ton of WYC, 0.33 ton of soybeans (or 0.27 ton soybean meal) was required, based on the Dutch digestible protein value (SI). The additional soy oil produced (0.05 ton) alongside the soy meal was assumed to substitute palm oil (0.23 ton fruit bunches) (Dalgaard et al., 2008). The no longer produced carbohydrate-rich palm cake, as by-product from palm oil, was assumed to induce a demand for additional barley (3.2 kg) in order to balance carbohydrate supply on the feed market. LUC related to Brazilian soybean, Malaysian palm fruit, and Dutch barley production was considered in the analysis (specified in section 2.3.7).

Scenario 5 (S5) comprised co-digestion of manure with roadside grass, originating from maintenance of side-strips along roads. In the Netherlands, roadside grass is usually composted and subsequently applied to agricultural soils for its fertilizing and soil amending capabilities (Ehlert et al., 2010). Avoided composting was thus considered the opportunity cost of using roadside grass for AD. The compost was assumed to be substituted with mineral N, P and K fertilizer.

2.2.3 Life cycle inventory and assumptions

2.2.3.1 Chemical composition and methane yields of the substrates
Table 2.1 presents the chemical compositions considered for pig manure and co-substrates, before and after digestion, the fraction of organic matter (OM) decomposed during AD and the CH_4 yield per ton of substrate. A mass balance approach was used to compute all compositions and flows within the system. Manure composition after storage in the animal house was based on De Vries et al. (2012) (Chapter 3). The composition of roadside grass was represented by an average composition from harvested spring and autumn grass (Ehlert et al., 2010).

2.2.3.2 Storage of manure and digestate
Table 2.2 shows the considered emissions of N (ammonia (NH_3), nitrous oxide (N_2O), nitrogen oxides (NO_x), and nitrogen gas (N_2)) and CH_4 from manure and digestate storage. As storage duration affects CH_4 emission and because pig manure used for AD was stored in the animal house shorter (yearly average of 1 month instead of 3 month without digestion), the difference of in-house CH_4 emissions was included in the assessment based on De Vries et al. (2012) (Chapter 3). Emissions of N_2O, NO, and N_2 occurring during in-house and outside storage of manure were not included, as these were assumed the same for both storage durations. Outside storage of raw manure and digestate (yearly average of 2 months) took place in a covered concrete tank. During storage, nutrient leaching (e.g. of nitrate NO_3^-, PO_4^{3-}, and K) was assumed negligible. Energy required for pumping manure and digestate was 1.2 kWh ton^{-1}.

2.2.3.3 Production of substituting products
The environmental impacts of producing substituting products comprised cultivation, processing, and transport. Since detailed inventory data are presented in the SI, general assumptions are presented here. Background (or generic) LCA data (e.g. heat and electricity production from fossil energy, mineral fertilizer production, herbicide production, diesel production, etc.) were taken from the Ecoinvent database v2.2 (EcoinventCentre, 2007).

Maize cultivation in S1 and S2 was based on average Dutch data. Maize was assumed to be fertilized with mineral fertilizer, as this is the marginal source for nutrients.

In S2, heat production occurred in an industrial gas-fired boiler >100 kW. The required natural gas (1.58 GJ) was computed based on an average lower heating value for glycerin of 16.5 MJ kg^{-1} and a boiler efficiency of 102% (EcoinventCentre, 2007).

In S3, barley production was based on average Dutch data (SI). As in S1 with maize, it was assumed that barley was fertilized with mineral fertilizer.

In S4, soybean production was based on Brazilian production circumstances occurring in the Central West and Southern region of Brazil (SI). Production of palm oil was based on production in Malaysia, as marginal source (SI). As in S3, barley production was based on average Dutch data.

Table 2.1. Composition of manure and co-substrates (kg ton^{-1} fresh matter) before and after digestion, OM decomposed and CH_4 produced

	DM	OM	N_{tot}	N_{min}	N_{org}	P_2O_5	K_2O	% OM decomposed	m^3 CH_4 produced ton^{-1} FM
Before digestion									
Pig manure after storage in animal house	90.3	60.3	7.35	4.35	3.0	4.2	7.2	38[e]	14.0[e]
Pig manure after outside storage[a]	90.0	60.0	7.20	4.20	3.0	4.2	7.2	-	-
Maize silage[a]	301	290	3.80	0.03	3.77	1.4	4.3	80[b]	115[g]
Glycerin[b]	842	794	n.d.	n.d.	n.d.	n.d.	n.d.	90[b]	406[h]
Beet tails[c]	136	110	2.30	0	2.30	0.7	2.3	80[b]	39.1[c]
Wheat yeast concentrate[d]	240	194	11.0	0.33	10.6	5.2	7.0	80[b]	88.6[d]
Roadside grass[f]	496	363	8.04	0	8.04	2.8	10.4	60[f]	38.5[f]
Composted roadside grass[f]	697	363	18.1	0	18.1	7.0	26.0	60[f]	-
After digestion									
Pig manure	67.4	37.4	7.52	5.06	2.46	4.3	7.4		
Maize silage	69.4	57.9	4.95	0.98	3.96	1.8	5.6		
Glycerin	127	79.4	n.d.	n.d.	n.d.	n.d.	n.d.		
Beet tails	48.0	22.0	2.52	0.50	2.02	0.8	2.5		
Wheat yeast concentrate	84.5	38.8	13.0	2.52	10.5	6.2	8.3		
Roadside grass	279	145	10.3	2.06	8.22	3.6	13.3		

n.d. = no data, ‘-‘ = not included, DM = dry matter, OM = organic matter, Nmin = mineral nitrogen, Norg = organic nitrogen FM = fresh matter.
[a] KWIN (2009-2010); [b] Assumptions based on Thomassen & Zwart (2008); [c] Kool et al. (2005); [d] Duynie (2008); [e] Timmerman et al. (2005); [f] Ehlert et al. (2010); [g] Amon et al. (2007); [h] Average of Thomassen and Zwart (2008) and Santibáñez et al. (2011).

Table 2.2. Emissions during storage and field application of products

	CH_4	NH_3-N	N_2O-N_{dir}	NO-N	N_2-N	N_2O-N_{ind}		NO_3-N	PO_4-P
	kg ton^{-1}	% TAN	% N	% N	% N	% NH_3-N + NO_x-N	% N leached	% N	% of P
Storage									
Pig manure storage in animal house	1.33[*,a] 0.29[#,a]	-	-	-	-		-	-	-
Pig manure outside storage	0.17[a]	2% N[b]	-	-	-	1.0[c]	-	-	-
Digestate outside storage	0.17[a]	2% N[b]	0.1[c]	0.1[b]	1.0[b]		-	-	-
Field application									
Pig manure	-	2.0[d]	1.3[f]		-			20.6[h]	
Digestate	-	2.0[d]	1.3[f]		-			19.8 - 22.5[h]	
CAN	-	2.5[d]	1.0[f]	0.55[g]	-	1.0[c]	0.75[c]	16.6[h]	0.6[i]
Urea	-	15.0[d]	1.0[c]		-			-	
Compost	-	5.8[e]	1.25[e]		-			26.6[h]	

'-' = not included, N_2O-N_{dir} = direct emission, N_2O-N_{ind} = indirect emission.
[a] De Vries et al. (2012),[*] = 3 months, [#] = 1 month storage; [b] Groenestein et al. (2012); [c] IPCC (2006); [d] Van Bruggen et al. (2011); [e] Brinkman et al. (2004); [f] Velthof and Mosquera (2010); [g] Stehfest and Bouwman (2006); [h] Based on Dekker et al. (2009), 19.8% (raw manure), 21.4% (S1), 21.2% (S2), 20.9% (S3), 22.5 (S4), and 22.3% (S5); [i] Based on the EDIP, 2003 method assuming that 10% P was leached and 6% reached the aquatic environment.

In S5, emission data for composting of roadside grass were based on the composting of municipal food waste (Brinkman et al., 2004). During composting, approximately 60% of the wet weight of roadside grass was reduced. Emission of CH_4, was considered to be 0.20 kg ton^{-1} of grass entering the facility, N_2O 1.5% of N, and NH_3 5.24% of N. In total, 10% of the N in the roadside grass was assumed to be lost during composting and 90% of the NH_3 to be removed with a bio-filter (Brinkman et al., 2004). Per ton of grass, energy requirement of composting was 29 kWh. Leaching of NO_3^-, PO_4^{3-}, and K during composting was excluded due to a sealed concrete floor.

2.2.3.4 Anaerobic digestion

AD was considered to take place in two stages in a continuous stirred tank reactor (CSTR). Operation occurred at mesophilic temperature (around 35°C) with a hydraulic retention time of 60 days, based on common practice in The Netherlands, and required 66 MJ electricity per ton of substrate and 110 MJ heat per ton of substrate (Börjesson & Berglund, 2007; Peene et al., 2011). The CHP had an electric capacity of 500 kW with an overall energetic efficiency of 80% and an electric efficiency of 35%. Electricity used for the process was assumed to be taken from the grid, whereas produced electricity was supplied to the grid. The required heat for the digestion process resulted from the heat produced from the biogas engine. Of the remaining heat, 50% was assumed to avoid marginal heat.

During AD, organic N in the substrates is partly converted into mineral N (N_{min}). It was assumed that 20% of the organic N in the initial substrate was mineralized (De Vries et al., 2012, Chapter 3).

Fugitive emissions of CH_4 from the digestion plant and the biogas engine were assumed to be 1.5% of total produced CH_4 (i.e. 1% from the digestion plant and 0.5% from the gas engine) (IPCC, 2006). Emission of N_2O from the biogas engine was 0.1 kg TJ^{-1} of electricity produced, whereas emission of NO_x was 0.42 g m^{-3} of biogas produced (VROM, 2010).

2.2.3.5 Application of products and avoided mineral fertilizer

Manure and digestate were assumed to be injected into arable land. For the avoided compost (S5), it is assumed that it would have been applied by a solid manure spreader to arable land, whereas mineral fertilizers would have been applied with a broadcast spreader. Data on the environmental impacts of field application processes were taken from the Ecoinvent database (EcoinventCentre, 2007).

Emissions of N and PO_4^{3-} during application and the N fertilizer replacement values (NFRVs) for manure, digestate, compost, and mineral fertilizers are presented in Table 2.2. The NFRVs were considered to be: 65% for manure and digestates, 15% for compost, and 100% for mineral N fertilizer (Brinkman et al., 2004; DR, 2012). The replacement value for P and K was assumed to be 100% for all products (De Vries et al., 2012, Chapter 3).

2.2.3.6 Transport of products

Transport of products occurred by lorry (16 - 32 tons) between several life cycle stages (Fig. 2.1). For the transport of the raw manure to the AD plant a distance of 31 km was considered, whereas an average distance of 104 km was assumed for transporting the digestates to agricultural fields. This 104 km was a weighted average of local transport

(34% over 31 km), external transport within the Netherlands (48% over 120 km) and transport outside of the Netherlands (18% over 200 km) (De Vries et al., 2012, Chapter 3; Peene et al., 2011). All other products were assumed to be transported over 50 km.

2.2.3.7 Land use change emissions

LUC emissions related to displaced cultivation of barley in S1 - S4 were based on Tonini et al. (submitted for publication), who related the displacement of spring barley in Denmark to the conversion of specific biomes worldwide and quantified the associated greenhouse gas emissions (SI). This was assumed to be representative for Western Europe. The net land expansion was adjusted by using the average Dutch yield for barley leading to 1.22 ha expanded per ha of displaced barley. An average LUC factor of 309 ton CO_2 ha^{-1} of displaced barley was applied, which was annualized over a 20 years period (1.55 kg CO_2 m^{-2} y^{-1}). LUC emissions in S4 for soybean cultivation were calculated specifically for this study (i.e. 1.67 ton CO_2 ton^{-1} soybeans y^{-1}, or 0.47 kg CO_2 m^{-2} y^{-1}). For the calculations, it was assumed that 20% of the increased soybean demand resulted from yield increase, whereas 80% resulted from an increase in soybean area in the tropical open forest in Brazil (23%) and savanna in the Cerrado region (77%) (SI). Similarly, the LUC emission for reduced palm fruit production (i.e. 1.95 ton CO_2 ton^{-1} palm fruit y^{-1}, or 3.7 kg CO_2 m^{-2} y^{-1}) was assumed to result from a 30% yield increase and from 70% increase in cultivation area (i.e. 70% conversion of tropical moist forest in in Southeast Asia and 30% conversion of peat land, see SI).

2.2.3.8 Soil carbon storage

Soil C storage was included based on Hermann et al. (2011) (SI). Changes in soil C storage, and thus in the amount of C released as CO_2 to the atmosphere, occur due to differences in C composition among manure, digestate, and compost. During digestion, C is converted into CH_4 and CO_2 resulting in less C applied to the soil with the digestate and thus less C to be stored in the soil compared to undigested matter. Compared to manure, an increased proportion of C is converted to relatively stable humus-C in the soil by grass-based compost (Hermann et al., 2011). It was assumed that 35% of the C in the manure and digestates (representing 50% of OM) was converted into humus-C. For grass compost this was assumed to be 51%. The C converted to humus-C was assumed to remain in the soil C pool over a time horizon of 100 years (Hermann et al., 2011) (SI).

2.2.4 Sensitivity analysis

The sensitivity of results and conclusions to several parameters were tested: minimum and maximum values for LUC, higher fugitive methane emissions from the digestion facility, a higher electric efficiency of the biogas engine, and increased NFRV of the digestates.

2.2.4.1 Minimum and maximum values for land use change

LUC emission factors contain considerable uncertainty depending on the applied methods and data, and therefore will have a considerable effect on final results (Plevin et al., 2010). The uncertainty range from Tonini et al. (submitted for publication) was used to address the minimum and maximum values for LUC emission related to displacement and production of spring barley (S1 - S4), i.e. a minimum of 140 ton CO_2 ha^{-1} (0.70 kg CO_2 m^{-2} y^{-1}) and maximum of 477 ton CO_2 ha^{-1} (2.38 kg CO_2 m^{-2} y^{-1}). Furthermore, deforestation related to soybean cultivation has shown signs of decrease in recent years due to policy changes by Brazilian government (Prudêncio da Silva et al., 2010). To consider a minimum LUC factor for soybean production, data from Prudêncio da Silva et al. (2010) were used who reported an average value of 0.058 kg CO_2-eq m^{-2} (0.28 ton CO_2-eq ton^{-1} soybeans). This factor included 1% of land used for soybean production transformed from rainforest and 3.4% transformed from Cerrado. For a minimum LUC value related to Malaysian palm fruit cultivation, data from the Ecoinvent database were used, i.e. 0.47 kg CO_2 m^{-2} (0.25 ton CO_2 ton^{-1} palm fruit produced, related to the conversion of Malaysian rainforest (SI)) (EcoinventCentre, 2007).

To consider maximum LUC values for soybean cultivation, the same method as in the baseline scenario was applied. However, instead of tropical open forest being converted, only tropical evergreen forest was assumed to be converted (i.e. includes more C and leads to higher CO_2 emission, see SI). Furthermore, the increase in demand was met by 100% expansion, i.e. no yield increase was considered. This led to a value of 14.5 ton CO_2 ton^{-1} soybeans y^{-1} (4.1 kg CO_2 m^{-2} y^{-1}). Maximum LUC values for palm fruit were calculated by assuming expansion on 50% peat land and 50% tropical moist forest (SI). Furthermore, as with soybeans, the increase in demand was met by 100% expansion. Moreover, a higher emission from peat land was considered (112 ton CO_2 ha^{-1} y^{-1}, SI) resulting in a maximum (avoided) LUC emission of 4.33 ton CO_2 ton^{-1} palm fruit y^{-1} (8.2 kg CO_2 m^{-2} y^{-1}).

2.2.4.2 Fugitive methane emissions from the digestion plant

Fugitive emissions of CH_4 from the digestion plant increase the amount of GHG released to the atmosphere and consequently reduce the bio-energy produced. Fugitive emissions have been reported to be as much as 5.2% of produced CH_4 under normal operating conditions (Flesch et al., 2011). To consider the magnitude of this impact, emission of CH_4 from the digestion plant was increased from 1% to 5%.

2.2.4.3 Electric efficiency of the biogas engine

Electric efficiency of the biogas engine is important with regard to the amount of electricity (versus heat) produced, and consequently the amount of avoided fossil fuels. The electric efficiency was increased from 35% to 45% to consider the effect of more displaced fossil-based electricity. The total efficiency was kept equal to the baseline scenario.

2.2.4.4 NFRV of digestates

The NFRV of digestate is often said to be higher compared to undigested manure due to mineralization of N, increasing its availability for crops (Hamelin et al., 2011). To consider the impact of a higher fertilizing capacity of the digestate, the NFRV of the digestates was increased from 65% to 75% in the sensitivity analysis.

2.2.5 Life cycle impact assessment

All emissions and resources used were included in the assessment and categorized under seven environmental impact categories: climate change (CC in kg CO_2-eq; including CO_2, CH_4, and N_2O), terrestrial acidification (TA in kg SO_2-eq; including sulfur dioxide (SO_2), NO_x, and NH_3), marine eutrophication (ME in kg N-eq; including NO_3^-, NO_x, and NH_3), freshwater eutrophication (FE in kg P-eq; including PO_4^{3-}); particulate matter formation (PMF in kg PM_{10}-eq; including particulates < 10 µm and NH_3, SO_2, and NO_x as precursors), fossil fuel depletion (FFD in MJ), and land use (m^2). The scenarios and impact assessments were modeled and computed in Microsoft Excel and by using the ReCiPe midpoint v.1.04 impact assessment method (Goedkoop et al., 2009).

2.3 RESULTS AND DISCUSSION

2.3.1 Impact assessment of anaerobic mono- and co-digestion

2.3.1.1 Climate change

Mono-digestion reduced CC by 16 kg CO_2-eq per ton of substrate, as compared to the situation where manure is not digested (Fig. 2.2). This was mainly due to avoided manure storage and application (i.e. altered management), but also to avoided fossil-based electricity and heat. Addition of co-substrates in S1 - S4 increased CC (36 - 105 kg CO_2-eq) mainly as a result of induced LUC. LUC contributed 104 - 199 kg CO_2-eq in S1 - S4. In S5, the reduction in CC of 89 kg CO_2-eq resulted from avoided fossil-based electricity and heat and the avoided composting. Avoided composting prevented emission of N_2O and the energy use that occurs during composting. In all scenarios, net transport contributed 16 - 19 kg CO_2-eq, whereas the AD process contributed 18 - 42 kg CO_2-eq.

2.3.1.2 Fossil fuel depletion

Mono-digestion reduced FFD by 117 MJ, as compared to the situation where manure is not digested, reflecting a net reduction in energy use (46 kWh electricity was produced and 51 MJ of heat avoided in S1) (Fig. 2.2). Addition of co-substrates in S1 - S3 and S5 increased bio-energy production and resulted in higher reductions of FFD, with S1 having the highest reduction (FFD of 2398 MJ). Despite having the highest bio-energy production (305 kWh electricity produced and 651 MJ of heat avoided), S2 did not lead to a lower FFD (-1992 MJ) than S1 (210 kWh electricity produced and 431 MJ heat avoided). This was mainly caused by the bio-energy that would have been produced by the direct combustion of the glycerin (1.65 GJ), which had to be substituted by an equivalent amount of energy from natural gas. Despite bio-energy production in S4 (167 kWh electricity produced and 332 MJ heat avoided), the scenario had a net increase in energy demand due to production and transport of soybean meal. S5 (86 kWh electricity produced and 143 MJ heat avoided) had, after S1 and S2, the highest reduction in FFD (-1027 MJ), whereas S3 (87 kWh electricity produced and 145 MJ heat avoided) showed approximately half of the reduced FFD (-550 MJ) found in S5. In S5, this result is mostly due to the shift from energy consumption during composting to bio-energy production from AD. In all scenarios, transport and AD required roughly 200 - 300 MJ of energy.

2.3.1.3 Terrestrial acidification

Compared to raw manure application, mono-digestion increased TA by 0.09 kg SO_2-eq, through a higher NH_3 emission from digestate application. This increase resulted from a higher N_{min} content in the digestate. Addition of co-substrates in S1 - S3 generally showed equal performance as the mono-digestion scenario. In these scenarios, increased TA from the production of the substitute and the storage and application of the digestate was decreased by avoided electricity and heat. In S4, TA was considerably higher (1.61 kg SO_2-eq) due to emissions of NH_3, NO_x, and SO_2 from the production of the soybeans. S5 was the only scenario that reduced TA (-0.33 kg SO_2-eq), mainly due to the avoided NH_3 emissions during composting and application of compost.

2.3.1.4 Marine eutrophication

ME was approximately neutral for mono-digestion (-0.03 kg N-eq) as compared to the situation where manure is not digested (Fig. 2.2). For mono-digestion, the reduced ME from avoided manure storage and application was similar to the ME from digestate storage and application. ME increased with the addition of co-substrates in S1 - S4 (0.28 - 2.38 kg N-eq). This was mainly due to two factors: (1) emission of NO_3^- during production of the co-substrates and (2) addition of N from the co-substrates, which increased the N application from digestate. In the mono-digestion scenario as well as in S1, S2, and S3, ME was little affected (maximum of -0.2 kg N-eq) by the net mineral fertilizer production; in S4, however, reduced ME from net mineral fertilizer production was considerably higher (-0.6 kg N-eq) due to the high nutrient content in the WYC (Fig. 2.2). A net reduction of ME (-0.40 kg N-eq), again related to avoided production and application of compost, was seen in S5. The digestate in S5 was considered to have a higher nitrogen fertilizer value compared to compost, thus consequently reducing more mineral N fertilizer and thereby leaching of NO_3^-.

2.3.1.5 Freshwater eutrophication

Mono-digestion reduced FE (-0.012 kg P-eq), compared to the situation where manure is not digested, mainly as a result of avoided electricity production. Addition of co-substrates in S1 - S5 further reduced FE (-0.021 - -0.077 kg P-eq), but not in S4 due to the cultivation of soybeans contributing to FE through leaching of PO_4^{3-}, as relatively high emission per kg of dry matter occurs (SI). In S5, FE was reduced mainly through avoided electricity from energy production during AD, and through the avoided electricity from composting.

2.3.1.6 Particulate matter formation

Mono-digestion, as compared to the situation where manure is not digested, resulted in negligible PMF (0.02 kg PM_{10}-eq); for mono-digestion, emissions of NO_x from transport and NH_3 from digestate storage and application were counteracted by a reduction in PM_{10}, NO_x, and SO_2 emissions from the substituted fossil fuels. Addition of co-substrates in S1 - S5, decreased PMF (0.01 - -0.13 kg PM_{10}-eq), except for S4 (0.36 kg PM_{10}-eq) where more emissions of NH_3, NO_x, and SO_2 occurred during production and transport of soybean meal. In S5, reduced NH_3 emissions from composting resulted in reduced PMF.

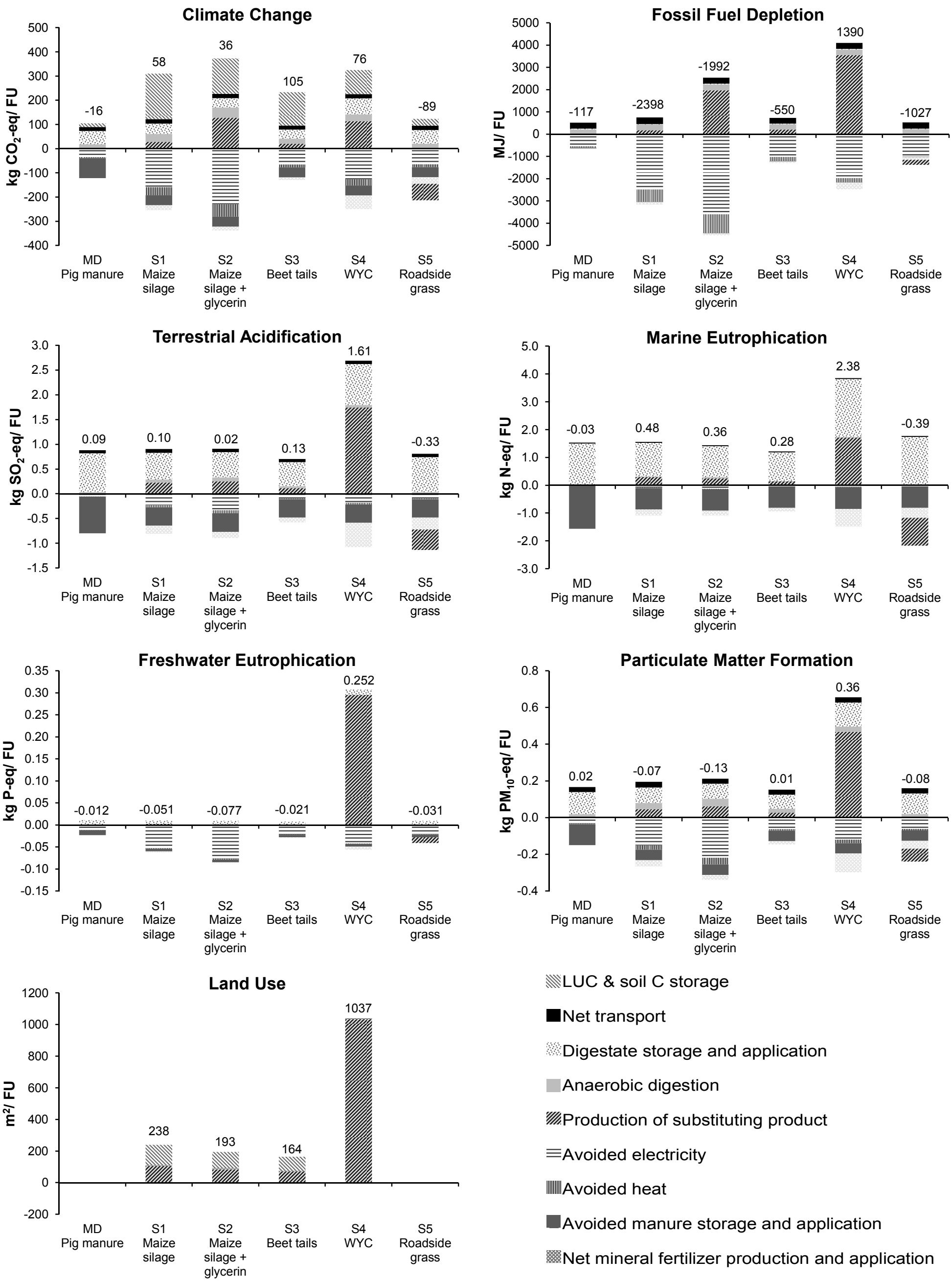

Fig. 2.2. Impact assessment results for the scenarios. Numbers above the bars present the net results. Net transport represents the induced and avoided transports as shown in Fig. 2.1 (excluding transport during co-substrate production).

2.3.1.7 Land use

Mono-digestion had negligible land use, because no co-substrates were used and, therefore, no substitute was required (land use in this case was only related to production facilities and capital goods production). S5 also had negligible land use as the roadside grass is a residual product that does not interact with crops and thus land use. Addition of co-substrates in S1 - S4 increased land use directly for cultivation of respectively maize (S1, 106 m^2 and S2, 85 m^2), barley (S3, 73 m^2) and soybeans (S4, 1037 m^2; increased cultivation of soybeans (1153 m^2), avoided oil palm cultivation (122 m^2), increased barley cultivation (4.4 m^2), and land use related to capital goods (2 m^2)). Furthermore, land use expanded in S1 - S3 due to displaced cultivation of barley (respectively 129, 103, and 89 m^2) (Fig. 2.2). S4 had the highest land use, which was due to the relative low DM yield per ha of cultivated soybeans compared to maize and barley (SI).

2.3.2 Sensitivity analysis

2.3.2.1 Minimum and maximum values for land use change

In case of minimum values for LUC, net CC decreased up to 109 kg CO_2-eq compared with base line results. This meant that in S1, S2 and S4, CC was reduced more than mono-digestion (Table 2.3). In S1, the impact of reducing LUC emissions was highest (a change of 109 kg CO_2-eq), because of the displaced barley and the related LUC emission factor, i.e. higher than for soybeans (section 2.3.7). In case of maximum LUC values, net CC increased up to 3730 kg CO_2-eq compared with base line results in S1 - S4. This increase was highest in S4 (i.e. 3654 kg CO_2-eq). These results indicate that the assumed LUC factor had a major impact on the conclusions of this study with regard to CC.

2.3.2.2 Fugitive methane emission from the digestion plant

Increasing fugitive CH_4 emissions from the digestion plant increased CC (11 - 75 kg CO_2-eq) and FFD (27 - 182 MJ) for all scenarios (Table 2.3). For mono-digestion, this offset almost all GHG reduction. These results indicate that controlling emission of CH_4 from the digestion process is essential to maximize the advantage concerning CC.

2.3.2.3 Electric efficiency CHP

Increasing the electric efficiency of the CHP reduced CC in all scenarios (up to 53 kg CO_2-eq, Table 2.3), FFD (up to 820 MJ), and, to a lesser extent, also FE (up to 0.021 kg P-eq). These results indicate an opportunity in further improving the environmental performance of AD by increasing efficient conversion of biogas into electricity.

2.3.2.4 NFRV of the digestate

Increasing the NFRV of the digestate by 10% decreased CC, TA, ME, and FFD in all scenarios, by up to respectively 13 kg CO_2-eq, 0.105 kg SO_2-eq, 0.16 kg N-eq, and 53 MJ. This decrease was mainly due to a higher amount of mineral N fertilizer that was substituted by the digestate, compared to the baseline results. In general, however, this did not change the ranking of the scenarios, although TA in S2 became lower than in the case of mono-digestion.

2.3.3 General discussion

2.3.3.1 Mono-digestion compared to co-digestion

Overall, mono-digestion of pig manure performed well from an environmental perspective as bio-energy was produced and most impact categories either remained neutral (ME, PMF, and land use) or were reduced (CC, FFD, and FE), compared to conventional storage and field application of raw manure. Bio-energy production by mono-digestion, however, was relatively low compared to co-digestion which was also observed by other authors (Thyø & Wenzel, 2007).

Adding co-substrates increased bio-energy production compared to mono-digestion, but showed that the environmental impact of producing the substitute was important for most impact categories. This notably applied for S4 where the addition of WYC resulted in increased environmental impact. As a protein-rich substrate, WYC competes with soy meal on the international market, and its production has a relatively high environmental impact (Prudêncio da Silva et al., 2010). Co-digestion with WYC, therefore, does not represent an attractive alternative to increase bio-energy production. On the other hand, in S5 the addition of roadside grass increased bio-energy production compared to mono-digestion and reduced all environmental impact categories. For roadside grass, moving to AD compared to composting represented improved management. As other studies have confirmed, anaerobic (co-)digestion is a better environmental alternative than composting for waste or residue management (Hermann et al., 2011; Patterson et al., 2011). This indicates that from an environmental perspective, such wastes and residues are preferred as co-substrates.

Adding maize silage, glycerin, and beet tails in S1 - S3 also represented attractive alternatives to increase bio-energy production and reduce FFD, FE, and PMF compared to mono-digestion. However, these scenarios led to increased CC, ME, and land use. For CC, LUC increased GHG emissions and reduced the attractiveness of maize silage and beet tails for co-digestion, meaning LUC must be considered when selecting a co-substrate. Moreover, using maize silage and beet tails adds nutrients to the total substrate and thus to the digestate. This may increase nutrient surpluses in areas where the digestate is produced, especially in cases where there is already a surplus of nutrients (i.e. in intensive livestock production areas, such as the Netherlands).

Overall, anaerobic mono-digestion of pig manure and co-digestion with wastes or residues presents a good opportunity to produce bio-energy and reduce environmental impact. However, co-digestion with potential animal feed stocks, increases the impact by inducing the need of a substitute and should, therefore, be avoided. Furthermore, to increase bio-energy production from mono-digestion, additional manure processing by means of, e.g. separation or pretreatment can be applied. Such treatment can be used to concentrate the decomposable organic matter in a single fraction or increase the fraction of decomposable organic matter. Including such additional processing should be evaluated from a life cycle perspective as high energy demands may counteract the produced bio-energy (De Vries et al., 2012, Chapter 3; Hamelin et al., 2011).

2.3.3.2 Sensitivity of the results

Uncertainty of the LUC magnitudes on CC appeared to be crucial for the co-digestion scenarios with maize silage and WYC (S1, S2, and S4), as conclusions for CC were altered for these.

It is, thus, of great importance to improve LUC estimations through further analysis in order to understand if the “real life” values are closer to the minimum or to the maximum of the range presented in Table 2.3. In any case, however, LUC contribution to CC is not zero, and as such should be addressed where crops or any substrate inducing a need for more crop production are used as substrates for bio-energy production (Plevin et al., 2010).

Fugitive CH_4 emissions from the digestion plant should be kept to a minimum, as these affect both CC and bio-energy production. These effects were also observed by Patterson et al. (2011) who pinpointed the effect of fugitive CH_4 emissions on CC during upgrading of the biogas. Current data of fugitive emissions, however, are generally based on rough estimates or few measurements (Flesch et al., 2011). Verification by further measurements on different types of digesters would, therefore, be essential for future LCAs on biogas production.

With respect to the assessed scenarios, increased electric efficiency of the gas engine did not change the ranking and conclusions. However, improving the electric efficiency of biogas engines, as this sensitivity analysis highlights, represents one option to enhance the overall environmental performance of biogas installations (more than 100% improvement for CC and FFD).

Increased NFRV of the digestates also did not alter the conclusions of the study. Nevertheless, increasing the fertilizing capacity of the digestate and its related management will improve the overall environmental performance of AD scenarios. However, NFRV levels are strongly dependent on local factors, such as climate, soil type and crop rotation and the related management of the digestate (i.e. method and timing of application) and should be considered site specific (De Vries et al., 2012, Chapter 3).

Other parameters affecting final results, but not included in the sensitivity analysis, include methane yields of the co-substrates and the initial use and related marginal suppliers for the substitutes. Methane yields will differ upon the quality and origin of the substrate, but also the type and management of the digestion process. In general, higher yields will mainly lead to lower CC and FFD and vice versa.

The environmental impact related to producing the substitute needed for the co-substrate used for AD will depend on variables, such as the extent of substitution and the product properties (Weidema, 2003). When, for example, the WYC in S4 has a low quality and as a result cannot be utilized for animal feed, it becomes a waste that would otherwise be composted or landfilled. In such a case, using WYC for bio-energy production by AD would lead to a much more environmentally sustainable result, as no interaction with feed would occur. Furthermore, the substitution ratio of co-substrates initially used as animal feed (i.e. how much feed is displaced per kg of WYC or beet tail taken away for AD) also depends on factors, such as digestibility, market prices, etc., and not solely on the protein and energy content. In most cases, a new feed ration will be computed in order to optimize prices, and product characteristics. In S3, for example, beet tails in animal feed might also be compensated by adding maize and grass silage, if these happen to be cheaper or more practical. The initial use of co-substrates should therefore be analyzed for each site-specific, geographical, temporal and economical context.

Table 2.3. Results of the sensitivity analysis in absolute values per functional unit (1 ton substrate) for the scenarios (only most affected impact categories are presented)

		Values for land use change emissions		Fugitive CH_4 emission from AD 5%		Electric efficiency +10%			NFRV digestates +10%			
		Minimum	Maximum									
Scenario		CC	CC	CC	FFD	CC	FE	FFD	CC	TA	ME	FFD
MD	bsl	-16		-16	-117	-16	-0.012	-117	-16	0.09	-0.03	-117
	se	-	-	-5	-90	-24	-0.015	-240	-27	0.003	-0.16	-160
S1	bsl	58		58	-2398	58	-0.051	-2398	58	0.10	0.48	-2398
	se	-51	166	110	-2272	22	-0.066	-2963	50	0.037	0.39	-2430
S2	bsl	36		36	-1992	36	-0.077	-1992	36	0.02	0.36	-1992
	se	-51	123	111	-1810	-17	-0.098	-2812	29	-0.041	0.27	-2022
S3	bsl	105		105	-550	105	-0.021	-550	105	0.13	0.28	-550
	se	30	180	126	-498	90	-0.027	-782	98	0.075	0.20	-578
S4	bsl	76		76	1390	76	0.252	1390	76	1.61	2.38	1390
	se	-17	3730	117	1490	47	0.241	940	63	1.51	2.22	1337
S5	bsl	-89		-89	-1027	-89	-0.031	-1027	-89	-0.33	-0.39	-1027
	se	-	-	-68	-976	-103	-0.037	-1258	-99	-0.41	-0.52	-1072

'-' = no change, MD = mono-digestion, bsl = baseline results, se = results of the sensitivity analysis, CC = climate change (kg CO_2-eq), FFD = fossil fuel depletion (MJ), FE = freshwater eutrophication (kg P-eq), TA = terrestrial acidification (kg SO_2-eq), and ME = marine eutrophication (kg N-eq), NFRV = nitrogen fertilizer replacement value.

2.3.3.3 Substrate availability and potential for bio-energy production and GHG mitigation
The availability of the substrates is limited (in 2010 in the Netherlands: approximately 11800 kilotons (kt) of pig manure, 125 kt of beet tails, 4 kt of glycerin, 250 kt of WYC, and 700 kt of roadside grass), as these are constrained resources depending on the production of another main product. This will limit both the total bio-energy production potential by means of AD of these substrates, but also economic viability is limited due to competition with other markets and because bio-energy is highly subsidized. Currently, prices of co-substrates are increasing strongly due to higher demand. This strengthens the point made earlier that greater focus on mono-digestion is needed, in particular on the development of technologies to enhance biogas production from manure. However, the total bio-energy production potential and GHG reduction potential of mono-digestion remains relatively low. If all pig manure in the Netherlands were mono-digested, this would represent about 0.5% of the total electricity use (117 billion kWh used in 2010 or 2.2% of the electricity consumed by households, 24.7 billion kWh in 2010) and a reduction of 0.1% of total emitted GHGs (211 Mton emitted in 2010 or 2.2% of agricultural GHG emissions, 8.6 Mton in 2010). Including the digestion of all other substrates in this study (assuming only half of the maize silage produced), and additionally all cattle manure, would roughly double the electricity production and reduce GHGs slightly more. Hence, anaerobic mono- and co-digestion of manure and co-substrates provides a potential to produce bio-energy and reduce environmental consequences, but on its own cannot fulfill increasing future bio-energy demands.

/ 2.4 CONCLUSIONS

Anaerobic mono-digestion of pig manure produced bio-energy and improved overall environmental performance as compared to conventional manure management, but represents a limited source for bio-energy. Co-digestion with maize silage and beet tails, which compete with animal feed, and glycerin increased bio-energy production (up to 568%) and reduced terrestrial acidification, particulate matter formation, and freshwater eutrophication, but increased marine eutrophication, land use and climate change (through LUC). Co-digestion with wheat yeast concentrate, a protein-rich co-substrate substituted with soymeal, increased all environmental impacts. Co-digestion with roadside grass, a residual or waste product, appeared most environmentally sustainable for increasing bio-energy production of mono-digestion.

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APPENDIX A. SUPPLEMENTARY DATA

Supplementary data associated with this article can be found in the online version, at dx.doi.org/10.1016/j.biortech.2012.08.124.

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Chapter 3

// ENVIRONMENTAL CONSEQUENCES OF PROCESSING MANURE TO PRODUCE MINERAL FERTILIZER AND BIO-ENERGY

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ABSTRACT

Liquid animal manure and its management contributes to environmental problems such as, global warming, acidification, and eutrophication. To address these environmental issues and their related costs manure processing technologies were developed. The objective here was to assess the environmental consequences of a new manure processing technology that separates manure into a solid and liquid fraction and de-waters the liquid fraction by means of reverse osmosis. This results in a liquid mineral concentrate used as mineral nitrogen and potassium fertilizer and a solid fraction used for bio-energy production or as phosphorus fertilizer. Five environmental impact categories were quantified using life cycle assessment: climate change (CC), terrestrial acidification (TA), marine eutrophication (ME), particulate matter formation (PMF), and fossil fuel depletion (FFD). For pig as well as dairy cattle manure, we compared a scenario with the processing method and a scenario with additional anaerobic digestion of the solid fraction to a reference situation applying only liquid manure. Comparisons were based on a functional unit of 1 ton liquid manure. System boundaries were set from the manure storage under the animal house to the field application of all end products. Scenarios with only manure processing increased the environmental impact for most impact categories compared to the reference: ME did not change, whereas, TA and PMF increased up to 44% as a result of NH_3 and NO_x emissions from processing and storage of solid fraction. Including digestion reduced CC by 117% for pig manure and 104% for dairy cattle manure, mainly because of substituted electricity and avoided N_2O emission from storage of solid fraction. FFD decreased by 59% for pig manure and increased 19% for dairy cattle manure. TA and PMF remained higher compared to the reference. Sensitivity analysis showed that CH_4 emission from manure storage, NH_3 emission during processing, and the replaced nitrogen fertilizer by the mineral concentrate were important parameters affecting final results. It was concluded that processing fattening pig and dairy cattle manure to produce mineral fertilizer increased overall environmental consequences in terms of CC (except for dairy cattle manure), TA, PMF, and FFD compared to current agricultural practice. Adding the production of bio-energy reduced CC and FFD. Only when NH_3 emission from processing was low and bio-energy was produced, overall equal or better environmental performance was obtained for TA and PMF. It was emphasized that real time measurements should be done to enhance the environmental assessment of manure processing technologies. Results of this study present the full environmental consequences of manure processing and key parameters affecting the environmental impact of manure management. Outcomes can be used for decision making and further tackling of environmental problems related to manure management.

3.1 INTRODUCTION

The environmental impact from animal manure and its management (i.e., storage and application) has increased considerably through growth of livestock production worldwide. In the Netherlands, for example, national production of pig and dairy cattle manure increased from about 46 million tons in 1950 to 68 million tons in 2009 (CBS, 2011). Manure contributes to the following environmental impacts: acidification and particulate matter formation, mainly through volatilization of ammonia (NH_3) and nitrogen oxides (NO_x); climate change through emissions of greenhouse gases (GHG); eutrophication, mainly through leaching of nitrate (NO_3^-) and phosphate (PO_4^{3-}) to soil and surface water; and depletion of fossil energy sources as a result of management (Prapaspongsa et al., 2010; Sandars et al., 2003; Thomassen et al., 2008).

These environmental impacts have led to international and national regulations (e.g., Gothenburg Protocol, NEC-Directives, and Nitrates Directive) designed to reduce emissions related to animal manure and management. This has induced surpluses in several regions of the world including the Netherlands, increasing manure removal costs for farmers. To decrease these costs and the environmental impact, manure processing technologies have been developed, including anaerobic digestion (AD), biological treatment, composting, incineration, and gasification (Burton & Turner, 2003). These technologies were mainly developed to reduce GHG emissions, NH_3 volatilization and fossil fuel depletion by producing bio-energy. However, the whole life cycle of these technologies, including the storage and application of end products should be addressed to evaluate their true environmental performance.

The environmental impact of manure processing technologies has been analyzed along the entire life cycle of the manure and its end products by means of life cycle assessment (LCA) in several studies (Hamelin et al., 2011; Lopez-Ridaura et al., 2009; Prapaspongsa et al., 2010). GHG emissions were reduced through AD of manure as a result of bio-energy production (electricity and heat) and the substitution of mineral fertilizer. Reductions by up to 147 kg carbon dioxide equivalents (CO_2-eq.) per ton of pig manure and 104 kg CO_2-eq per ton of solid fraction of separated pig manure were reached through AD (Hamelin et al., 2011; Prapaspongsa et al., 2010).

Acidification and eutrophication potentials did not vary, or very little, when digestion was applied. On the other hand these potentials have been shown to be increased through aeration of the liquid fraction from separated manure combined with composting of the solid fraction (Lopez-Ridaura et al., 2009).

A manure processing technology using liquid and solid separation and reverse osmosis (RO), currently being developed and investigated in the Netherlands, aims at producing a liquid nitrogen (N) and potassium (K) concentrate. The process produces as main products: mineral concentrate (MC), considered to have similar fertilizing properties as mineral N and K fertilizer, and a solid fraction that can be used as a substrate for AD and as a phosphorus (P) fertilizer. Although LCA studies have focused on the environmental impact of some manure processing technologies, the impact of this process has not been investigated.

The objective of this study was to assess the environmental impact of this new manure processing method for fattening pig and dairy cattle manure, and to compare it to conventional manure management practices. We used LCA to determine and compare the environmental impact of manure processing to produce mineral fertilizer, with and without AD, and compared it to current agricultural practice.

3.2 MATERIALS AND METHODS

3.2.1 LCA approach and functional unit

Life cycle assessment is a method to determine the environmental impact of a system providing a product or service. An LCA includes all pollutants and consumptions of finite resources from each stage in the life cycle, and allows a comparative analysis of the environmental impact of different production scenarios (ISO-14040, 2006). In particular, the LCA in this study aimed at assessing the environmental consequences of moving to a manure management system including manure processing relative to a reference without processing. We, therefore included in the analysis the change in environmental impact of all processes (also called marginal processes or suppliers) affected by this change in manure management (Weidema et al., 1999).

For a comparative assessment, the environmental impact is related to a functional unit (FU) that expresses the function of the system in quantitative terms. The function of the system is to process liquid manure into a MC that can be applied as mineral N and K fertilizer and a solid fraction suitable for bio-energy production or application as P fertilizer. As the available manure was the starting point, a FU of 1 ton untreated liquid fattening pig or dairy cattle manure was applied. The same chemical composition of manure was used in the references and the scenarios. This ensured that in all cases equal amounts of nutrients and dry matter were introduced into the system.

3.2.2 Manure management system and scenarios

3.2.2.1 System boundaries

The LCA included the environmental impacts from manure storage in the animal house and outside storage; processing of manure; storage, distribution, and field application of the end products; and the transport of materials between different life cycle stages (Fig. 3.1). To assess changes in distribution and transport distances, we distinguished between four locations for product application: local application on a dairy farm with grassland, local application on an arable or a dairy farm with arable land, external (i.e. off farm) application on an arable farm, and application on an arable farm outside the Netherlands. The system further included environmental impacts related to production of chemicals used for processing (e.g., flocculants), consumed electricity, and substituted electricity in the case of bio-energy production. The system boundaries also encompassed impacts from the production, transport, and application of avoided mineral fertilizer, i.e., N, P and K avoided from mineral fertilizer as a result of using nutrients from manure. The analysis further included environmental emissions and resource use from the production of capital goods except for the manure storage and processing plants.

The system excluded the impacts from animal production, as we assumed that a change in the animal production sector would not be driven by a change in manure management. Additionally, biogenic CO_2 emissions were not incorporated in the calculations as they are considered to be short cyclic carbon taken up by crops earlier (IPCC, 1997). The emission of P was excluded because manure processing was assumed to not affect the total amount of P in manure and end products. Input of P to the soil and crop and output through leaching, therefore, were the same for all references and scenarios.

Marginal products for mineral fertilizer production were assumed to be: ammonium nitrate for N, triple superphosphate for phosphorus pentoxide (P_2O_5), and potassium chloride for potassium oxide (K_2O). Marginal electricity production was based on current Dutch statistics and EU production outlooks from the International Energy Agency. The long term marginal electricity source for the Netherlands was estimated to be a mix of coal (28%), natural gas (67%), and wind (5%) (IEA, 2011; IEA, 2008). The utilization of excess heat from AD, i.e. heat produced in addition of the required heat for the process, was not included as heat offset possibilities are still limited in the Netherlands (Dumont, 2010).

3.2.2.2 Manure processing

Manure processing was done in five full-scale pilot plants operating in the Netherlands (Hoeksma et al., 2011). These pilot plants processed up to 50,000 tons of manure annually and aimed at producing a concentrated N-K liquid and a remaining solid fraction mainly through three processing steps: 1. separation of solids and liquids by means of dissolved air floatation, 2. separating the liquid from the solid remains by a sieve belt press or a screw press, and 3. de-watering of the effluent with reverse osmosis (Fig. 3.2). The plants produce three end products: MC, solid fraction, and permeate, i.e., water remaining after reverse osmosis. The MC and solid fraction were applied in crop production as fertilizers.

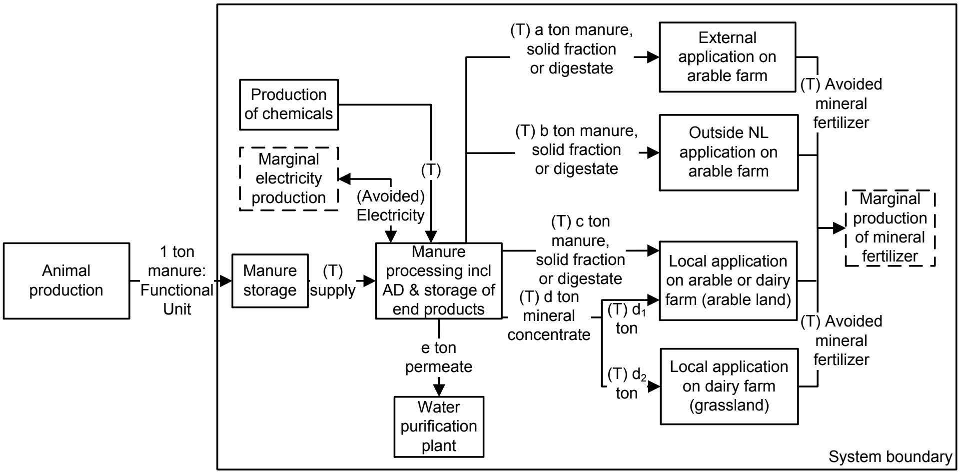

Fig. 3.1. Manure management system with input of 1 ton pig or dairy cattle manure and distribution of end products as considered in the assessment. Black arrows represent mass flows of materials. The two-way arrow for electricity production and consumption points out that electricity is consumed as well as produced. (T) represents transportation.

The solid fraction was also used as a substrate for AD to produce bio-energy where after it was applied. Permeate was treated in a water purification plant and discharged to surface water (Fig. 3.1).

3.2.2.3 Definition of scenarios

For processing pig and dairy cattle manure, we compared the environmental impact of four scenarios relative to two reference situations. A reference for pig (PRef) as well as dairy cattle manure (CRef) was considered because their manure management systems differ considerably (Table 3.1). Furthermore, manure from fattening pigs was considered for the pig manure scenarios as this is the most common type of pig manure in the Netherlands.

The scenarios represented central processing plants. Scenario 1 implied processing of fattening pig or cattle manure into MC, solid fraction and permeate (PSc1 and CSc1), whereas scenario 2 also included AD of solid fraction to produce bio-energy (PSc2 and CSc2, Table 3.1).

Manure was stored for an average period of three months in the animal house in PRef and CRef. Additionally, in PRef, pig manure was stored for one month in a covered outside storage tank, which was excluded in PSc1 and PSc2 because manure processing reduced the need for storage space given that manure is collected from the farms on a monthly basis (De Vries et al., 2011). The end products were stored for an average period of three months in a covered circular concrete tank, except for the solid fraction, which was stored in an open shed. They were then applied to the field (Table 3.1). All emissions and resource use for the processes were included in the assessment.

3.2.3 Life cycle data inventory and assumptions

3.2.3.1 Chemical composition of manure and end products

The chemical composition of manure after storage (Table 3.2) was based on KWIN (2009-2010) and corrected for emissions from the storage system to obtain the composition after

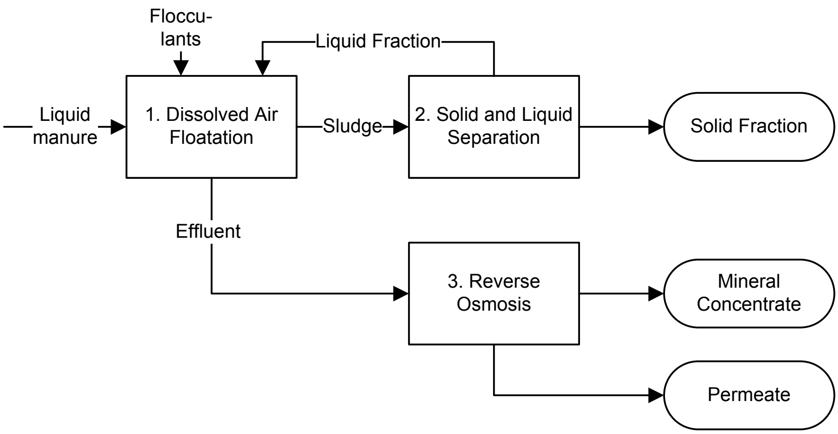

Fig. 3.2. Schematic overview of the manure processing technology and its intermediate and end products (after Hoeksma et al. (2011)).

Table 3.1. Considered processes in the pig and dairy cattle manure references and processing scenarios

Scenario	Storage in house	Outside storage	Manure processing	Anaerobic digestion	Product storage	Field application
Pig						
PRef	X	X	-	-	-	X
PSc1	X	-	X	-	X	X
PSc2	X	-	X	X	X	X
Cattle						
CRef	X	-	-	-	-	X
CSc1	X	-	X	-	X	X
CSc2	X	-	X	X	X	X

PRef = pig manure reference, PSc1 = pig manure scenario 1, PSc2 = pig manure scenario 2, CRef = dairy cattle manure reference, CSc1 = dairy cattle manure scenario 1, CSc2 = dairy cattle manure scenario 2. 'X' indicates included processes whereas '-' indicates excluded processes.

excretion following a mass balance approach. Distribution of mass and nutrients to the end products was based on data from the pilot plants (Table 3.2). Data used for pig manure were also used for cattle manure.

3.2.3.2 Storage of manure and end products

Emission of nitrogen occurred from manure and product storages as NH_3, nitrous oxide (N_2O), nitrogen monoxide (NO) and nitrogen gas (N_2) (Table 3.3). Emissions of NH_3 from processing and storage of end products was estimated as two times the emission from manure storage (total 4% N; 2% of N entering the processing plant and 2% of N entering storage). A higher emission was assumed as a result of more contact area with outside air during processing and storage. Emissions of N_2O, NO and N_2 from storage of MC were not included as they were considered to be negligible (Mosquera et al., 2010). Emission of N_2O from storage of solid fraction was based on solid manure storage (Groenestein et al., 2012). Leaching of NO_3^-, P and K during storage was assumed to be negligible because it is obligatory to have sealed concrete floors in manure and product storage systems in the Netherlands. Indirect emissions of N_2O were included as 1% of NH_3-N + NOx-N and 0.75% of NO_3-N after application (IPCC, 2006b).

Emission of methane (CH_4) occurred during storage of manure and of end products. Methane emission from manure storage prior to processing was modeled specific to the conditions of this study (De Mol & Hilhorst, 2003); the modeled data captured changes in emission related to changes in manure storage retention time between the references (3 months) and the scenarios (1 month) (De Vries et al., 2010). Methane emission from digestate storage was considered to be equal to outside storage of pig manure (Table 3.3); emissions during the storage of end products were based on Mosquera et al. (2010) and scaled relative to the ratio of emission from raw manure storage, and storage of solid (42 times lower) and liquid fractions (12 times lower).

Table 3.2. Calculated distribution of mass and nutrients for processing and chemical composition of manure and end products after storage

Product	Distribution of mass and nutrients[b]			Chemical composition						
	Mass (%)	OM (%)	N, N_{min}, P, K (%)	DM (kg ton^{-1})	OM (kg ton^{-1})	N_{tot} (kg ton^{-1})	N_{min} (kg ton^{-1})	P_2O_5 (kg ton^{-1})	K_2O (kg ton^{-1})	Density (kg m^{-3})
PM after storage[a]	-	-	-	90.0	60.0	7.60	4.60	4.2	7.2	1040
PM after in house storage	-	-	-	90.4	60.4	7.63	4.63	4.2	7.2	1040
PM after excretion	-	-	-	93.7	63.7	9.34	6.01	4.2	7.2	1040
Mineral concentrate	39	12	53, 70, 5, 79	27.1	18.1	9.90	7.77	0.5	14.7	1031
Solid fraction	19	88	45, 28, 95, 19	416	278	14.9	3.68	20.8	7.2	n.d.
Permeate	42	0	2, 3, 0, 1	0.17	0.11	0.32	0.27	0.0	0.2	1001
Digested solid fraction	-	-	-	351	213[c]	17.1	7.25	20.8	7.2	n.d.
CM after storage[a]	-	-	-	86.0	64.0	4.40	2.20	1.6	6.2	1005
CM after excretion	-	-	-	92.3	70.3	4.66	2.22	1.6	6.2	1005
Mineral concentrate				25.8	19.2	5.71	3.65	0.2	12.6	1031
Solid fraction		Equal to pig manure		395	294	8.58	3.03	7.9	6.2	n.d.
Permeate				0.16	0.12	0.18	0.07	0.0	0.2	1001
Digested solid fraction	-	-	-	352	251	9.89	3.36	7.9	6.2	n.d.

'-' = not applicable, n.d. = not determined. OM = organic matter, DM = dry matter, N_{tot} = total nitrogen, N_{min} = mineral nitrogen (NH_4^+-N), PM = pig manure, and CM = cattle manure.
[a] KWIN (2009-2010).
[b] De Vries et al. (2011); Hoeksma et al. (2011).
[c] Calculated based on 50% C in the organic matter, 37.5 m^3 CH_4 ton^{-1} solid fraction and a CH4 content of 60% in the biogas. Includes storage losses after digestion (0.17 kg CH_4 ton^{-1}).

Table 3.3. Emission factors and energy use during storage, processing, and field application of manure and end products and methane yields of anaerobically digested solid fractions

	Unit	Storage in house		Outside storage	Processing/ AD		Product storage			Field application							
		PM	CM	PM	PM	CM	MC	Dig	SF	PM, CM, Dig		AN		MC		SF	
										Gr	Ar	Gr	Ar	Gr	Ar	Gr	Ar
NH_3-N	% TAN	27[a]	10[a]	2% N[a]	4% N					19[i]	2[i]	2.5[i]		6.0[i]	0.64[i]	40[i]	22[i]
N_2O-N	% N		0.1[a]		-		-	0.1[b]	2[a]	0.4[j]	1.3[j]	1.3[j]	1[j]	0.6[j]	1.95[j]	0.4[j]	1.3[j]
N_2-N	% N		1[a]		-		-	1[c]	10[c]	-	-	-	-	-	-	-	-
NO-N	% N		0.1[a]		-		-	0.1[c]	2[c]	0.55[k]							
NO_3-N	% N	-	-	-	-		-	-	-	20.3[l]		15.8[l]		18.1[l]		22.6[l]	
NFRV[a]	%	-	-	-	-		-	-	-	62;45[m]		100[m]		60[m]	60;80[m]	31[m]	31;41[m]
$CH_{4\,long}$[a]	kg ton^{-1}	1.33[d]	3.32[d]	0.17[d]	-		0.014[e]	0.17[d]	0.004[e]	-	-	-	-	-	-	-	-
$CH_{4\,short}$[a]	kg ton^{-1}	0.29[d]	0.21[d]		-					-	-	-	-	-	-	-	-
$CH_{4\,yield}$	m^3 ton^{-1}	-	-	-	37.5[f]	25[f]	-	-	-	-	-	-	-	-	-	-	-
Energy	kWh ton^{-1}	1.7[g]		0.5[g]	9.0[h]		0.5[g]		-	Ecoinvent database[n]							

'-' = not included. AD = anaerobic digestion, PM = pig manure, CM = cattle manure, MC = mineral concentrate, Dig = digestate, SF = solid fraction, AN = ammonium nitrate, Gr = grassland, Ar = arable land, TAN = total ammoniacal nitrogen (NH_4^+ and NH_3), NFRV = nitrogen fertilizer replacement value, $CH_{4\,long}$ = methane emission factor for long term (3 months) storage, and $CH_{4\,short}$ = methane emission factor for short term (1 month) storage.

[a] Groenestein et al. (2012). [b] IPCC (2006a). [c] N_2-N and NO-N emission factors calculated as ratio of N_2O-N (Oenema et al., 2000). [d] De Mol & Hilhorst (2003). [e] Mosquera et al. (2010). [f] Solid fraction from fattening pig manure Timmerman et al. (2009). Solid fraction from dairy cattle manure (Van Dooren, 2010 Unpublished data). [g] Wesnæs et al. (2009). [h] Energy requirement for processing (De Vries et al., 2011). [i] Van Bruggen et al. (2011); Huijsmans & Hol (2010). [j] Velthof & Hummelink (2011); Velthof & Mosquera (2010). [k] Stehfest & Bouwman (2006). [l] Dekker et al. (2009). [m] De Vries et al. (2011); DR (2009). [n] EcoinventCentre (2007).

3.2.3.3 Manure processing

Separation of liquid manure and de-watering consumes electricity and chemicals for cleaning. Production emissions of these products were included in the assessment and taken from the ecoinvent database (EcoinventCentre, 2007). Electricity demand for processing was 9.0 kWh ton^{-1} manure entering the processing unit (Table 3.3). About 0.39 liters of flocculating additives (polyacrylamide) per ton of manure were used for separating solid particles from the liquid fraction. In addition, 0.022 liters of sodium hydroxide (NaOH) and 0.081 liters of sulfuric acid (H_2SO_4) per ton of manure was used for cleaning the installations (Hoeksma et al., 2011).

3.2.3.4 Anaerobic digestion

AD of the solid fraction was applied in PSc2 and CSc2. Digestion took place in a digester with a retention time of 60 days. The produced biogas was used in a combined heat and power plant (CHP) with an electric capacity of 250 kWh (Zwart et al., 2006). The energetic and electric efficiencies of the CHP were respectively 80 and 35% (Van der Leeden et al., 2003).

Emissions of CH_4, N_2O and NO_x and consumption of energy occurred during digestion and the combustion of the biogas. Methane losses were 1.5% of produced CH_4 (1% from the installation and 0.5% from the gas engine) (IPCC, 2006b). Emissions of N_2O were 0.1 kg N_2O TJ^{-1} of produced electricity and emissions of NO_x were 0.42 g NO_x m^{-3} of produced biogas (IPCC, 1997; VROM, 2010). Digestion required 66 MJ of electricity per ton substrate and 166 MJ heat per ton substrate (Berglund & Börjesson, 2006). Electricity was taken from the grid whereas heat originated from the CHP.

During AD the composition of solid fraction changed, as part of the organic nitrogen was converted into mineral nitrogen. To factor this in, we considered a 20% increase of N_{min} during AD (Ovinge, 2008; Schröder et al., 2008).

3.2.3.5 Distribution of products and transport distances

Distribution of manure (flows a, b, and c in Fig. 3.1) in the reference situations (PRef; CRef) was calculated based on Dutch national statistics, an average defined arable and dairy farm, and legal application standards of N and P_2O_5. Of the arable farms, 57% was on clay and 43% on sandy soil. For dairy farms these proportions were respectively 27% and 59% and additionally 14% was on peat soil. On the average arable farm, the total annual N, P_2O_5, and K_2O demand was: 179 kg N ha^{-1}, 85 kg P_2O_5 ha^{-1}, and 171 kg K_2O ha^{-1} (De Vries et al., 2011). On the dairy farm, total annual N, P_2O_5, and K_2O demand was: 274 kg N ha^{-1}, 97 kg P_2O_5 ha^{-1}, and 360 kg K_2O ha $^{-1}$. The legal application standards for animal manure applied on the arable farm were 170 kg N ha^{-1} and 85 kg P_2O_5 ha^{-1} and on the dairy farm 250 kg N ha^{-1} and 100 kg P_2O_5 ha^{-1} (combined grassland and arable land) (MEAAI, 2010). Application amounts of K_2O from animal manure were dependent on application limits of N and P_2O_5.

As a consequence of these limits, an average of 39% of fattening pig manure on province level was transported and applied to another province in the Netherlands (external application) (CBS, 2011; De Vries et al., 2011). Additionally, 2.7% of the surplus pig manure was exported outside the Netherlands and assumed to be applied in Northern France or Germany (Luesink, 2009 Personal communication). Exported manure was disinfected by heating it to 70 degrees C and consumed approximately 24 kWh electricity per ton of manure (Melse et al., 2004). Emissions of nitrogen during disinfection were not considered as they were expected to contribute very little to the end result.

On average 13.8% of the dairy cattle manure on farm level was transported and applied to an external arable farm (De Vries et al., 2011). No export of cattle manure outside the Netherlands was assumed as this occurs rarely. Manure applied on farm was distributed relatively to the ratio of nitrogen applied to grassland (86%) and arable land (14%).

In the scenarios it was assumed that the MC was authorized to be used over and above the application standards of nitrogen from animal manure, but not over the total nitrogen application standards, to represent its possibility of being used as mineral fertilizer. All MC, therefore, was applied in the local area. In the pig manure scenarios 56% was applied on grassland and 44% on arable land (De Hoop et al., 2011). Mineral concentrate was applied first in the local area where after solid fraction or digestate was applied until one of the application standards was reached. The remainder was transported off farm and if necessary outside of the Netherlands

Transport distances were based on data from the manure processing plants (DR, 2010 Unpublished data) and expert judgment (Table 3.4). Emission data and resource use for all transportation were taken from the ecoinvent database (EcoinventCentre, 2007). Distances for application outside the Netherlands were estimated distances to Northern France and Germany. Transport distance of chemicals used for processing was 150 km.

3.2.3.6 Manure product application and avoided fertilizer

Manure, MC, and digestate were applied with a manure injector on grassland and arable land. Solid fraction was applied by means of a solid manure spreader and incorporated into the soil directly after application (arable land). Mineral fertilizer was applied with a broadcast spreader. The environmental impact from production and combustion of diesel and capital goods for spreading of the products were taken from the ecoinvent database (EcoinventCentre, 2007). All application areas were assumed to have similar management.

During and after application of manure and end products emissions of NH_3, N_2O, NO and leaching of NO_3^- occurred (Table 3). Ammonia emission factors for the application of MC were adjusted relatively to emission factors for the application of manure. Absolute NH_3

Table 3.4. Transport distances and method of transportation in the references and scenarios

Scenario	Supply of manure (km)	Supply of mineral fertilizer (km)	Local transport (km)	External transport (km)	Outside NL transport (km)
PRef	-	50[b]	31[a]	120[a]	200[b]
PSc1&2	13.9[a]		31[a]	120[a]	200[b]
CRef	-		1[b]	50[b]	-
CSc1&2	13.9[a]		13.9[a]	50[b]	-
Transport method	Lorry >32 ton	Lorry 16 - 32 ton	Lorry >32 ton	Lorry >32 ton	Lorry 16 - 32 ton

'-' = not included.
[a] (DR, 2010 Unpublished data).
[b] Estimated transport distances. One km distance in CRef with tractor and trailer.

emissions for MC were recorded to be similar to manure (Huijsmans & Hol, 2010). Taking the higher mineral nitrogen content of MC into account, the emission factors of MC were calculated as 0.32 times the emission factor of manure (i.e., the ratio between the emission factor of liquid manure and MC).

Nitrous oxide emission factors for application of MC were adjusted in a similar way. Based on Velthof and Hummelink (2011), N_2O emission factors from MC were 1.5 times the emission factor of manure. All N_2O emission factors applied to grassland were weighted by soil type (i.e. the implementation of farms on different soils in section 2.3.5).

The nitrogen fertilizer replacement values (NFRVs, also called mineral fertilizer equivalent values) were used to calculate the avoided mineral N fertilizer from using manure products (Table 3). For cattle manure applied on farms, NFRV was 45%, as a consequence of grazing, and 60% in the case of off farm application (DR, 2009). These ratios were applied to adjust replacement values for MC and solid fraction. Replacement values for pig and dairy cattle manure applied on arable farms were weighted by soil type. Fertilizer replacement values for P_2O_5 and K_2O were considered as 100%. Furthermore, the NFRV for undigested solid fraction was also used for digested solid fraction, since it was indicated that the NFRV of digested manure increased in the first year after application but declined more rapidly afterwards and did not differ in the long term (Schröder et al., 2007).

In CSc1 and CSc2 over-application of K_2O on farm occurred (0.57 kg total), so was assumed not to substitute mineral fertilizer.

Nitrate leaching was computed as a percentage of the total N applied from each product. The leaching fractions for the products were based on N-balance calculations, i.e., after subtracting gaseous emission and N-uptake by crops (Dekker et al., 2009). Leaching after application of the MC was considered equal to leaching from liquid fraction after separation of liquid manure. The leaching from digestate was considered equal to leaching from undigested solid fraction.

3.2.4 Impact assessment

In the life cycle impact assessment, the emissions and resource use from the references and scenarios are accounted for and categorized into the environmental impact categories to which they contribute (Heijungs et al., 1992). Five impact categories were selected based on their relevance for manure management: climate change (CC expressed in kg CO_2-equivalants (eq.), including emission of CO_2, CH_4, and N_2O), terrestrial acidification (TA expressed in kg SO_2-eq., including emission of NH_3, NO_x, and SO_2), marine eutrophication (ME expressed in kg N-eq., including emission of NH_3, NOx, and leaching of NO_3^-), particulate matter formation (PMF expressed in kg PM_{10}-eq., including emission of particulates < 10 μm and NH_3, NO_x, and SO_2 as precursors of particulate matter), and fossil fuel depletion (FFD expressed in kg oil-eq., with 42 MJ kg oil-eq^{-1}). The scenarios and impact assessments were modeled and computed in SimaPro v.7.2 (PRé Consultants, the Netherlands) and by using the ReCiPe midpoint v.1.04 impact assessment method (Goedkoop et al., 2009).

3.2.5 Sensitivity analysis

A sensitivity analysis was conducted to assess the influence of changes in important parameters and underlying assumptions on the comparison between the scenarios and references and therewith the solidity of the end results. In the analyses, the effect of changing four parameters was tested: CH_4 emission from manure storage, NH_3 emission from manure processing, NFRV of MC, and excess heat utilization from AD.

3.3 RESULTS

3.3.1 Environmental consequences of processing manure to produce mineral fertilizer

The processing of pig manure and application of end products as fertilizer (PSc1) showed an increase in all environmental impact categories except for ME compared to the reference system. Climate change, FFD, TA, and PMF increased by respectively 9%, 33%, 19%, and 23% (Fig. 3.3). The increase in CC was caused mainly by emission of GHGs from storage of end products (Table 3.5). Although CH_4 emission from manure storage decreased, storage of solid fraction resulted in higher N_2O emissions from more denitrification compared to anaerobic storage of liquid manure (Table 3.6). Fossil fuel depletion increased as a result of energy demand for manure processing despite the energy demand for transportation having been approximately halved and avoided fossil fuel of mineral fertilizer produced. Less energy for transport was needed for two reasons. First, less weight had to be transported because water is removed during the process. Second, less long distance transport was required due to application of MC in the local area. TA and PMF increased due to NH_3 emission from manure processing together with NH_3 and NO_x emission from product storage. Storage of solid fraction resulted in higher NO_x emission. However, TA and PMF were governed by NH_3 emission from manure storage prior to processing, which was equal in all cases.

Processing dairy cattle manure and applying the end products (CSc1) showed a

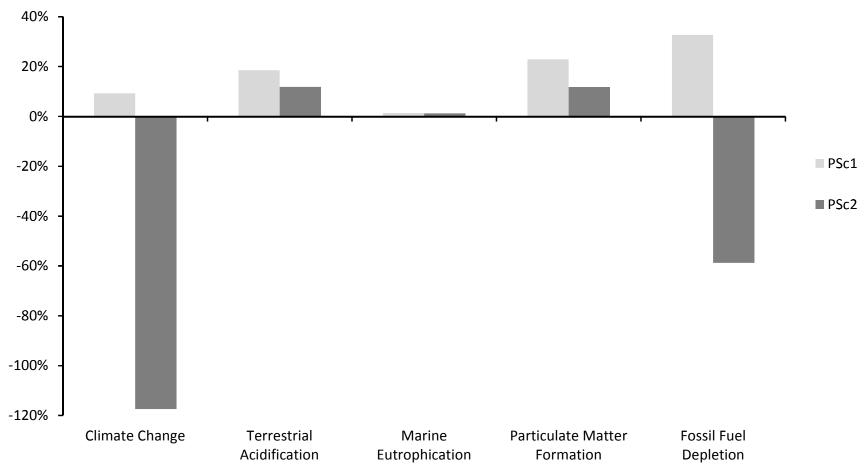

Fig. 3.3. Relative change in the environmental impact of the fattening pig manure scenarios (PSc1 & PSc2) compared to the reference (PRef = 0%)..

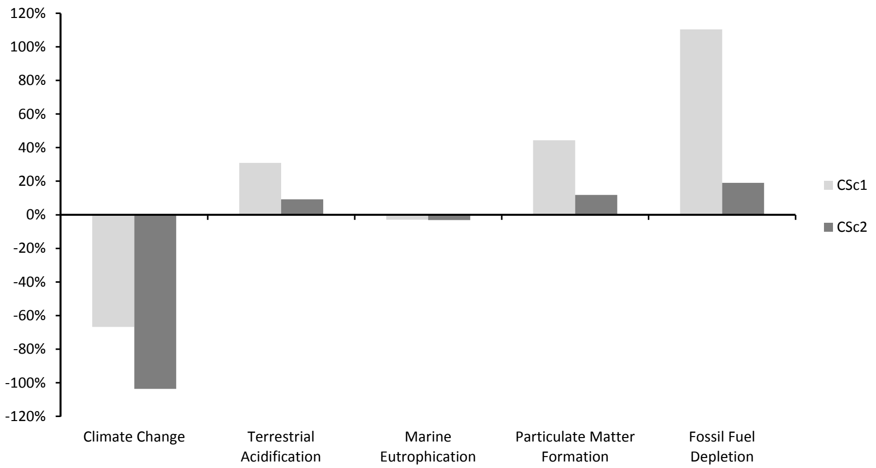

Fig. 3.4. Relative change in the environmental impact of the dairy cattle manure scenarios (CSc1 & CSc2) compared to the reference (CRef = 0%).

decrease in CC of 67% and an increase in FFD of 110%, in TA of 31%, and in PMF of 44% compared to the reference situation (Fig. 3.4). ME did not change. The decrease in CC was caused by less CH_4 emission from manure storage due to a shorter storage time, which was not offset by increased N_2O emission from storage of solid fraction. Fossil fuel depletion increased as a result of energy demand for manure processing and transportation of manure and end products. Energy for transportation increased, because products had to be transported to and from the processing location whereas in the reference situation only surplus cattle manure was transported locally. TA and PMF in CSc1 increased for the same reasons as in PSc1.

3.3.2 Environmental consequences of processing manure to produce bio-energy

The second scenario for pig and dairy cattle manure (PSc2 and CSc2) included the AD of solid fraction for bio-energy production. In PSc2, although TA and PMF increased due to higher NH_3 emissions from manure processing and product storage, other measures mainly decreased. CC reduced 117% and FFD 59% compared to the reference situation (Fig. 3.3). TA and PMF were lower compared to PSc1, as storage of solid fraction was avoided, but was higher (12%) than in the reference situation. Again, ME did not change. Climate change and FFD reduced mainly due to substitution of fossil electricity (85 MJ) as a result of bio-energy production. Furthermore, CC reduced as a result of less CH_4 emission from manure storage and less N_2O emission from storage of solid fraction as it was assumed to be digested shortly after production (Table 3.6). Fossil fuel depletion reduced not only because of substituted fossil electricity, but also because of less energy for transport compared to PRef. The produced energy more than counteracted the required energy for processing.

Table 3.5. Impact assessment results for the life cycle stages in the pig manure reference and scenarios

	Total*	Manure storage	Manure processing & product storage	Anaerobic digestion	Manure application	MC application	Solid fraction/ digestate application	Avoided fertilizer	Transport
Climate change (kg CO_2-eq)									
PRef	33.8	51.0	-	-	51.2	-	-	-75.7	7.2
PSc1	36.9	19.9	42.0	-	-	25.1	20.8	-74.8	3.8
PSc2	-5.9	19.9	12.4	-12.9	-	25.1	23.1	-77.2	3.8
Terrestrial acidification (kg SO_2-eq)									
PRef	5.0	5.3	-	-	0.31	-	-	-0.65	0.03
PSc1	5.9	4.8	0.99	-	-	0.36	0.33	-0.62	0.02
PSc2	5.6	4.8	0.91	-0.02	-	0.36	0.11	-0.64	0.02
Marine eutrophication (kg N-eq)									
PRef	0.93	0.21	-	-	1.5	-	-	-0.80	0.01
PSc1	0.95	0.19	0.10	-	-	0.73	0.67	-0.76	0.01
PSc2	0.95	0.19	0.05	-0.01	-	0.73	0.76	-0.79	0.01
Particulate matter formation (kg PM_{10}-eq)									
PRef	0.62	0.69	-	-	0.06	-	-	-0.15	0.01
PSc1	0.76	0.63	0.15	-	-	0.05	0.05	-0.14	0.01
PSc2	0.69	0.63	0.12	-0.01	-	0.05	0.02	-0.14	0.01
Fossil fuel depletion (kg oil-eq)									
PRef	-6.4	0.53	-	-	0.62	-	-	-10.3	2.8
PSc1	-4.3	0.48	3.4	-	-	0.18	0.21	-10.0	1.5
PSc2	-10.2	0.48	3.4	-5.5	-	0.18	0.09	-10.2	1.5

'-' = not included, *Totals do not always correspond to the sum of columns due to rounding

Table 3.6. Impact assessment results for the life cycle stages in the cattle manure reference and scenarios

	Total*	Manure storage	Manure processing & product storage	Anaerobic digestion	Manure application	MC application	Solid fraction/ digestate application	Avoided fertilizer	Transport
Climate change (kg CO_2-eq)									
CRef	69.0	87.0	-	-	19.0	-	-	-37.7	0.7
CSc1	22.9	9.2	27.9	-	-	10.5	6.9	-33.7	2.2
CSc2	-2.5	9.2	11.1	-7.7	-	10.5	7.0	-34.8	2.2
Terrestrial acidification (kg SO_2-eq)									
CRef	1.4	0.67	-	-	0.99	-	-	-0.29	0
CSc1	1.8	0.67	0.58	-	-	0.24	0.56	-0.27	0.01
CSc2	1.5	0.67	0.53	-0.01	-	0.24	0.33	-0.28	0.01
Marine eutrophication (kg N-eq)									
CRef	0.61	0.03	-	-	0.95	-	-	-0.37	0
CSc1	0.60	0.03	0.07	-	-	0.42	0.40	-0.33	0
CSc2	0.59	0.03	0.03	0	-	0.42	0.45	-0.34	0
Particulate matter formation (kg PM_{10}-eq)									
CRef	0.16	0.09	-	-	0.14	-	-	-0.07	0
CSc1	0.24	0.09	0.09	-	-	0.04	0.08	-0.06	0
CSc2	0.18	0.09	0.07	0	-	0.04	0.05	-0.06	0
Fossil fuel depletion (kg oil-eq)									
CRef	-3.9	0.48	-	-	0.46	-	-	-5.1	0.26
CSc1	0.4	0.48	3.4	-	-	0.18	0.21	-4.7	0.84
CSc2	-3.1	0.48	3.4	-3.3	-	0.18	0.08	-4.8	0.84

'-' = not included, *Totals do not always correspond to the sum of columns due to rounding.

Table 3.7. Results of the sensitivity analysis for the tested parameters

	Baseline	Tested parameters				
		CH_4 from manure storage	NH_3 manure processing	NFRV of MC		Heat use AD
				-20%	+20%	
Climate change (kg CO_2-eq)						
PRef	33.8	-	-	-	-	-
PSc1	36.9	62.9	36.6	48.2	25.6	-
PSc2	-5.9	20.1	-6.7	5.5	-17.2	-12.8
CRef	69.0	-	-	-	-	-
CSc1	22.9	101	22.8	29.7	16.2	-
CSc2	-2.5	75.3	-3.1	4.2	-9.3	-6.6
Terrestrial acidification (kg SO_2-eq)						
PRef	5.0	-	-	-	-	-
PSc1	5.9	-	5.5	6.0	5.8	-
PSc2	5.6	-	5.2	5.7	5.5	-
CRef	1.4	-	-	-	-	-
CSc1	1.8	-	1.6	1.8	1.7	-
CSc2	1.5	-	1.3	1.5	1.4	-
Marine eutrophication (kg N-eq)						
PRef	0.93	-	-	-	-	-
PSc1	0.95	-	0.94	1.08	0.81	-
PSc2	0.95	-	0.94	1.08	0.81	-
CRef	0.61	-	-	-	-	-
CSc1	0.60	-	0.60	0.67	0.52	-
CSc2	0.59	-	0.60	0.67	0.52	-
Particulate matter formation (kg PM_{10}-eq)						
PRef	0.62	-	-	-	-	-
PSc1	0.76	-	0.71	0.77	0.74	-
PSc2	0.69	-	0.64	0.71	0.67	-
CRef	0.16	-	-	-	-	-
CSc1	0.24	-	0.21	0.25	0.23	-
CSc2	0.18	-	0.16	0.19	0.17	-
Fossil fuel depletion (kg oil-eq)						
PRef	-6.4	-	-	-	-	-
PSc1	-4.3	-	-4.4	-3.3	-5.3	-
PSc2	-10.2	-	-10.3	-9.1	-11.2	-12.8
CRef	-3.9	-	-	-	-	-
CSc1	0.4	-	0.4	1.0	-0.2	-
CSc2	-3.1	-	-3.2	-2.6	-3.7	-4.7

'-' = no change.

Processing of dairy cattle manure and AD of solid fraction (CSc2) reduced CC by 104%, but increased FFD by 19%, TA by 9%, and PMF by 12% compared to the reference situation. As in CSc1, ME did not change. Climate change decreased as a result of less CH_4 emission from manure storage, less N_2O emission from storage of solid fraction, and because of the substitution of fossil based electricity (56 MJ). FFD increased as a result of low energy production and energy demand for processing and transportation (Table 3.6). TA and PMF increased for the same reason as in CSc1.

3.3.3 Sensitivity analysis

3.3.3.1 Methane emission from manure storage

In this study we modeled CH_4 emissions from manure storage specifically for the described circumstances. We assumed that storage time of manure was reduced to 1 month only in case of manure processing although in practical circumstances, even with manure processing, manure storage time might be longer. We therefore tested this assumption by exploring the effect of a 3 month storage time in the scenarios. Results showed an increase in CC for all scenarios (Table 3.7). In PSc1 and PSc2, CC was about 26 kg CO_2-eq higher than their baseline situations, whereas in CS1 and CSc2, this increase was about 78 kg CO_2-eq. For CSc1 and CSc2, this meant a change in the comparison between the scenarios and reference indicating the importance of shortening the storage time of manure to reduce CC. Furthermore, it shows the necessity of accurately estimating CH_4 emission from manure storages in LCAs.

3.3.3.2 Ammonia emission from manure processing

In this study, we applied an estimated NH_3 emission factor of 4% of N including both emission during storage of the end products (2%) and emissions during manure processing (2%) (Table 3.3). Data on NH_3 emissions during processing are scarcely available, and therefore over or under estimation may occur. Since we considered testing a higher emission irrelevant, (as this would increase TA and PMF and to lesser extent ME and CC), a lower emission rate during processing (0.3% of N in manure entering the processing plant) was tested (Melse & Verdoes, 2005). Results showed a decrease in TA and PMF of approximately 10% in CSc1, 13% in CSc2 and a decrease of 7% in both PSc1 and PSc2 (Table 3.7). The total impact for TA and PMF in PSc2 was approximately equal to the reference. The impact for CSc2 was even lower than its reference. This indicates that for improving the environmental performance of manure processing, controlling NH_3 emission during processing is essential.

3.3.3.3 NFRV of mineral concentrate

The NFRV of MC has been reported to vary considerably depending on factors such as, soil type, method of application and weather conditions (Velthof, 2009). To assess the influence of a change in the NFRV on the impact assessment, this parameter was varied plus and minus 20%. Results showed that mainly ME, CC and FFD decreased with a 20% increase of the NFRV and increased with a 20% decrease of the NFRV (approximate variation for ME was 14%, CC 29 - 265%, and FFD 10 - 147%).

3.3.3.4 Excess heat utilization from anaerobic digestion
The effect of including heat use of AD on the impact assessment was explored to represent existing initiatives of heat utilization. The substituted marginal source for heat in the Netherlands was assumed to be a mix of heat based on natural gas (79%) and heat based on light fuel oil (21%) (CBS, 2009; Menkveld & Beurskens, 2009). Heat from natural gas was divided into heat from boilers smaller than 100 kW with low NO_x emission technology (55%) and heat from industrial furnaces with low NO_x emission technology (24%) as these sources are the most common in the Netherlands (EcoinventCentre, 2007; Menkveld & Beurskens, 2009). Results showed a reduction in CC and FFD (respectively 118% - 160% and 26% - 50%) in the scenarios with AD (Table 3.7), thereby in CSc2, FFD was lower compared to the reference situation. This indicates that utilization of excess heat from AD strongly improves the environmental performance of manure management concerning CC and FFD.

3.4 DISCUSSION

Overall, processing pig and dairy cattle manure to produce mineral fertilizer increased the environmental impact. In environmental terms, processing without AD does not represent an attractive alternative to current agricultural practice, as it increases FFD, CC, TA, and PMF. In the pig manure scenarios, the additional energy required for processing outweighed the reduction in energy required for transportation. This has also been observed in other studies (Lopez-Ridaura et al., 2009). In the cattle manure scenarios, even additional energy for transportation was needed. This indicates that other drivers e.g., economic viability or social acceptance, are more likely to propel initiatives for manure processing instead of the related environmental impact as considered in this study.

The importance of controlling nitrogen emissions from manure processing and product storage (NH_3, NO_x, and N_2O) is stressed by the modeled increases in CC, TA, and PMF, as environmental impact is affected both directly and indirectly due to less substituted mineral fertilizer. The importance of nutrient recovery for mineral fertilizer substitution has been indicated in other studies as well (Prapaspongsa et al., 2010). Furthermore, as Dinuccio et al. (2008) have mentioned, storing separated fractions from mechanical separation of manure has the potential to increase CC. Emission data from storage of separated fractions are still rare. These emissions are difficult to quantify as they depend on specific circumstances such as, storage type, storage time and climatologic conditions. Our initial estimates, therefore, were based on a combination of comparative lab results and best available data. Our model results show that it is important to further quantify these emissions under different conditions and include them in environmental assessments of manure management techniques.

Processing pig manure and digestion of the solid fraction for bio-energy production presented a better alternative, as it added a strong environmental advantage by reducing CC and FFD. This is in agreement with other studies which showed a similar reduction in CC of approximately 40 kg CO_2-eq, including manure storage with a natural crust cover (Hamelin et al., 2011; Prapaspongsa et al., 2010). It also indicates that it is preferable to avoid producing end products with potentially high denitrification rates during storage as this results in increased CC. Furthermore, AD of the solid fraction from pig and cattle manure reduced TA and PMF compared to the scenarios without AD, as storage of solid fraction was avoided. However, TA and PMF remained higher compared to the references. This may be partly due to the assumption that NH_3 emission factors during application of digestate was assumed equal to undigested manure. The higher N_{min} in the digestate, therefore, increased total NH_3 emission. However, absolute emissions during application have been reported to be equal compared to undigested manure due to higher infiltration rates of digestate into the soil (Amon et al., 2006). In that case, including AD would lead to more reduction of TA and PMF improving its environmental potential compared to current practice.

Surprisingly, processing dairy cattle manure for bio-energy production did not lower FFD more than the reference. This indicates that processing of cattle manure in this fashion provides only little environmental benefit, reducing only CC. Moreover, the method presented in this research is costly with processing costs approximately 9 - 13 euro per ton of manure (De Hoop et al., 2011). Because cattle manure management differs strongly from pig manure management, simpler technologies requiring less energy may provide a better solution for handling cattle manure surpluses (Evers et al., 2010). Studies on the environmental consequences of such methods have not been conducted.

Compared to the references, in the scenarios ME did not change (maximum variation of 3%). The main reason was that emissions of NO_3^-, NO_x, and NH_3 counteracted each other in the different scenarios although they contribute in different degrees to ME, i.e., when emission of NO_3^- was lower, emissions of NH_3 and NO_x were higher and vice versa. This indicates that trade-off between different substances within an impact category may occur and require attention.

Important parameters affecting final results, as sensitivity analysis showed, include CH_4 emission from manure storage, NH_3 emission during processing, and the NFRV of the MC. Methane emission from manure storage has been reported elsewhere as an important parameter affecting the greenhouse gas balance and therewith CC from manure management systems (IPCC, 2006b; Lopez-Ridaura et al., 2009). Data on CH_4 emission from manure storages should, therefore, be carefully considered, and it is advised to use models, as in this study, based on a higher Tier method in the IPCC guidelines to obtain specific data related to the circumstances of the conducted study. NH_3 emission directly affects TA and PMF and to a lesser extent ME and CC and should therefore be kept to a minimum. This could be achieved by for example reducing contact with outside air, to ensure lower NH_3 emission and overall equal or better performance compared to current practice. Moreover, it shows the necessity of obtaining more detailed data on NH_3 emissions, as well as other N-substances, such as N_2O, NO and N_2, occurring during processing to enhance LCA studies of manure management as very often emissions from processing may be underestimated. The NFRV of MC mainly affected ME, CC and FFD. It will depend on circumstances, such as soil type, weather conditions, cropping system, and time of application of the manure product. NFRVs used in calculating mineral fertilizer substitution rates should, therefore, be tailored to the specific conditions applicable over the long term. As Schröder et al. (2005) states, a correct assessment of the NFRV for each manure product is important in reducing the environmental impact of manure management in terms of NO_3^- leaching.

Marginal production of electricity was not addressed in the sensitivity analysis as recent studies have shown that a change in marginal electricity will not affect the conclusion of the study (De Vries et al., 2011; Hamelin et al., 2011).

This study included CH_4, N_2O and NH_3 emissions from in house storage of manure as these emissions contribute strongly to CC, TA, and PMF. It also indicates that future work should consider the loss of N from manure storage prior to processing to determine a proper mineral fertilizer substitution rate. Additionally, although studies on reducing emissions from animal houses have been done e.g., (Aarnink et al., 1996; Canh et al., 1998; Monteny et al., 2006), new developments are needed, such as separating feces and urine under the slats (Aarnink & Ogink, 2007), and should be assessed to indicate improvements of the environmental performance of manure management.

As N and P application standards in the Netherlands will be lowered in the coming years to comply with the EU Nitrates Directive (MEAAI, 2010), local manure surpluses will likely increase, inducing more transportation of manure and its derived products. This, however, should not affect the conclusions of this study, as changing distribution and transport distances has only a limited effect on the environmental impact of manure management. Moreover, mineral fertilizer replacement rates may also change due to lowering of the application standards. The reference system, however, will also change in conjunction with those standards and, therefore, conclusions of this study will not change (i.e., the comparison between scenarios and references will stay the same). On the other hand, availability of other fertilization products could change fertilization strategies on farms and therewith the environmental impact. This should be studied in more detail as it was out of scope in this study.

Finally, future processing scenarios are also expected to include the processing of digestate from AD. Currently, however, this approach has practical difficulties as digestates vary in composition as a result of varying input materials and because processing conditions change from plant to plant (Hoeksma et al., 2011). It is expected that AD of liquid manure will increase energy production as compared to AD of the solid fraction and therewith further reduce CC and FFD (De Vries et al., 2011).

3.5 CONCLUSION

Processing of fattening pig and dairy cattle manure by using liquid and solid separation and reverse osmosis to produce mineral fertilizer increased overall environmental impact in terms of climate change (CC) (except for dairy cattle manure), terrestrial acidification (TA), particulate matter formation (PMF), and fossil fuel depletion (FFD) compared to current agricultural practice. Marine eutrophication (ME) did not change. Adding the production of bio-energy enhanced the environmental performance by substituting fossil electricity and reducing storage emissions from solid fraction. Utilization of excess heat increased this trend for CC and FFD. However, the addition of AD did not present a better option compared to current practice concerning TA and PMF, and FFD for cattle manure, unless when NH_3 emissions from processing were kept low. In that case, equal or better environmental performance was obtained for TA and PMF.

Key parameters affecting the environmental performance were identified as NH_3 emission from manure processing and product storage together with N_2O and NO_x emissions from product storage as a result of denitrification; controlling these was essential to reduce the environmental impact of manure processing and to improve the potential for substituting mineral fertilizer. Additionally, CH_4 emission from manure storage should be modeled as precisely as possible to the circumstances being studied, to correctly assess its environmental consequences. Overall, this emphasizes a continuous need of real time measurements of these emissions to 'feed' future LCA studies.

Results of this study show the environmental consequences and key parameters affecting the environmental impact of manure management as it considers the full life cycle of the processing and application of all end products. It also shows that innovations that appear worthwhile for reducing environmental impact do not always deliver the expected results when considering all consequences within the system. Furthermore, it highlights the importance of particular emissions during both processing and storage. For those tackling environmental problems around manure management, this assessment has provided a number of key outcomes to inform their decision making.

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Chapter 4

// LIFE CYCLE ASSESSMENT OF SEGREGATING FATTENING PIG URINE AND FECES COMPARED TO CONVENTIONAL LIQUID MANURE MANAGEMENT

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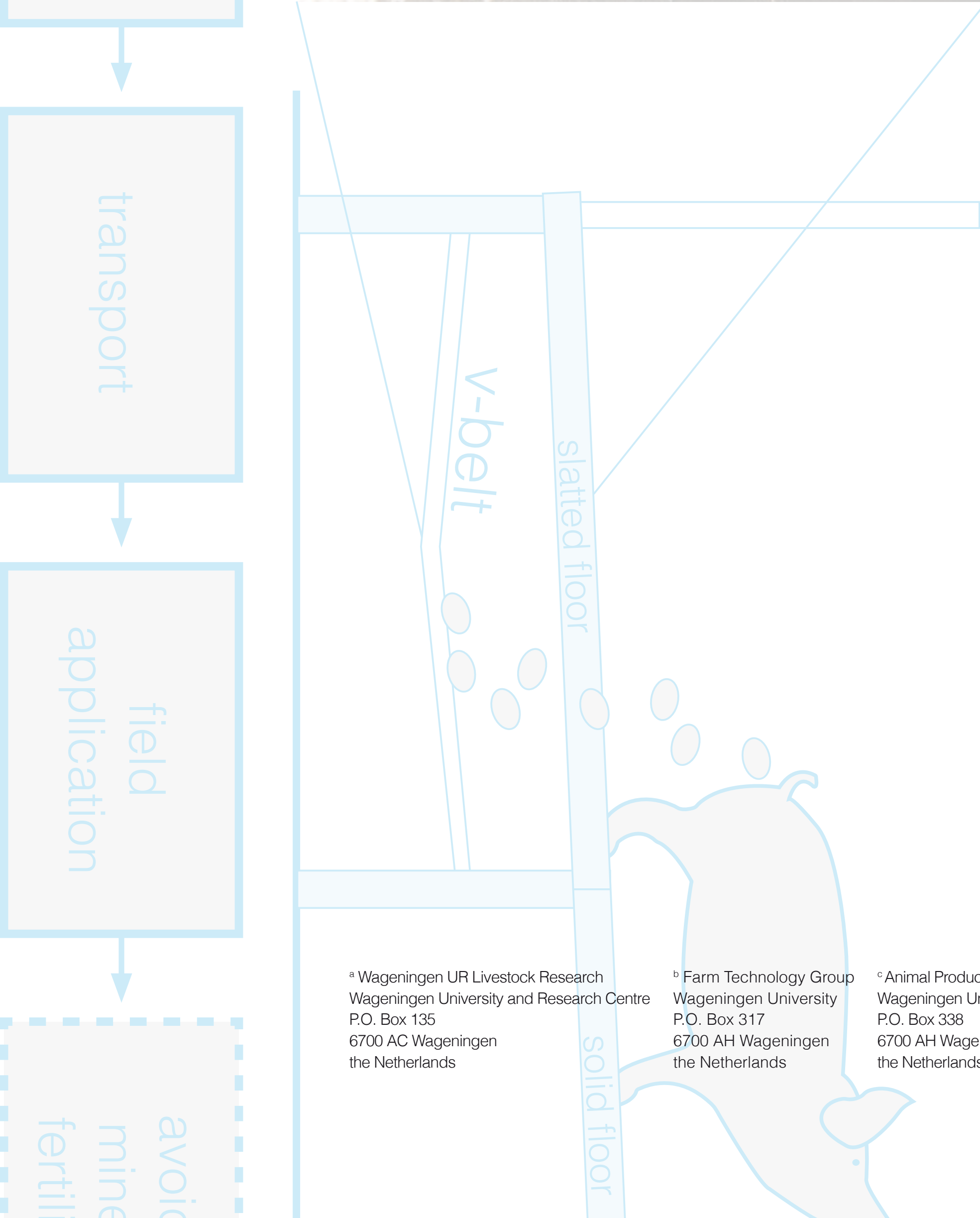

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ABSTRACT

Gaseous emissions from in-house storage of liquid animal manure remain a major contributor to the environmental impact of manure management. Our aim was to assess the life cycle environmental consequences and reduction potential of segregating fattening pig urine and feces with an innovative V-belt system and to compare it to conventional liquid manure management, i.e. the reference. Moreover, we aimed at analyzing uncertainty of the outcomes related to applied emission factors. We compared a reference with two scenarios: segregation with solid, aerobically, stored feces and with liquid, anaerobically, stored feces. Results showed that, compared to the reference, segregation reduced climate change (CC) up to 82%, due to lower methane emission, reduced terrestrial acidification (TA) and particulate matter formation (PMF) up to 49%, through lower ammonia emission, but increased marine eutrophication (ME) up to 11% through nitrogen oxide emission from storage and nitrate leaching after field application. Fossil fuel depletion did not change. Segregation with liquid feces revealed lower environmental impact than segregation with solid feces. Uncertainty analysis supported the conclusion that segregating fattening pig urine and feces significantly reduced CC, and additionally, segregation with liquid feces significantly reduced TA and PMF compared to the reference.

4.1 INTRODUCTION

Gaseous emissions of methane (CH_4), and nitrogen, in the form of ammonia (NH_3), nitrous oxide (N_2O), and nitrogen oxides (NO_x) from in-house manure management remain major contributors to the environmental impact in the life cycle of liquid animal manure (Chadwick et al., 2011; De Vries et al., 2012a, Chapter 3). In light of a growing worldwide consumption of animal products, including pig meat, these gaseous emissions will continue to contribute to environmental problems, such as climate change, terrestrial acidification, marine eutrophication, and health issues resulting from particulate matter formation (Steinfeld et al., 2006). Additionally, emission of odor may result in nuisance leading to negative associations with the livestock sector. Emissions of nitrogenous gases not only affect the environment directly, but also reduce the fertilizing capacity of manure or its derived end products. This reduced fertilizing capacity decreases the potential to supply nutrients to crop production and substitute mineral fertilizer (Lopez-Ridaura et al., 2009).

The amount of gaseous emissions, such as greenhouse gases (GHGs; CH_4, N_2O, and carbon dioxide (CO_2)) and acidifying gases (NH_3 and NO_x), are regulated by respectively the Kyoto Protocol, and in the European Union by national emission ceilings (NEC) (EU, 2001). For the Netherlands, it was stated in the NEC of 2001 that NH_3 emissions should have been reduced to 128 kilotons annually by the year 2010, of which agriculture contributes 96 kilotons (NL-Agency, 2012). Although this goal was achieved, the directive is being updated with new standards to be achieved by 2020 (EC, 2012).

Manure processing technologies developed for improving manure management and reducing gaseous emissions have focused mainly on secondary separation and processing, i.e. dealing with manure as a mixture of urine and feces (Melse & Verdoes, 2005). The environmental impacts of processing technologies were assessed by life cycle assessment (LCA) (e.g. see De Vries et al. (2012a) (Chapter 3) and Prapaspongsa et al. (2010)) considering all environmental consequences from a change in management. Manure processing technologies keeping urine and feces separated (or segregated) directly after excretion, however, have received little attention so far.

Technologies that segregate urine and feces from (fattening) pigs by using a belt system, e.g. the Hercules and Kempfarm® system, or filter nets (Ogink et al., 2000; Van Kempen et al., 2003) have shown to reduce gaseous emissions, including NH_3 emissions up to 75%, CH_4 emissions up to 80%, and odor emission up to 74% as compared to conventional manure management systems (Aarnink et al., 2007; Lachance et al., 2005). The environmental impact of such technologies, however, was not considered from a life cycle perspective. Furthermore, uncertainty in assumptions and variation in emission factors strongly influence results of LCAs, and should be addressed in such environmental assessments (Huijbregts, 1998).

In addition to the environmental impact, segregating urine and feces can easily be combined with use of bedding material, such as straw to enhance animal welfare (Tuyttens, 2005). Such additions are generally problematic in conventional liquid manure systems. Moreover, economic costs were reported to be similar to conventional systems (Aarnink et al., 2007) .

In this paper, we aim to assess the life cycle environmental consequences and reduction potential of segregating fattening pig urine and feces by an innovative V-belt system (Kempfarm®). We indicate critical management factors affecting the environmental performance of manure management and shifting of N emissions within and between environmental compartments and life cycle stages. Finally, we aim to analyze the uncertainty in the results related to variation in emission factors.

4.2 MATERIALS AND METHODS

4.2.1 LCA approach and impact assessment

We used LCA to compare the environmental impact of segregating pig urine and feces with the impact of conventional manure management, i.e. liquid manure storage, transport, and field application. Environmental consequences from changing manure management were included in the system boundary (i.e. consequential LCA) (Weidema et al., 2009). This implied that for electricity supply and mineral fertilizer production we used so called marginal suppliers. Marginal suppliers were based on De Vries et al. (2012a) (Chapter 3). The geographical scope of the study was Western Europe, where the Netherlands was considered for marginal suppliers, compositions of manure, urine and feces, and NO_3^- leaching.

We evaluated five environmental impact categories most relevant to manure management: climate change (CC), terrestrial acidification (TA), marine eutrophication (ME), particulate matter formation (PMF) and fossil fuel depletion (FFD) (De Vries et al., 2012a, Chapter 3; Prapaspongsa et al., 2010). For PMF, we included only NH_3, NO_x, and SO_2 as precursors. Direct emission of PM_{10} was excluded, because this was not affected by the different management systems (Aarnink et al., 2007). For eutrophication, we excluded phosphorus (P) leaching as in all systems an equal amount of P entered and left the system. Field application was considered to be subject to equal management leading to unchanged leaching of P (De Vries et al., 2012a, Chapter 3). To compare the environmental impacts, we used the ReCiPe midpoint v.1.04 impact assessment method (Goedkoop et al., 2009) in SimaPro v.7.3 (PRé Consultants, the Netherlands).

4.2.2 Functional unit and manure management scenarios

4.2.2.1 Functional unit

The function of the considered system is to manage liquid manure or the segregated urine and feces. As the starting point is the excretion of urine and feces by the animal, we applied a functional unit (FU) of 1 ton mixed (i.e. liquid manure) or segregated fattening pig urine and feces originating from the same excreta to be able to compare scenarios. The same excreta ensured that equal amounts of nutrients and dry matter entered the system.

4.2.2.2 Manure management scenarios

The reference (Ref) was based on conventional liquid manure management, including three months in-house storage under a slatted floor (60% of pen area), one month outside storage

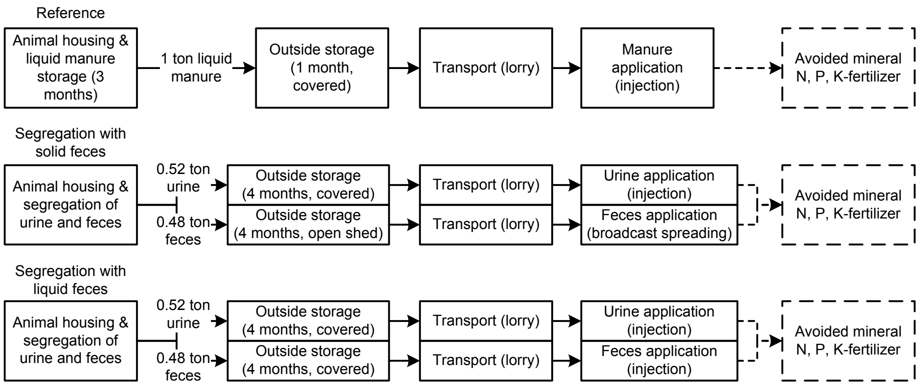

Fig. 4.1. Schematic representation of the scenarios for manure management: the reference with liquid manure and the alternative scenarios with segregation of urine and feces. ‘Solid’ and ‘liquid’ feces indicate different storage conditions, i.e. respectively aerobic and anaerobic depending on use of bedding material. The dotted lines and boxes represent avoided production of mineral fertilizer.

in a covered concrete tank, transport, and field application (Fig. 4.1) (De Vries et al., 2012a, Chapter 3). The storage times represent yearly averages, as farmers are only allowed to apply manure from February 16 until the 1st of September in the Netherlands (DR, 2012). We assumed that manure was transported 31 km from the farm to the field by a lorry with a capacity of 32 tons (De Vries et al., 2012a, Chapter 3; EcoinventCentre, 2007). Subsequently, manure was injected into the soil of arable land with a conventional crop rotation, including potatoes, sugar beets, wheat, and onion, using a manure injector (De Vries et al., 2011). Manure application was assumed to substitute mineral N, P, and K-fertilizer by respectively 62%, 100%, and 100% (De Vries et al., 2012a, Chapter 3; Wulf et al., 2006).

Scenarios with segregation of urine and feces included in-house segregation, four months outside storage of urine and feces separately (i.e. a yearly average), transport, and field application (Fig. 4.1). Segregation was obtained by a V-shaped belt under a concrete slatted floor (see abstract art). Urine flows down the belt constantly, firstly to the middle of the belt, and secondly to the end of the belt into a collection pit (i.e. by gravitational force). Feces were removed two times per day by rotating the belt. Outside storage of urine occurred in a covered concrete storage tank. Feces storage was presumed to be aerobic (as solid feces) or anaerobic (as liquid feces), depending on use of bedding material. The use of bedding material will increase aeration in the storage of solid feces, leading to higher (de-)nitrification and N_2O emission (Chadwick et al., 2011). Currently, we do not know exactly at what feces composition (mainly dry matter content) the feces behave as ‘solid’ or when feces behave as ‘liquid’. Therefore, we assumed that solid and liquid feces have the same composition, but can behave as different types. In one scenario it was considered that solid feces were stored aerobically in an open shed, whereas in a second scenario liquid feces were stored anaerobically in a covered concrete tank. We assumed transport distances equal to Ref. Subsequently, urine was applied on the field with a manure injector. Solid feces were applied with a solid manure spreader and were directly incorporated into the soil after application, whereas liquid feces were assumed to be injected. Urine and feces were applied to the same arable crop rotation as in Ref. Environmental impact related to the production and use of bedding materials were excluded.

Table 4.1. Chemical composition and excretion rates of liquid fattening pig manure, urine and feces in their respective life cycle stage

Product & life cycle stage	DM (kg ton^{-1})	OM (kg ton^{-1})	N_{tot} (kg ton^{-1})	N_{min} (kg ton^{-1})	P_2O_5 (kg ton^{-1})	K_2O (kg ton^{-1})	Density (kg m^{-3})	pH (-)	Excretion rate (ton ap^{-1} yr^{-1})
PM after excretion	111[a]	80.9[a]	10.6[b]	7.20[b]	4.25[b]	6.75[b]	1040[c]	7.5[c]	1.2[b]
PM after in-house storage	90.2[a]	60.2[a]	8.82[a]	5.77[a]	4.25	6.75			
PM after outside storage	90.0[a]	60.0[a]	8.64[a]	5.59[a]	4.25	6.75			
Urine after excretion	18.5[e]	6.2[e]	8.22[e]	8.16[e]	0.71[e]	5.95[e]	1010[d]	8.9[d]	0.54[d]
Urine after segregation	18.5	6.2	7.39[a]	7.33[a]	0.71	5.95			
Urine after storage	18.5	6.2	7.24[a]	7.18[a]	0.71	5.95			
Feces after excretion	210[e]	161[e]	13.1[e]	6.16[e]	8.06[e]	7.61[e]	1120[d]	6.5[d]	0.50[d]
Feces after segregation	210	161	12.3[a]	5.35[a]	8.06	7.61			
Solid feces after storage*	210	161	10.3[a]	3.38[a]	8.06	7.61			
Liquid feces after storage*	210	161	11.9[a]	4.95[a]	8.06	7.61			

PM = pig manure, DM = dry matter, OM = organic matter, N_{tot} = total nitrogen, N_{min} = mineral nitrogen (NH_4^+-N), ap = animal place. * Solid and liquid feces indicate the storage conditions, i.e. respectively aerobic and anaerobic. [a] Corrected for emissions from storage according to Table 4.2. [b] (CBS, 2010; KWIN, 2009-2010) ; [c] (KWIN, 2009-2010); [d] (Aarnink et al., 2007); [e] Calculated according to formulas 1 - 4 in SI.

Table 4.2. Applied emission factors, energy consumption and fertilizer replacement values in the respective life cycle stages of the reference and scenarios with segregation. Min. and max. values between brackets represent 95% confidence intervals

Life cycle stage		NH_3-N (kg ton^{-1})	N_2O-N (g ton^{-1})	N_2-N (g ton^{-1})	NO-N (g ton^{-1})	CH_4 (kg ton^{-1})	Odor (OU s^{-1} ton^{-1})	NO_3-N (% N)	NFRV (%)	Energy (kWh ton^{-1})
In-house	Ref	1.72[a] (1.65-2.40)[b]	4.24[c] (0.77-7.7)[d]	42.4[e]	4.24[e]	13.1[c] (6.54-19.6)[j]	19.2[a] (14.4-24)[b]	-	-	1.7[g]
	Seg	0.83[f] (0.79-2.1)[d,h]	67.4[f] (12.3-123)[d]	1.06[e]	0.11[e]	0.91[f] (0.45-1.36)[d]	5.68[f] (4.43-7.11)[b]	-	-	0.48[f]
Outside storage	Ref	2% N[i] (± 50%)[j]	-	-	-	0.17[k]	n.d.	-	-	1.2[g]
	Urine	2% N[i] (± 50%)[j]	-	-	-	0.014[k]	n.d.	-	-	1.2[g]
	Feces solid	2% N[i] (± 25%)[j]	2% N[i,j]	10% N[e]	2% N[e]	0.004[k]	n.d.	-	-	-
	Feces liquid	2% N[i] (± 50%)[j]	0.1% N[i,j]	1% N[e]	0.1% N[e]	0.17[k]	n.d.	-	-	1.2[g]
Field application	Ref	2% TAN[l] (± 60%)[j]	1.3% N[m,j]	-	0.55% N[n]	-	n.d.	17.4[o]	62[p]	Ecoinvent[q]
	Urine	0.64% TAN[l] (± 60%)[j]	1.95% N[m,j]	-	0.55% N[n]	-	n.d.	14.3[o]	85[p]	
	Feces solid	22% TAN[l] (± 60%)[j]	1.3% N[m,j]	-	0.55% N[n]	-	n.d.	19.2[o]	41[p]	
	Feces liquid	2% TAN[l] (± 60%)[j]	1.3% N[m,j]	-	0.55% N[n]	-	n.d.	19.5[o]	41[p]	
	Mineral Fertilizer	2.5% TAN[l] (± 20%)[j]	1% N[m,j]	-	0.55% N[n]	-	n.d.	14.4[o]	100[p]	

Ref = reference, Seg = segregation, OU = odor units, NFRV = nitrogen fertilizer replacement value, TAN = total ammoniacal nitrogen, n.d. = no data, '-' = not included. [a] (VROM, 2012); [b] Min based on VROM (2012), Max from Mosquera et al. (2010a); [c] (Mosquera & Hol, 2012); [d] Min and max from Aarnink et al. (2007); [e] (Oenema et al., 2000) (ratio N_2O:N_2:NO = 1:10:1 for liquid manure and 1:5:1 for solid feces); [f] (Aarnink et al., 2007); [g] (Wesnæs et al., 2009); [h] Max based on Van Kempen et al. (2003); [i] Dutch national statistics based on Groenestein et al. (2012); [j] Uncertainty (N_2O-N; -50%, +100% and CH_4 ± 50%) (IPCC, 2006a; IPCC, 2006b); [k] Modeled values based on De Mol & Hilhorst (2003) and De Vries et al. (2012a); [l] Dutch national statistics (Huijsmans & Hol, 2010; Huijsmans et al., 2007; Van Bruggen et al., 2011); [m] Dutch national statistics, representing weighted average of clay (57%) and sandy (43%) soils (De Vries et al., 2012a; Velthof & Hummelink, 2011; Velthof & Mosquera, 2010);[n] (Stehfest & Bouwman, 2006); [o] Calculation based on Dekker et al. (2009); [p] Representing weighted averages to clay and sandy soils (DR, 2009; Versluis et al., 2005); [q] (EcoinventCentre, 2007).

4.2.3 Data inventory and assumptions

4.2.3.1 Product chemical composition

Excretion rate and chemical composition of liquid manure in Ref was based on average data from the Dutch fattening pig sector (Table 4.1). Excretion rates and the composition of urine and feces were based on an eight month measurement cycle, consisting of two growing periods, of an experimental set-up of the Kempfarm® system (Aarnink et al., 2007). Because the compositions of liquid manure, urine, and feces were taken from different literature sources and equal amounts of nutrients and dry matter had to enter the systems in the Ref and scenarios for comparability reasons (FU), we scaled the compositions of urine and feces to those of liquid manure in Ref. This is illustrated in the supporting information (SI) for the total N composition. For Ref and scenarios, we calculated a mass balance to determine the compositions of the liquid manure and urine and feces in every life cycle stage (Table 4.1 and Table S1).

4.2.3.2 Process emissions

During storage, segregation, and field application NH_3, N_2O, N_2, NO_x, CH_4 emit as well as odor and nitrate (NO_3^-) leaches (Table 4.2). Emission factors for in-house storage and segregation were based on respectively Dutch national data and Aarnink et al. (2007). Emission factors for the Kempfarm® system were measured over eight months (two growing periods) in a small-scale set-up at a commercial farm. Ammonia concentrations were measured continuously, whereas greenhouse gases were measured four times (during 24 hours for each sample) within the experiment (Aarnink et al., 2007). Emission factors for N_2O from storage of solid feces were presumed the same as for solid manure storage (i.e. aerobic) and emission factors for storage of liquid feces were presumed the same as liquid manure storage (i.e. anaerobic) (Groenestein et al., 2012). CH_4 emission factors were based on various references (De Mol & Hilhorst, 2003; De Vries et al., 2012a, Chapter 3; Mosquera et al., 2010b). Methane emissions during storage of urine and feces were scaled relative to the ratio of emission from liquid manure in Ref (a factor of 42 times lower for solid feces and 12 times lower for urine) based on a laboratory experiment (Mosquera et al., 2010b). In this study liquid manure and separated solid and liquid fractions were stored for two months at 14 °C. Methane emissions were measured 20 times during the study (Mosquera et al., 2010b).

Based on De Vries et al. (2012a) (Chapter 3), we assumed emissions from application of urine to be similar to those of de-watered liquid fraction (mineral concentrate), whereas emissions from application of feces were assumed similar to those of the solid fraction remaining after liquid manure separation (Table 4.2). We included indirect N_2O emissions from NH_3 and NO_x emission (IPCC, 2006b).

NO_3^- leaches from N application to the soil and can be calculated as a leached fraction of the soil N-surplus, i.e. the difference between N-input into the soil and N-uptake by crops (Dekker et al., 2009; Schröder, 2005). N-input into soil and N-uptake by the crops depend, among others, on the chemical composition of fertilizer products (i.e. N_{min} and N_{org}) and related gaseous emissions during application (NH_3 and N_2O). We included these factors by adjusting the NO_3^- leaching factor of each fertilizer product for their respective composition and gaseous emissions (Table 4.1 and 2). Furthermore, we used a leaching fraction of 0.43

and a 95% confidence interval of 0.36 - 0.51 (Schröder et al., 2011). This value represents a weighted average of leaching fractions on arable land with sandy (leaching fraction of 0.56) and clay (leaching fraction of 0.34) soils in the Netherlands.

4.2.3.3 Background emissions
Background emissions are emissions not directly occurring from the management system, but from supporting systems, i.e. electricity production, transportation, mineral fertilizer production and capital goods production. All environmental impact data related to background systems were taken from the Ecoinvent database (EcoinventCentre, 2007).

4.2.4 Uncertainty analysis
Uncertainty analysis was done to achieve insight in how variation in emission factors, i.e. their 95% confidence intervals (95% CI), affected final results and the comparison of scenarios. We used Monte Carlo simulation (1000 runs) for the separate scenarios and for comparisons of scenarios to assess the probability of a higher and lower outcome of the respective comparison. Differences were assumed significant when outcomes were higher/ lower in >97.5% of the comparisons (i.e. 2-sided test at $P < 0.05$).

The following emission factors and related impact categories were included in the uncertainty analysis: N_2O, NH_3, CH_4, and NO_3^- (CC, TA, PMF, and ME). For assessed emission factors, we defined a probability density function (PDF) and the 95% CI. PDFs and CIs were based on literature data (Table 4.2). Overall, PDFs were assumed as normal distributions (i.e. Gaussian), whereas for N_2O emission we applied a lognormal distribution (Payraudeau et al., 2007).

Ideally, within Monte Carlo simulation a correlation matrix for dependent factors is established (e.g. when NH_3 emission from application is low, N_2O emission may be higher). However, due to the use of different literature sources to determine the 95% CI, we could not establish such correlations and assumed independence of factors (Björklund, 2002; Payraudeau et al., 2007). Where possible, however, dependency was included via calculation. NO and N_2 emission from storage were calculated according to their ratio related to N_2O emission, and therefore are interrelated (Table 4.2). Similarly, NO_3^- leaching was calculated depending on gaseous emissions (N_2O and NH_3) (section 4.2.3.2). Other factors affecting NO_3^- leaching, such as uncertainty in N-uptake by crops were excluded. Overall, the dependent factors will change according to the randomly selected values of the emission factors during Monte Carlo simulation. The assumed independency between emission factors, however, will most likely lead to an overestimated uncertainty in the final results.

4.3 RESULTS AND DISCUSSION

4.3.1 Impact of segregation with solid feces
Segregation with solid feces reduced CC with 66% compared to Ref (Table 4.3). This reduction was due to lower CH_4 production from methanogenic bacteria during in-house segregation, as compared to storage of liquid manure in Ref (328 vs. 23.5 kg CO_2-eq).

Storage of solid feces, however, resulted in higher N_2O emission from (de-)nitrification (56.5 kg CO_2-eq) than outside storage of liquid manure in Ref (0.83 kg CO_2-eq). Moreover, application of urine and feces resulted in higher total N_2O emission (107 kg CO_2-eq) than application of liquid manure in Ref (total 10.5 kg CO_2-eq). Higher N_2O emissions from storage and application in the segregation scenario, however, did not counteract the reduction in CH_4 emission. The segregation scenario with solid feces, therefore, resulted in a net reduction of GHGs.

Segregation with solid feces also reduced TA with 29% and PMF with 22% compared to Ref (Table 4.3). This was mainly due to lower NH_3 emission from in-house segregation compared to conventional manure storage (2.48 vs. 5.11 kg SO_2-eq). Reduced in-house NH_3 emission counteracted increased NO_x emission (0.14 kg SO_2-eq) from (de-)nitrification during storage of feces and increased NH_3 emissions during application.

Segregation increased ME with 11% compared to Ref (Table 4.3), because of increased NO_x emission during storage of solid feces. FFD was hardly affected (difference of 0.36 kg oil-eq), because it was mainly related to the total replaced mineral fertilizer and to a lesser extent to transport, which were both similar in the scenario with segregation and Ref.

4.3.2 Impact of segregation with liquid feces

Segregation with liquid feces reduced CC with 82% compared to Ref (Table 4.3). This reduction was due to the same causes as in the scenario with segregation and solid feces. Storage of liquid feces, however, resulted in lower N_2O emission compared to solid feces (3.35 vs. 56.5 kg CO_2-eq) due to anaerobic conditions, but at the same time only slightly increased CH_4 emission compared to storage of solid feces (0.05 vs. 2.04 kg CO_2-eq) (Table 4.2 and Table 4.3).

Segregation with liquid feces reduced both TA and PMF with 49% compared to Ref. Causes were again similar to the scenario with segregation and solid feces. Additionally, liquid feces resulted in lower NO_x emission from (de-)nitrification during storage, compared to solid feces (0.01 vs. 0.14 kg SO_2-eq), and lower NH_3 emission during field application, as liquid feces were now injected compared to surface spreading of solid feces (0.14 vs. 1.06 kg SO_2-eq).

Segregation increased ME with 9% compared to Ref (Table 4.3). This was mainly related to higher NO_3^- leaching after application as a result of more N retained in the feces fraction through avoided N emissions from in-house segregation and storage. Again, FFD was hardly affected (4%) compared to Ref, as total avoided mineral fertilizer and transport were almost equal.

4.3.3 Uncertainty analysis

Fig. 4.2 shows results of uncertainty analysis. Mean values for CC were 318 (95% CI, 160 - 486) kg CO_2-eq for Ref, 108 (95% CI, 48.5 - 180) kg CO_2-eq for the scenario with segregation and solid feces, and 58.3 (95% CI, 5.32 - 132) kg CO_2-eq for scenario with segregation and liquid feces. Relative uncertainty was higher in the scenarios with segregation due to relatively higher N_2O emission with more variation and lower CH_4 emission with less variation.

Table 4.3. Contribution of substances to the impact categories for the scenarios and each life cycle stage (LCS) expressed per FU

Scenario & LCS	Climate change (CC) (kg CO_2-eq)				Terrestrial acidification (TA) (kg SO_2-eq)				Marine eutrophication (ME) (kg N-eq)				Particulate matter formation (PMF) (kg PM_{10}-eq)				FFD (kg oil-eq)
Reference	CO_2	N_2O	CH_4	Total*	SO_2	NO_x	NH_3	Total	NO_3^-	NH_3	NO_x	Total	NH_3	NO_x	SO_2	Total	Total
In-house st.	1.24	10.1	328	339	0	0.01	5.11	5.11	-	0.19	0	0.20	0.67	0	0	0.67	0.48
Outside st.	0.88	0.83	4.25	5.98	0	0	0.52	0.53	-	0.02	0	0.02	0.07	0	0	0.07	0.34
Transport	2.87	0.03	-	3.01	0	0.01	0	0.01	0	0	0.01	0.01	0	0	0	0.01	1.16
Field app.	1.12	58.7	-	59.8	0	0.06	0.33	0.40	1.5	0.01	0.04	1.56	0.04	0.02	0	0.07	0.44
Avoided fert.	-27.5	-59	-	-88	-0.15	-0.11	-0.48	-0.75	-0.77	-0.02	-0.08	-0.87	-0.06	-0.04	-0.03	-0.17	-11.5
Total	-21.3	10.5	332	320	-0.15	-0.03	5.48	5.30	0.73	0.21	-0.02	0.91	0.72	-0.01	-0.03	0.65	-9.12
Seg. solid feces																	
In-house seg.	0.66	35.8	23.5	60	0	0.08	2.48	2.56	-	0.09	0.06	0.15	0.32	0.03	0	0.36	0.35
Urine st.	0.46	0.36	0.18	1.01	0	0	0.23	0.23	-	0.01	0	0.01	0.03	0	0	0.03	0.18
Feces st.	-	56.5	0.05	56.5	0	0.14	0.35	0.49	-	0.01	0.1	0.11	0.05	0.06	0	0.1	-
Transport	2.87	0.03	-	3.02	0	0.01	0	0.01	0	0	0.01	0.01	0	0	0	0.01	1.16
Urine app.	0.61	36.5	-	37.1	0	0.03	0.07	0.1	0.54	0	0.02	0.56	0.01	0.01	0	0.02	0.24
Feces app.	1.44	35.4	-	36.9	0	0.04	1.06	1.1	0.96	0.04	0.03	1.02	0.14	0.02	0	0.16	0.52
Avoided fert.	-27	-57	-	-85.9	-0.15	-0.11	-0.47	-0.73	-0.75	-0.02	-0.08	-0.85	-0.06	-0.04	-0.03	-0.16	-11.3
Total	-20.9	107	23.7	109	-0.14	0.19	3.72	3.77	0.74	0.14	0.13	1.01	0.49	0.08	-0.03	0.51	-8.88
Seg. liquid feces																	
In-house seg.	0.66	35.8	23.5	60	0	0.08	2.48	2.56	-	0.09	0.06	0.15	0.32	0.03	0	0.36	0.35
Urine st.	0.46	0.36	0.18	1.01	0	0	0.23	0.23	-	0.01	0	0.01	0.03	0	0	0.03	0.18
Feces st.	0.42	3.35	2.04	5.82	0	0.01	0.35	0.36	-	0.01	0.01	0.02	0.05	0	0	0.05	0.16
Transport	2.87	0.03	-	3.02	0	0.01	0	0.01	0	0	0.01	0.01	0	0	0	0.01	1.16
Urine app.	0.61	36.5	-	37.1	0	0.03	0.07	0.1	0.54	0	0.02	0.56	0.01	0.01	0	0.02	0.24
Feces app.	0.50	39.1	-	39.6	0	0.04	0.14	0.18	1.11	0.01	0.03	1.15	0.02	0.02	0	0.04	0.20
Avoided fert.	-27.9	-60	-	-90.3	-0.15	-0.11	-0.5	-0.77	-0.8	-0.02	-0.08	-0.9	-0.06	-0.04	-0.03	-0.17	-11.8
Total	-22.4	54.3	25.7	56.3	-0.15	0.05	2.77	2.68	0.86	0.10	0.04	0.99	0.36	0.02	-0.03	0.33	-9.48

* Totals include remaining substances and are rounded and, therefore, do not always correspond to the sum of columns and rows. seg = segregation, st = storage, app = application, and fert = fertilizer.

Mean values for TA ranged from 2.7 (95% CI, 0.79 - 4.66) kg SO_2-eq to 5.3 (95% CI, 4.18 - 6.40) kg SO_2-eq and for PMF from 0.34 (95% CI, 0.074 - 0.58) kg PM_{10}-eq to 0.65 (95% CI, 0.50 - 0.80) kg PM_{10}-eq. The relative uncertainty was higher in the scenarios with segregation due to more variation in their NH_3 emissions as compared to Ref.

Mean values for ME ranged from 0.91 (95% CI, 0.78 - 1.04) kg N-eq to 1.01 (95% CI, 0.86 - 1.16) kg N-eq. Relative uncertainty was similar in all scenarios, as it was mainly related to NO_3^- leaching, having similar variation in all scenarios.

Results of Ref and scenario comparisons showed that in 99.8% and 98.6% of the runs, scenarios with segregation and liquid or solid feces respectively resulted in lower CC compared to Ref (SI, Figure S1) ($P < 0.05$). Segregation and solid feces resulted in lower values than Ref in 89.6% of the runs for TA and 82.7% of the runs for PMF ($P > 0.05$), whereas segregation and liquid feces resulted in lower values than Ref in 99.3% of the runs for TA and 98.6% of the runs for PMF ($P < 0.05$). For ME, both segregation scenarios resulted in lower values only in approximately 30% of the runs compared to Ref ($P > 0.05$). A comparison between segregation scenarios showed that, on average, segregation with liquid feces had lower environmental impact compared to segregation with solid feces, however these differences were not significant ($P > 0.05$).

4.3.4 General discussion

The outcomes show that segregating fattening pig urine and feces significantly reduced CC, and additionally segregation with liquid feces significantly reduced TA, and PMF compared to Ref. Mean values for ME were increased in the scenarios with segregation. Shifting of N emissions within and between impact categories and life cycle stages was highlighted by considering the entire life cycle of manure management; thus emphasizing the necessity of a life cycle perspective when addressing environmental improvement options. Nitrogen was lost in different chemical compounds (i.e. NH_3, N_2O, NO_x, NO_3^-) depending on factors, such as manure product type and storage system. For example, segregation and solid feces reduced NH_3 emissions from in-house segregation, but increased NO_x emission from storage and NH_3 emission during application, whereas segregation and liquid feces increased NO_3^- leaching through more N retained in the fertilizer products. These changes affected TA, PMF, and ME. Moreover, within CC, N_2O and CH_4 from storage varied depending on the oxygen maintained in the storage environment, i.e. more CH_4 was produced under anaerobic conditions, whereas more N_2O was produced under aerobic conditions. Differences in N_2O emission for segregation with liquid or solid feces could not be established due lack of data. Both N_2O and CH_4 contribute in a considerably different extent to CC (a multiplier of 298 for N_2O and 25 for CH_4 (IPCC, 2006b).

Using urine and feces for crop nutrition may enhance crop N-uptake compared to liquid manure when management, e.g. application timing, can be improved. In turn this reduces NO_3^- leaching and ME. However, efficient use of N remains subject to many factors, including soil characteristics, weather conditions, and N application standards, and therefore requires further attention for quantification under different circumstances (Schröder, 2005).

A critical management factor affecting environmental performance in scenarios with segregation was the storage condition (i.e. aerobic or anaerobic) and related management of feces. On average, segregation with liquid feces resulted in lower environmental impact

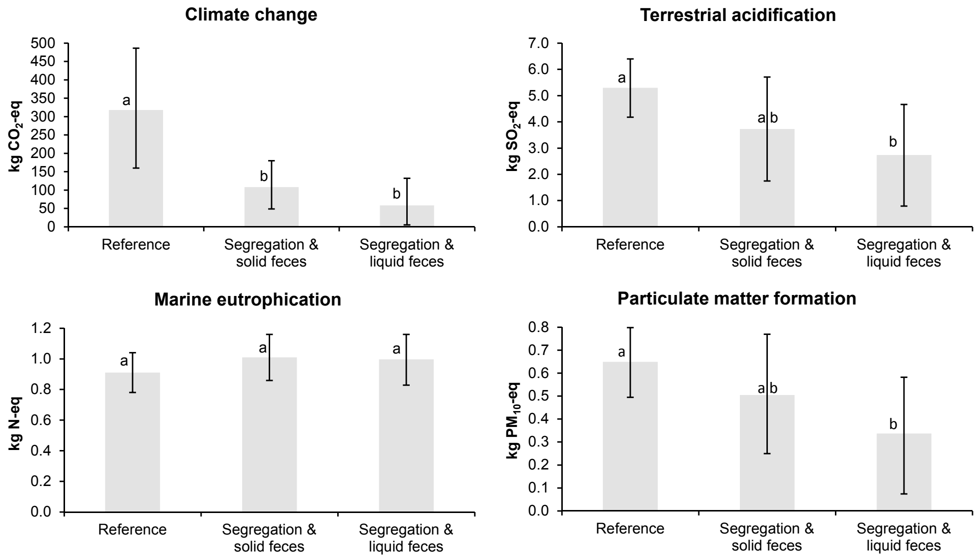

Fig. 4.2. Results of the uncertainty analysis for Ref and scenarios with segregation after 1000 Monte Carlo simulations. Bars represent mean values, error bars represent the 95% confidence intervals. Different letters above bars indicate significant differences.

compared to segregation with solid feces. It would, therefore, be preferable to avoid solid feces production. The most promising management alternative would be to anaerobically digest the solid feces reducing the need of aerobic storage; less N_2O and little CH_4 emit during storage of digestate (De Vries et al., 2012b, Chapter 2; Hamelin et al., 2011). Other alternatives include composting or drying, but these technologies have shown to increase the environmental impact (Lopez-Ridaura et al., 2009).

Methane emissions in the scenarios with segregation were low compared to Ref, as they were mainly related to enteric fermentation of pigs (Aarnink et al., 2007). Furthermore, emissions of CH_4 from storage of urine and feces contributed little to the total (approximately 8%). Methane emissions from liquid and solid fractions were reported to be lower compared to liquid manure due to less available carbon in the liquid fraction and increased oxygen contents during storage of solid fractions, i.e. leading to increased N_2O emission (Chadwick et al., 2011; Mosquera et al., 2010b). The used differences in CH_4 emission between liquid manure and urine and feces during outside storage, however, were high compared to other data. Dinuccio et al. (2008), measured differences between liquid manure, separated liquid fraction, and solid fraction were 1.3 times higher and 4.9 times lower respectively. On the one hand, this might indicate that CH_4 emissions from urine and feces storage are underestimated here. On the other hand, the emission factors and ratios in Dinuccio et al. (2008) were determined under different circumstances (e.g. 30 days at 25 ˚C). Hence, determining appropriate emission factors remains critical and requires additional monitoring in practice. However, when testing the effect of lower CH_4 emission ratios for urine and feces compared to liquid manure, conclusions did not change (CC increased 5%); highlighting the relatively small contribution to CC.

Segregation provides a potential to reduce NH_3 emission compared to conventional manure management and can contribute to achieve the goals of the NEC. When all pig manure in the Netherlands (11.8 million tons in 2010) would be segregated, assuming liquid feces, this provides a reduction of roughly 13 kiloton NH_3 (11% of total NH_3 emitted: 122 kiloton in the Netherlands in 2010). Furthermore, segregation reduces in-house odor emission (approximately 70%) compared to conventional manure storage and may result in less nuisance in residential areas, as critical threshold values will not be exceeded as often. Odor impacts, however, remain strongly dependent on local climatologic conditions and the location of the farm (VROM, 2012).

Other issues, not considered here, include animal welfare and costs of the systems. The possibility of adding bedding material with the Kempfarm® system provides the potential to improve animal welfare, as bedding material results for instance in less tail biting with pigs (Day et al., 2002; Zonderland et al., 2008). Environmental issues resulting from the addition of bedding material, such as PMF, require further investigation. Costs of an animal house including the Kempfarm® segregation system were reported to be similar as compared to conventional housing systems, representing a potential to reduce environmental impact at similar costs (Aarnink et al., 2007). Economic consequences of the manure management chain including manure offset costs, however, were out of scope.

In the scenarios with segregation, FFD did not change compared to conventional manure management. However, segregation offers an opportunity to reduce transportation and as such FFD and CC. Differences in transportation were not included here, as they depend on the specific location of farms and were studied in detail by others, e.g. De Vries et al. (2012a) (Chapter 3) and Lopez-Ridaura et al. (2009). Transportation of manure products is especially important in locations with high livestock densities with local nutrient surpluses (e.g. P). Therefore, liquid manure or other fractions have to be transported to other regions. In the scenarios with segregation, transport of feces with most of the P will reduce the need of transporting water, and thus weight, reducing FFD and CC. Moreover, when feces are exported and applied on soils with low P-saturation, this may reduce overall P-leaching, as the P-surplus will be reduced in soils more close to the farms (Behrendt & Boekhold, 1993). However, such leaching and management changes also depend on the specific location and were excluded here.

Uncertainty analysis in LCAs is essential, but due to several reasons, such as lack of data and knowledge of distribution functions, it is very often not included (Payraudeau et al., 2007). In this study results of uncertainty analysis supported that segregation reduced CC, and additionally segregation with liquid feces reduced TA and PMF compared to Ref. Uncertainty was strongly related to variation in NH_3 emission and to variation in N_2O emission. Similar results were found by other authors (Basset-Mens & van der Werf, 2005; Payraudeau et al., 2007). On the one hand, this highlights the necessity to include appropriate emission factors and their variation in relation to the studied circumstances. On the other hand, the variation of emission factors originates from different sources, including feed, type of system and management, and climatologic conditions (i.e. representing different sources of uncertainty, e.g. Huijbregts (1998) and Walker et al. (2003)). The variation of emission factors used here included different sources, as they were taken from different literature references and measured over different time periods and systems.

Sources of variation in emission factors, therefore, could not fully be uncoupled and require more attention when aiming at reducing uncertainty in the end results. Nevertheless, including such analysis, with best available data, is essential for determining the magnitude of uncertainty in the end results which affect interpretation of results and conclusions, but even more important, final decisions.

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ASSOCIATED CONTENT

Supporting Information. Contains the scaling of urine and feces compositions and Figure S1 presenting results of the uncertainty comparison between scenarios. This material is available free of charge via the Internet at: pubs.acs.org/doi/abs/10.1021/es302951a.

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Chapter 5

// INTEGRATED MANURE MANAGEMENT TO REDUCE ENVIRONMENTAL IMPACT: I. STRUCTURED DESIGN OF STRATEGIES

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ABSTRACT

In livestock production, management of animal manure is a major cause of nitrogen (N), phosphorus (P), and carbon (C) loss. The losses of N, P, and C contribute to adverse environmental impacts, such as climate change, terrestrial acidification, and marine eutrophication. Manure management technologies to reduce losses and impacts have been developed, but often focus on a single compound only or a single stage in the management system and lead to trade-offs, such as pollution swapping. The aim of this study was to design strategies for integrated manure management (IS) that show that the environmental impact can be reduced throughout the manure management system by at least 75% and prevent pollution swapping. We used a structured design approach based on engineering design (ED) that exists of eight main steps: 1. define the goal of the assignment and the system boundaries, 2. formulate a brief of requirements stating the needs for environmental reduction, 3. analyze the functions in the current manure management system, 4. list and describe emission processes and their process variables that lead to N, P, and C losses and fossil energy use, 5. describe the functions needed in the manure management system to limit the emission processes or resource use, 6. generate principle-options that can fulfill the functions, 7. generate technical solutions for the principle-options, and 8. combine the principle-options and technical solutions into strategies for integrated manure management. In the design of strategies we considered the management of liquid and solid dairy cattle manure applied to grass and maize, and liquid pig manure applied to wheat, all under North West European conditions. The IS included the segregation of pig and dairy cattle urine and feces to reduce CH_4, NH_3, and N_2O emission, addition of zeolite to solid cattle manure to reduce NH_3 emission, bio-energy production from biogas that avoids fossil-based electricity and heat, acidification of urine during storage and acidification of feces prior to application, sealed storages, and improved application timing, place, and method of application. It was concluded that we were able to successfully design IS with high potential to reduce environmental impact. The design approach adapted from ED, proved to be useful to structure the design process, to provide insight into interactions of emission processes, and find principal-options causing pollution swapping.

5.1 INTRODUCTION

Management of animal manure, i.e. collection inside the housing system, storage, processing, and field application causes losses of nitrogen (N), phosphorus (P), and carbon (C). These losses lead to environmental and human health impacts, including climate change (CC), terrestrial acidification (TA), marine eutrophication (ME), and particulate matter formation (PMF). N is lost mainly as ammonia (NH_3), nitrous oxide (N_2O), nitrogen oxide (NO), and harmless nitrogen gas (N_2), and as nitrate (NO_3^-), which leaches and runs-off to ground and surface waters (Bouwman et al., 2011). P, not taken up by crops, is retained in the soil, where it is susceptible to leaching and run-off (Schröder et al., 2011). C, that contributes to CC, is mainly lost as methane (CH_4), and as carbon dioxide (CO_2) from fossil fuel use (Berglund & Börjesson, 2006). N and P losses result in low nutrient use efficiencies, such as N use efficiency. Reducing N losses and increasing N use efficiency will lead to reduced environmental impact from crop and animal production systems (Cassman et al., 2002; Spiertz, 2010). N, P, and C losses occur from three basic emission processes, that include two main steps: the production or conversion of a compound to another compound, and the volatilization of the compound, that is the transfer from the manure to air. The processes are: 1. enzymatic conversion of urea to ammonium (NH_4^+) and NH_3, and the volatilization of NH_3, 2. anaerobic digestion, the conversion of organic matter to CH_4 and CO_2 and the volatilization of CH_4 and CO_2, and 3. nitrification and denitrification, conversion of NH_4^+ to NO_3^- and NO_3^- to N_2 with production and volatilization of N_2O and NO as by-products. The conversion and volatilization rates of these processes are affected by process variables, such as temperature, C and N composition and pH of manure.

Technologies for manure management were developed to reduce losses of N, P, and C and intervene on the production or volatilization process. These technologies often focused on reduction of a single compound, like NH_3, or in a single management stage, like outside storage. Examples include: covering of manure storages, anaerobic digestion and other forms of processing, and injection of liquid manure instead of broadcast spreading (Burton & Turner, 2003; Sommer & Hutchings, 2001). As a result, the targeted emissions were reduced, but others were increased. This trade-off is referred to as 'pollution swapping'. Pollution swapping is difficult or seemingly impossible to prevent, because of complex interactions between emission processes and their process variables (Groenestein, 2006; Jarvis & Menzi, 2004). For example, covering manure storages reduces odor and NH_3 emission both up to 95% (Bicudo et al., 2004), but often increases N_2O emission by more than 4.5 fold (Berg et al., 2006). Injection and incorporation of liquid and solid manure both reduce NH_3 emission up to 90% as compared to broadcast spreading (Sommer & Hutchings, 2001), but increases N_2O emission by more than 3 fold (Velthof & Mosquera, 2011). More holistic assessments of the environmental impact of manure management technologies, by applying life cycle assessment (LCA), also revealed that current technologies cause pollution swapping (De Vries et al., 2012a, Chapter 3; Lopez-Ridaura et al., 2009). However, De Vries et al. (2013) (Chapter 4) recently demonstrated that pollution swapping between CH_4, NH_3, and N_2O in the housing system can be prevented by keeping pig urine and feces separate immediately after excretion, further referred to as segregation. But NO_3^- leaching after field application tended to increase. Thus current technical solutions often induce pollution swapping, because they do not take into account the underlying emission processes. To prevent pollution swapping in the whole system, we therefore, need to reconsider the emission processes and their interactions in a structured way along the manure management system. From this knowledge, new designs can be created that reduce environmental impact throughout the manure management system, i.e. ‘strategies for integrated manure management’. This can be done by applying engineering design (ED), as described by Siers (2004) for technical design and applied to agricultural systems by Bos et al. (2009). However, this method requires adaption for application to the manure management system.

The aim of this study was to design strategies from integrated pig and dairy cattle manure management that show that a reduction of at least 75% can be achieved in six environmental impact categories. These strategies were created by following a structured design approach based on ED.

5.2 MATERIALS AND METHODS

5.2.1 Engineering design

The following main steps used in ED were taken to design new strategies (Bos et al., 2009; Cross, 2008; Siers, 2004): 1. define the goal of the design assignment and the system boundaries, 2. formulate a brief of requirements to be achieved by future manure management strategies, 3. analyze the functions in the current manure management system, 4. list and describe emission processes, their process variables that lead to N, P, and C emissions and fossil energy use, 5. describe the functions that are needed in the new manure management system to limit the emission processes or resource use by intervening on the process variables, 6. generate principle-options to fulfill the functions, 7. generate technical solutions to execute the principle-options, and 8. combine the principle-options and technical solutions into strategies for integrated manure management (IS).

5.2.1.1 Goal of the design assignment and system boundary (step 1)

The goal of the design assignment was to create strategies for integrated manure management to reduce the environmental impact throughout the manure management system, or in other words to prevent pollution swapping. This meant we focused on the design for the environment, and excluded other stakeholders and factors, such as costs and the final implementation. Environmental impact categories included were: CC, TA, ME, PMF, fossil fuel depletion (FFD), and the phosphorus surplus in the soil. The manure management system in our study included the excretion of urine and feces in the animal house, storage and processing, transport, field application with soil tillage and in-field traffic, and crop uptake until harvest (Fig. 5.1). For the IS, we considered production and management of liquid dairy cattle manure applied to grass and maize, solid dairy cattle manure applied to grass, and liquid pig manure applied to wheat, all under North West European (EU) conditions with sandy and clay soils. For further details also see Chapter 6.

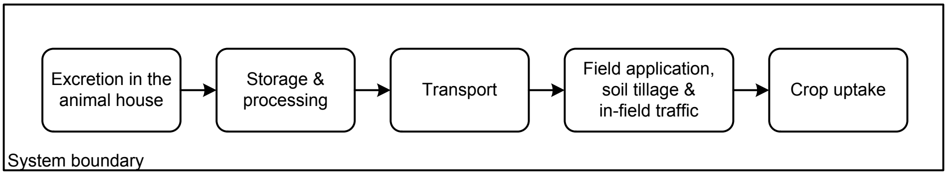

Fig. 5.1. Processes included in the manure management system and the system boundary.

5.2.1.2 Brief of requirements (step 2)

ED includes an elaborate inventory of needs and requirements of the stakeholders that sets the targets to be achieved. In our study, we based the brief of requirements on the 'long-term needs' to reduce the environmental impact. Long-term needs focus on being in the 'environmentally safe range' and go beyond the current regulations for addressing pollutants, such as the EU National Emission Ceilings and the Nitrates Directive.

Emissions of acidifying and eutrophying compounds, such as NH_3, NO_x, and NO_3^- were reduced in the past years compared to the base year 1990 (EC, 2012; EU, 2013). Additional reduction, however, is needed especially in more sensitive natural areas, such as Natura2000 zones, and areas with intensive livestock production, such as the Netherlands, the Po valley in Italy, and Brittany in France, where the coast is plagued with algae because of N-surplus. Currently, N concentrations in surface waters in the Netherlands, remain to be 2 times higher than considered sustainable (CBS et al., 2012b; Van Puijenbroek et al., 2010). Worldwide, we know that drastic reductions in emission of reactive N, such as NH_3 and NO_x, are required to avoid severe damage to environmental services (Galloway et al., 2008). Agriculture, and specifically manure management, remains to be one of the largest contributors to emission of acidifying and eutrophying compounds (CBS et al., 2012a). With regard to global CC impact, at least a 50% reduction in greenhouse gas emission is needed to remain below 2 degrees increase in global temperature and to be carbon neutral at the end of this century (EC, 2008). Expressed in monetary value, agriculture remains one of the most greenhouse gas intensive sectors (EEA, 2013).

5.2.1.3 Function analysis of the current system (step 3)

The current manure management system was decomposed into its basic functions. The goal of this analysis was to identify and understand the relationships between all functions in the current manure management system. Later in the design process, functions were redefined, added, or removed from the system in order to serve the environment while preventing pollution swapping and meet the requirements (section 2.1.5). The analysis was done by creating an Integration Definition for Function (IDEF0) diagram (Anonymous, 2010). An IDEF0 diagram represents the decomposition of a system from its main function(s) (the TOP level) into sub-functions in lower levels of the system (level A0, A1, etc.). Each function shows inputs and the conversion into outputs as well as the mechanisms needed to support the function and the controls that apply to the function. Controls included the need for N and P application timing, placement and method of application related to the specific crop.

5.2.1.4 Emission processes and process variables (step 4)
Emission is the result of the production of a compound and the release to the air (volatilization), soil or groundwater (leaching and runs-off). These processes together with their process variables and resource use were listed and described.

5.2.1.5 Functions to limit the emission process (step 5)
Functions were defined that limit the emission process and resource use by intervening on the process variables. Functions limited either by intervening at the level of production, for example conversion of NH_4^+ to NH_3, or at the level of volatilization or run-off, for example volatilization of NH_3. The functions were formulated according to IDEF0.

5.2.1.6 Principle-options to fulfill the functions (step 6)
Principle-options to fulfill each function were generated. A principle-option represents the basic action and working principle that needs to be implemented in order to fulfill the function that limits the emission process. The principle-options do not yet include the final technical means. Principle-options can affect other process variables or emission processes. Interactions between principle-options and process variables were identified that may cause pollution swapping or simultaneously limit emission process(es). Recognizing these interaction is necessary to choose the final technical solutions in accordance with the goal.

5.2.1.7 Technical solutions (step 7)
Technical solutions were defined as a means for executing the defined principle-options in section 5.2.1.6. The technical solutions may include current technology, or technology that requires development. To be able to make optimal combinations of solutions to fulfill functions (step 8), several technical solution per principle-option were listed.

5.2.1.8 Creating strategies for integrated manure management (step 8)
Based on the previous obtained knowledge and expert judgment, the IS were created by selecting and combining principal-options and technical solutions for the needed functions in the manure management system.

5.3 RESULTS

5.3.1 Goal of the design assignment and brief of requirements (step 1 and 2)
The goals of the assignment was to reduce the environmental impact by designing strategies for integrated manure management. Taking into account the long-term needs to reduce environmental impact, we established a goal to achieve a reduction of at least 100% for CC and FFD impact and at least 75% for all other environmental impact categories compared to current manure management in NW EU.

5.3.2 Functions in the current system (step 3)
The main function of the current manure management system was defined as: manage

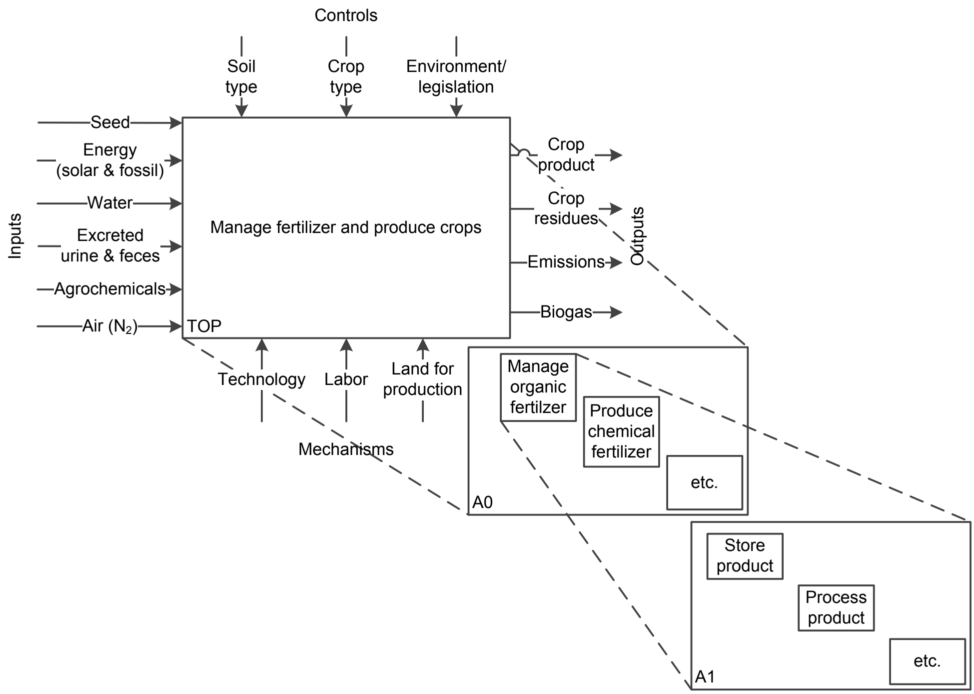

Fig. 5.2. Main function (TOP level), sub functions (A0 and A1 level), inputs, outputs, controls and mechanisms of the current manure management system according to the IDEF0 approach.

fertilizer and grow crops (TOP level in Fig. 5.2). This function was subsequently split out in lower levels or sub functions, level A0: manage organic fertilizer, produce chemical fertilizer, etc. The function manage organic fertilizer was split out into level A1: store product, process product, etc. (De Vries et al., in preparation). The functions convert the inputs: seed, energy, water, urine and feces, agrochemicals, and air into the outputs: crop product, crop residue, emissions, and biogas. The conversion occurs under the control of the specific soil and crop, and environment/ legislation and is supported by the mechanisms: technology, labor, and arable land.

5.3.3 Emission processes, functions, principle-options, and technical solutions (step 4 to 7)

Emission processes and resource use were subdivided into: CH_4 production and volatilization during storage, N_2O production and volatilization during storage and field application (direct N_2O emission) and NO_3^- leaching after application, indirect emission of N_2O following from NH_3 and NO_3^- emission, fossil energy use during management, NH_3 production and volatilization during storage and field application, N and P run-off and leaching after field application, soil C depletion, and particulate matter formation following from emission of NH_3, NO_x, and SO_2 (Table 5.1). For example, process 1 in Table 5.1 describes the process of CH_4 production and volatilization and gives the main process variables: availability of C, temperature, oxygen concentration, pH, and contact time and area with air. The next column in Table 5.1 describes the functions with which CH_4 production and volatilization can be limited because they intervene on the emission process at the level of the process variables: remove available C or separate available C from non-available C, lower temperature, increase oxygen concentration in the storage environment, lower pH (< 5) and capture CH_4 in the storage environment. These functions can be fulfilled by principle-options, answering the question: what can be done? Answers are: 1. segregate urine and feces, 2. anaerobically digest manure product, and 3. cool storage. The principle-options can be executed by technical solutions, answering the question: how can it be done? Answers are e.g.: 1. V-belt or filter nets, 2. continuous stirred tank reactor or plug flow digester, and 3. cool decks. In the last column of Table 5.1, interactions between principal-options with process variables of (an)other emission process(es) are indicated. For example, the principal-option 'increase oxygen concentration in the storage environment' interacts with emission process 2b, as aerobic conditions may stimulate nitrification and the volatilization of N_2O. It also interacts with resource use 5a and 5b, because energy is needed for aeration. With advancing understanding and technical developments this table can be expanded.

5.3.4 Strategies for integrated manure management (step 8)

The generated principle-options and technical solutions used for creating the IS are underlined in Table 5.1. In this section we describe the IS according to the stages in the manure management system (Fig. 5.1) for the cases as depicted in section 5.2.1.1.

5.3.4.1 Excretion in the animal house

In IS, segregation of urine and feces from pigs and dairy cattle occurs directly after excretion as this provides the potential to avoid production of CH_4 and NH_3 mainly through faster removal from the housing system (Aarnink et al., 2007; De Vries et al., 2013, Chapter 4). Segregating urine and feces also reduces methanogenic activity in the animal house through more aerobic conditions and lack of inoculating material (process 1 and 6a/b in Table 5.1). It also separates available C, such as organic acids (mainly in feces), from NH_4^+ (mainly in urine) which reduces N_2O production and volatilization in the following stages of manure management (process 2a/b in Table 5.1). Segregation of pig urine and feces is achieved by using a V-shaped belt underneath the slatted floors (De Vries et al., 2013, Chapter 4). The urine flows down the belt and the feces are removed two times per day by rotating the belt. In a dairy house segregation of urine and feces is obtained by a grooved floor system (Swierstra et al., 2001). The urine is drained through small openings in the grooves and the feces are scraped to a separate storage system with a manure scraper.

The IS for solid cattle manure includes the addition of zeolite to reduce NH_3 volatilization during in-house management, storage, and field application without increasing other emissions (process 6 in Table 5.1) (Shah et al., 2012).

Table 5.1. Emission processes and process variables, resource use, functions in the system to be designed, principle-options to fulfill the functions, technical solutions, and interactions between solutions. Underlined principle-options and technical solutions are used to construct the strategies for integrated manure management

#	Emission processes, or resource use, with the production and volatilization process (in bold), description of the process (plain text), and process variables (in italics)	Functions in the system to be designed that are needed to limit the emission processes or resource use	Principle-options to fulfill the functions	Technical solutions for the principle-options	Interacts with
1	**CH_4 production and volatilization during storage**				
	Available C in manure product under anaerobic conditions leads to uncontrolled CH_4 production and volatilization.[a]				
	- availability of carbon	Remove available C; Separate available C from non-available C	<u>Segregate urine and feces</u>; <u>Anaerobically digest manure product</u>	<u>V-belt</u>, <u>grooved floor</u>, filter net, Hercules system; plug flow, <u>CSTR</u>[c], batch	2a, 6a
	- temperature	Lower temperature (as low as possible)	Cool storage; <u>move product to (cooler) outside storage</u>	Cool decks, refrigeration; <u>pumping</u>, <u>conveyor</u>, manual	2a, 5a, 5c
	- oxygen concentration	Increase oxygen concentration in the storage environment	Aerate manure product		2b, 5a, 5c
	- pH	Lower pH (< 5)	<u>Acidify manure product</u>	<u>H_2SO_4</u>, HNO_3; <u>dosing</u>, <u>mixing</u>, manual addition	2, 3
	- contact time & area with air	Capture CH_4 in storage environment	Cover or <u>seal storage</u>; <u>Anaerobically digest manure product</u>	Roof, <u>cap</u>, cans, bags; plug flow, <u>CSTR</u>, batch	6b
2	**N_2O production and volatilization during storage and field application**				
a	Available NO_3^- and NH_4^+ under anaerobic conditions leads to (de)nitrification and volatilization of N_2O, NO, and NO_2.[b]				
	- availability of carbon	Remove available C; Separate available C from NH_4^+	<u>Segregate urine and feces</u>; <u>Anaerobically digest manure product</u>	<u>V-belt</u>, <u>grooved floor</u>, filter net, Hercules system; plug flow, <u>CSTR</u>, batch	1, 6a
	- oxygen concentration	Maintain soil porosity (prevent compaction)	<u>Reduce field trafficking</u>; <u>Minimize tillage</u>; Use light machinery	<u>Controlled traffic farming</u>; <u>non-inversion tillage</u>, direct seeding, no-till; robots, small tractors	5a, 5c, 7
	- pH	Lower pH of manure product (< 6) Maintain soil pH (around 6)	<u>Acidify manure product</u>	<u>H_2SO_4</u>, HNO_3; <u>dosing</u>, <u>mixing</u>, manual addition	2b, 3

	- temperature	Lower temperature (as low as possible)	Cool storage; move product to (cooler) outside storage; Adjust application timing to winter	Cool decks, refrigeration; pumping, conveyor, manual; (deep) injection, deep injection in pockets, banding, spreading and incorporation with plow or cultivator	1, 3, 5a, 5c
b	Available NH_4^+ and NO_3^- under overall aerobic conditions, but **local anaerobic conditions**, leads to (de)nitrification and volatilization of N_2O, NO, and NO_2.[b]				
	- See 2a; local oxygen concentration is highly uncontrollable	Process solid manure/ fraction	Anaerobically digest manure product	Plug flow, CSTR, batch	1, 2a, 6b
		Capture N_2O, NO, and NO_2 in storage environment	Cover or seal storage; Anaerobically digest manure product	Roof, cap, cans, bags; plug flow, CSTR, batch	1, 2a, 6b
		Inhibit denitrification and nitrification	Add inhibitors	Dosing, mixing, manual	3, 5a, 5c, 6a
3	**N_2O production and volatilization and NO_3^- leaching during and after field application related to N availability and N uptake by the crop**				
	High concentrations of NH_4^+ from application and mineralization of N_{org} lead to nitrification and volatilization of N_2O and high concentrations of NO_3^-.[c] In turn, high concentrations of NO_3^- during long periods of time lead to denitrification and volatilization of N_2O and leaching NO_3^-.[c] *- temperature*	Lower temperature (as low as possible)	Adjust application timing to winter or early spring	(Deep) injection, deep injection in pockets, banding, spreading and incorporation with plow or cultivator	7
	- concentration	Facilitate N_{org} mineralization and uptake	Adjust application timing of N_{org} to before or early in the growing season; Apply precise in rooting zone	(Deep) injection, deep injection in pockets, banding, spreading and incorporation with plow or cultivator; GPS, machine vision	2a, 6a, 6b, 7
		Facilitate N_{min} uptake	Adjust application timing and rates to 5 - 10 times per season; Apply precise in the rooting zone	(Deep) injection, deep injection in pockets, banding; GPS, machine vision	2a, 6a, 6b, 7
		Inhibit (de)nitrification	Add inhibitors	Dosing, mixing, manual	2b, 5a, 5c, 6a

4		**Indirect emission of N_2O following from other emissions**				
		Emission of NO_3^-, NO_x and NH_3 leads to denitrification and finally volatilization of N_2O	Reduce N losses	See other solutions	See other solutions	
5		**Fossil energy use during management (resource use and environmental impact)**				
	a	Use of fossil energy sources leads to depletion	Reduce fossil energy use	Reduce total energy use; use renewable energy	Wind, solar, hydro power, biomass	External
			Produce bio-energy	Anaerobically digest manure product	Plug flow, CSTR, batch	1, 2a, 2b, 5c,
	b	N losses lead to lower crop nutritional value of manure products and requires, within the same cropping system, more mineral N fertilizer which needs (fossil) energy to produce	Reduce N losses	See other solutions	See other solutions	
	c	Using fossil-based energy contributes to environmental impact[d].	Reduce fossil energy use	See 5a	See 5a	External
	d	Use of co-substrates, such as animal feed, in current practice of biogas production increases environmental impact[d].	Convert available C in manure product to CH_4	Mono-digest manure product	Plug flow, CSTR, batch	External
6		**NH_3 production and volatilization during storage and field application**				
	a	Conversion of urea to NH_3 leads to higher concentrations of NH_3.[e]	Absorb NH_3	Add absorbent	Zeolite, lava-meal, soil; dosing, mixing, manual addition	
		- urea and enzyme contact	Separate urea and enzyme	Segregate urine and feces	V-belt, grooved floor, filter net, Hercules system	
		- temperature	Lower temperature (as low as possible)	Cool storage; move product to (cooler) outside storage; Apply product in winter	Cool decks, refrigeration; pumping, conveyor, manual; (deep) injection, deep injection in pockets, banding, spreading and incorporation with plow or cultivator	3, 5a, 5c
		- enzyme activity	Inhibit uric acid conversion	Add inhibitors	Dosing, mixing, manual addition	
		- pH	Lower pH (< 6)	Acidify manure product	H_2SO_4, HNO_3; dosing, mixing, manual addition	1, 2, 3
	b	Contact of manure with open air leads to NH_3 volatilization.[e]	Prevent contact with open air	Apply below soil surface	(Deep) injection, deep injection in pockets, spreading and incorporation with plow, cultivator	2a, 2b

		- temperature	Lower temperature (as low as possible)	Cool storage; move product to (cooler) outside storage; Apply product in winter	Cool decks, refrigeration; pumping, conveyor, manual; (deep) injection, deep injection in pockets, banding, spreading and incorporation with plow or cultivator	3, 5a, 5c
		- pH	Lower pH (< 6)	Acidify manure product	H_2SO_4, HNO_3; dosing, mixing, manual addition	1, 2, 3
		- air contact time	Capture NH_3 inside the animal housing system or storage	Clean air; Cover or seal storage; Anaerobically digest manure product	Acid scrubber, bio-scrubber; roof, cap, cans, bags; plug flow, CSTR, batch	1, 2a, 5a, 5c
		- concentration of NH_3	Facilitate N_{org} and N_{min} uptake	See 3a, 3b	See 3a, 3b	
			Dilute manure product	Apply water with storage and application	Dosing, mixing, manual addition, irrigate, broadcast spreading of water	6a, 5a, 5c
7		**N and P run-off and leaching after field application**				
	a	N and P ratios of liquid manure most often do not correspond to the N and P requirements of crops leading to over application or under application of N or P.[f]	Separate N and P	Segregate urine and feces; Separate liquid manure; Produce struvite; Strip N	V-belt, grooved floor, filter net, Hercules system; screw press, sieve belt press, decanter; precipitation; NH_3 stripping	1, 2a, 2b, 5a, 5c, 6a, 6b
	b	Concentrations of P in soil above the saturation limit (soil dependent) increases losses of P under downward (leaching) and horizontal (run-off) water movement conditions.[f]	Maintain suitable P concentration	Export surplus P; Adjust application timing, rate, place, see 3	Lorry, tractor and trailer, conveyor; see 3	5a, 5c
	c	Concentrations of NO_3^- in soil above binding capacity (soil dependent) increases losses under downward (leaching) and horizontal (run-off) water movement conditions.[f]	Maintain suitable N concentration	Export surplus N; Adjust application timing, rate, place, see 3	Lorry, tractor and trailer, conveyor; see 3	5a, 5c
			Facilitate nutrient retention	Minimize soil disturbance	Non-inversion tillage, direct seeding, no-till	5a, 5c
8		**Soil carbon depletion (resource use)**				
		Intensive tillage leads to degradation of organic matter and increased carbon emission. This may reduce soil C stocks.	Prevent soil disturbance	Minimize soil tillage	Non-inversion tillage, direct seeding, no-till	5a, 5c
9		**Particulate matter formation following from other emissions**				
		Gaseous emissions (NH_3, NO_x, SO_2 as precursors) lead to particulate matter formation.	Reduce N losses; Reduce fossil energy use	See other solutions	See other solutions	

[*] CSTR = continuous stirred tank reactor

[a] In current practice, available C in liquid manure leads to uncontrolled CH_4 emission from storages. In the strategies for integrated manure management (IS) we avoid this by segregating urine and feces and anaerobically digesting the segregated feces. This provides valuable bio-energy, but also removes available C which limits N_2O emission from denitrification when applied to the field.

[b] N_2O emissions are influenced by available C, aerobic conditions in storage (nitrification) and (local) anaerobic conditions in storage and during field application through (de)nitrification. Also pH and temperature play an important role. In the IS, available C is removed by segregation and anaerobic digestion of the feces. The urine is acidified in the storage by dosing and mixing to pH < 6 which reduces N_2O production as well as NH_3 volatilization. Digested feces are acidified just prior to application which reduces N_2O production and NH_3 volatilization after field application. By applying controlled traffic farming, the soil is more aerated with oxygen which reduces N_2O production. For solid cattle manure, sealed storage was applied that induced anaerobic conditions and reduced the air exchange with the atmosphere leading to reduced N_2O production and volatilization, but also reduced NH_3 and CH_4 volatilization.

[c] In current practice, application of manure occurs once or twice per growing season. This leads to high concentration of NH_4^+ and consequently NO_3^-. In the IS, this is tackled by applying smaller amounts of urine more often during the growing season. Digested feces are applied just prior to the growing season to leave some time for the N_{org} to mineralize and provide available NH_4^+ already for the crop at planting. Digestion is applied to adjust the N_{org}/N_{min} ratio and stabilize the N_{org} fraction.

[d] External affects to the manure management system. Using fossil energy contributes to emissions of various compounds, including CO_2, NO_x, and SO_2 and contribute to different environmental impacts, such as climate change, terrestrial acidification, and particulate matter formation. The emissions, however, occur outside the manure management system. Production of biogas from manure is economically limited due to the low biogas production potential. C-rich co-substrates are added to increase biogas production. However, co-substrates often compete with animal feed products which require a substitute. Producing a substitute increases the environmental impact (De Vries et al., 2012b).

[e] NH_3 emission occurs from in-house and outside storage and after application of liquid manures. The pH of liquid manure is often about 7.5 to 8. This stimulates the volatilization of NH_3. Moreover, air contact and urease-enzyme contact is stimulated in current practice. In the IS, segregation reduced NH_3 emission through fast removal to sealed storage. Additionally, urine was acidified during storage and urine and feces were deep injected in nests or pockets to reduce NH_3 volatilization and were applied 5-10 times per growing season to reduce N concentration and N_2O production and volatilization. For solid cattle manure, additives (zeolite) in the housing system were added to reduce NH_3 volatilization during storage and application. Field application was combined with rainfall or water application to further reduce NH_3 volatilization.

[f] In the current system, the spatial distribution of nutrients through the soil profile is very variable. In the IS, urine and digested feces are applied directly at the plant roots to facilitate uptake of N and P and reduce losses.

5.3.4.2 Storage & processing

Pig as well as cattle urine is acidified with sulfuric acid (H_2SO_4) directly in the storage to keep the pH below 6 and avoid NH_3 volatilization, and also after application (process 6a/b in Table 5.1) (Webb et al., 2013). Pig and cattle feces are anaerobically digested to remove easily degradable C that reduces N_2O production and volatilization after field application. Anaerobic digestion also mineralizes N_{org} to N_{min} which increases crop availability of N (process 1, 2a, and 3 in Table 5.1). Moreover, bio-energy is produced from digestion resulting in avoided CH_4 emission to the atmosphere during storage and reduces the need / use of fossil energy (process 5 in Table 5.1) (De Vries et al., 2012b, Chapter 2; Schouten et al., 2012). Further, all IS include sealed outside storage of all products to avoid contact with outside air avoiding volatilization of NH_3, N_2O, NO, N_2, and CH_4 (process 1, 2a, and 6a/b in Table 5.1).

5.3.4.3 Field application, soil tillage & in-field traffic

In IS, pig and dairy cattle urine are applied 5 - 10 times during the growing season to reduce N concentrations in the field (process 3 in Table 5.1) and by deep injection into pockets to reduce NH_3 emission and improve root contact (process 6b in Table 5.1). Urine is applied at a depth of 10 cm at relatively dry conditions to avoid N_2O production (process 2a and 3a/b in Table 5.1) (Webb et al., 2013). Digested pig and cattle feces are applied by deep injection to reduce NH_3 volatilization and are acidified prior to injection to further reduce NH_3 volatilization and limit nitrification (process 2a and 6a/b in Table 5.1). Digested feces applied to arable land occurs just prior to or with planting at a depth of 5 cm within reach of roots (process 3 in Table 5.1). Digested feces applied to grassland is carried out by deep injection in pockets at a depth of 10 cm, as roots are already established. We assume that deep injection in pockets with acidified feces do not lead to increased N_2O formation due to less available C, lower NH_4^+ concentration as compared to urine and lower pH. Solid cattle manure is applied prior to the growing season by broadcast spreading and addition of water by irrigation, broadcast spreading or a rainfall event. Water increases the infiltration rate and reduces NH_3 volatilization (process 6b in Table 5.1).

In the IS, soil tillage occurs by non-inversion tillage for the arable crops to save fossil fuel and maintain soil structure (process 2a, 5, 7c, and 8 in Table 5.1). Field trafficking occurs by controlled traffic farming in all IS to avoid soil compaction and anaerobic conditions that promote denitrification leading to production and volatilization of N_2O (process 2a in Table 5.1) (Venterea et al., 2005; Vermeulen & Mosquera, 2009).

5.4 DISCUSSION

The structured design approach based on ED proved to be useful for constructing strategies for integrated manure management. The method provided the structure to comprehend the entire system including the listing of emission processes, defining the required functions, principal-options, and technical solutions and selecting the combination of principal-options according to the goal. Extending the method to the complete manure chain gave us insight into different levels of the system, from the basic emission processes to the technical means that are needed. It also showed us that principal-options for different functions can simultaneously limit more than one emission process and prevent pollution swapping. This indicates that the prevention of pollution swapping generally should be addressed at the basic functions and principle-options instead of at the level of final technical solutions. Consequently, the IS represent a fundamental reevaluation of the manure management system instead of a mere 'technical fix'. This approach, therefore, goes beyond conventional approaches used in farming systems analysis, where understanding and quantification of the current system is sought, instead of addressing the needs and the required functions in a new system. This feature of considering the basic functions and interactions in a system to create new designs, therefore, is very useful for application in other biosystems design.

IS were designed to reduce the environmental impact of manure management to fulfill the brief of requirements and therefore have to be assessed in quantitative terms. A quantification of the N, P, and C losses and the environmental impacts of the designed strategies is described in detail in Chapter 6. In the future, IS should be developed taking other stakeholders into consideration, such as animals, farmers (costs), and technology developers (Bos et al., 2009). For every stakeholder, a specific brief of requirements is needed. Extending the number of stakeholders, however, should also at this level not lead to trade-offs between different interests, but the design process should aim at finding agreement and congruency on the goal to be reached (Bos et al., 2009). The information provided here leaves a sound basis and outlook for further development of the IS. It indicates what can be reached based on current knowledge of biological and technical principles.

In the design approach, IS were created by combining principle-options and technical solutions. Designing IS, therefore, included creativity in combining the principle-options and technical solutions based on the information available (Bos et al., 2009). The designed IS represent strategies aimed to fulfill the design goal and brief of requirements, which were achieved in the IS, but they do not necessarily represent the maximum feasible reduction in environmental impact, nor a 'one-size-fits-all' solution. In the process of engineering design it is essential that the design steps and choices can be traced and verified for future use rather than exactly reproduced (Eekels & Roozenburg, 1991). The main quality requirement for design, therefore, is how the outcomes fulfill the needs as reflected in the brief of requirements.

Solutions not considered in our study include, for example, application of air scrubbers or separation of liquid manure into liquid and solid fraction. These solutions were excluded due to possible pollution swapping. Air scrubbers were found to increase fossil energy use and may increase N_2O emission and water use, but as an end-of-pipe technique also does not have a positive effect on indoor air quality, because NH_3 (and odor) are already produced (De Vries & Melse, 2013). Separation of liquid manure increased the risk of greenhouse gases, mainly from CH_4 and N_2O production and volatilization from the storage of the different fractions and requires energy for separation (Dinuccio et al., 2008). These technical solutions are applied in current practice, but are not preferable from an environmental point of view.

5.5 CONCLUSION

By using an adapted approach to engineering design, we were able to successfully design strategies for integrated pig and dairy cattle manure management with the aim to reduce environmental impact. The design approach revealed to be a valuable tool to structure the process and address all emission processes and resource use, functions of the manure management system, principle-options for the functions, and technical solutions needed. By this means, we were able to identify possible pollution swapping and create strategies to prevent it. The method provides a structural basis for design of biosystems and can be extended to other research areas.

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Chapter 6

// INTEGRATED MANURE MANAGEMENT TO REDUCE ENVIRONMENTAL IMPACT: II. ENVIRONMENTAL IMPACT ASSESSMENT OF STRATEGIES

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ABSTRACT

Manure management contributes to adverse environmental impacts through losses of nitrogen (N), phosphorus, and carbon (C). In this study, we aimed to assess the potential of newly designed strategies for integrated manure management (IS) to reduce environmental impact. An important aspect of the strategies was preventing pollution swapping. Life cycle assessment was used to compute climate change (CC), fossil fuel depletion (FFD), terrestrial acidification (TA), marine eutrophication (ME), particulate matter formation (PMF), N use efficiency (NUE), and phosphorus over application rate (POA), relative to the crop demand for N. We used a North West European reference (Ref) and the Dutch current situation of manure management (NL) to illustrate the potential of the IS to reduce environmental impact. Manure management in Ref included production and management of liquid and solid dairy cattle manure applied to maize and grass, and liquid pig manure applied to wheat. A Monte Carlo uncertainty simulation was done to assess the effect of variation in N and C losses and N uptake by crops on the comparison with Ref, IS, and NL. Results showed that the IS reduced all environmental impacts in all manure product and crop combinations and more than doubled the NUE (69% compared to maximum 33%). Main causes were: segregation of pig and dairy cattle urine and feces inside the housing system reduced methane (CH_4) and ammonia (NH_3) emissions; addition of zeolite to solid dairy cattle manure reduced NH_3 emission, sealed storages in all IS reduced volatilization of N and C; bio-energy production from the feces reduced the production of fossil electricity and heat; and finally N emissions in the field were reduced by ammonia emission reducing application techniques and improved application management (tillage, field traffic en synchronization of manure product application with crop demand). Compared with the Ref, NL had lower TA, PMF, POA, and higher NUE, except for solid cattle manure applied to grass. This result indicates that the Dutch regulations to reduce NH_3 emissions were successful, but that CC can be improved. Compared with NW EU practice, IS reduced environmental impact up to 176% for CC, up to >700% for FFD, up to 92% for TA, up to 98% for ME, up to 95% for PMF, up to 103% for POA and more than doubled the NUE. We concluded that the designed IS avoid pollution swapping in the entire manure chain.

6.1 INTRODUCTION

In livestock production, management of animal manure leads to major losses of nitrogen (N), phosphorus (P), and carbon (C). Manure management includes collection inside the housing system, storage (inside and outside), processing, and field application. In the European Union (EU), about 149 Mtons of liquid pig manure, 448 Mtons of liquid cattle manure and 295 Mtons of solid cattle manure are produced (Henning Lyngsø et al., 2011), of which the Netherlands contributes considerably with 7% of the liquid pig manure, 10% of the liquid cattle manure. The production of solid manure is low with 0.04% of the EU production (CBS, 2011). Production of solid cattle manure in the Netherlands is currently increasing, as a result of initiatives to improve animal welfare. In the EU, only about half of the nitrogen (N) and 70% of the phosphorus (P) excreted by animals is recycled as crop nutrient (Bouwman et al., 2009; Oenema et al., 2007), the rest is lost to the environment causing adverse environmental impacts, such as: climate change, terrestrial acidification, and marine eutrophication. To reduce environmental impacts, European directives, such as the Nitrates Directive (91/676/EEC), the National Emission Ceilings (NEC) Directive (2001/81/EC) and the Water framework Directive (200/60/EC) were implemented in order to reduce emissions from all Member States (EC, 2012; EU, 2013).

To reduce losses of N, P, and C and, therefore, improve the efficiency of using N and P from manure, different strategies have been proposed (Burton & Turner, 2003; Sommer & Hutchings, 2001). Most strategies, however, focus on a single part of the manure management system, such as reducing ammonia (NH_3) emission from outside manure storage by covering, or reducing NH_3 emission from manure application by injection instead of broadcast spreading. Such single-issue strategies often caused reduction of one type of emission while increasing another type of emission, a phenomenon referred to as ‘pollution swapping’. With a structured design approach we formulated strategies for integrated manure management of pig and cattle manure to reduce environmental impact throughout the manure management system (Chapter 5). With these strategies we aimed to reduce emissions of N, P and C and use of fossil energy along the manure management system, or in other words, prevent pollution swapping. As a validation, a quantitative assessment of the potential of these strategies to reduce the environmental impact is required. Life cycle assessment is (LCA) is a generally accepted method to quantify the environmental impact along the life cycle of a product (ISO-14040, 2006).

The aim of this study was to assess the potential of the newly designed strategies for integrated pig and dairy cattle manure management, as designed by De Vries et al. (Chapter 5), to reduce environmental impact. We quantified the environmental impact, the N use efficiency (NUE), and P_2O_5 over application rate along the manure management system and demonstrated the potential to reduce environmental impact for the case of North Western Europe and the Netherlands.

6.2 MATERIALS AND METHODS

6.2.1 LCA approach
In this study, we considered the changes in environmental impact of the strategies, or in other words, performed a consequential LCA (Finnveden et al., 2009). We therefore included all environmental impacts from processes that were affected by changes in the manure management system (Weidema et al., 2009).

6.2.2 Manure management and system boundaries
The manure management system included the manure storage in the animal house, outside manure storage, manure processing, transport, and field application of manure, soil tillage and in-field traffic, and crop uptake of N until harvest. External processes included production of mineral fertilizer and production of electricity, heat, and fuel (Fig. 6.1). Avoided mineral fertilizer production was included, because the nutrients in the manure products (N, P, and K) were considered to substitute nutrients from mineral N, P, and K fertilizers. Similarly, electricity and heat production were avoided with production of bio-energy. Animal production, crop management and transport were outside the system boundary, as they were assumed not to be affected by manure management strategies. Furthermore, emission from transport of manure was not considered, as we assumed the same distances to apply for all situations. Emissions associated with the production of capital goods, such as the installations for manure processing, were excluded from the calculations.

6.2.3 Unit for comparison
The main function of the manure management systems compared was to manage livestock excreta from the moment of excretion until field application as fertilizer. We, therefore, used a common unit of 1 ton excreted urine and feces, either mixed in liquid manure, or kept separate. The same chemical composition of excreta was used to ensure that the same amount of nutrients and dry matter entered each management system.

6.2.4 Definition of the NW EU reference, Dutch situation and strategies
We applied the IS to current North West European (NW EU) practice (Ref) and used the Dutch situation to represent current progressive manure management (NL). NW EU represents intensive livestock and manure production. In Ref and NL, liquid cattle manure was applied to grassland and arable land for production of silage maize, whereas liquid pig manure was used for wheat production. Solid dairy cattle manure was applied to grassland.

In Ref, liquid pig and cattle manure were produced in a housing system with slatted floors, and stored in-house for an average period of 4 - 6 months (Table 6.1) (Burton & Turner, 2003; De Vries et al., 2012a, Chapter 3). Liquid pig and cattle manure were stored in an outside storage tank, without cover, for an average period of 1 month. Subsequently, pig and cattle manure were applied to the field by broadcast spreading, without incorporation into the soil within 24 hours. Application to arable land occurred only once in spring and twice to grassland during the growing season (Menzi et al., 1998). Liquid manure

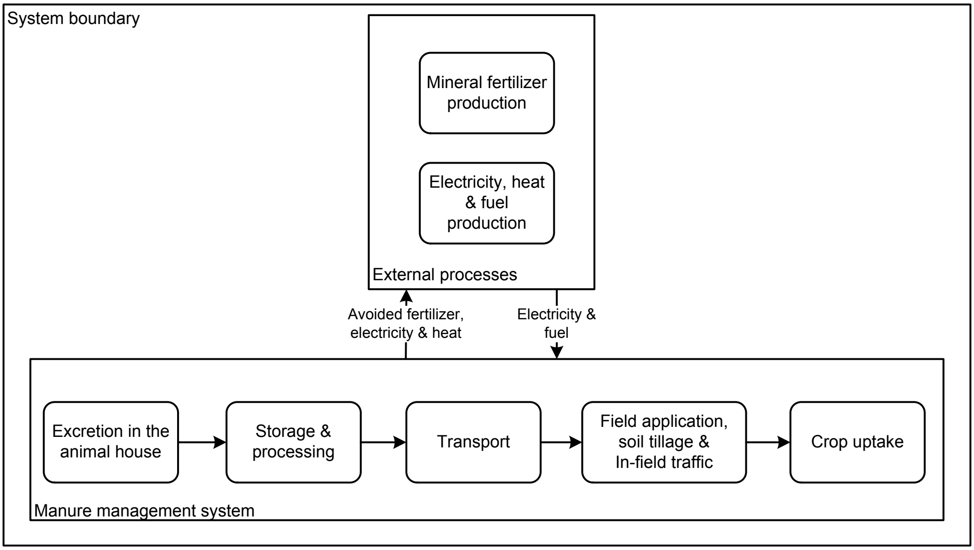

Fig. 6.1. Manure management system and external production processes that are included in the system boundary: electricity and fuel, and substituted processes: mineral fertilizer, electricity, and heat.

processing did not occur in Ref. Solid cattle manure in Ref was produced in tied stalls. After in-house storage for a few days, it was removed to an outside storage, where it was stock piled uncovered for an average period of 4 months and spread in spring using a solid manure spreader (Menzi et al., 1998). For tillage and field traffic, inversion tillage by plowing arable land was assumed, whereas random traffic on arable and grassland for all cropping systems was assumed.

In NL, liquid pig manure was produced in a housing system with partly slatted floors (60%) and stored in-house for an average period of 3 months. Subsequently, it was stored in a covered outside storage for an average period of 1 month (De Vries et al., 2012a, Chapter 3) and applied to arable land by deep injection (Van Bruggen et al., 2011). Application occurred once at the start of the growing season. Liquid cattle manure in NL was produced in a cubicle house with slatted floors and stored in-house for 4 months (Table 6.1). Subsequently, it was applied to grassland by shallow injection or to arable land by deep injection (Van Bruggen et al., 2011). Application on grassland occurred once prior to the growing season and three times during the growing season, to arable land once, at the start of the growing season (CBGV, 2012). Solid cattle manure in NL was produced in tied stalls. After in-house storage for a few days the manure was stockpiled and stored outside under a roof for an average period of 4 months. Application occurred by a solid manure spreader once prior to the growing season in spring (Ellen et al., 2007). For tillage and field traffic, inversion tillage on arable land and random traffic on arable and grassland was assumed for all systems.

Strategies for integrated manure management (IS) were designed and described by De Vries et al. (Chapter 5), and summarized in Table 6.1. Strategies included the separation of urine and feces for pigs and dairy cattle in the housing system, called segregation. Pig and dairy cattle urine were acidified, and feces were kept in sealed storages and applied by deep injection. Solid cattle manure was stored and mixed with zeolite and applied by broadcasting combined with rainfall or irrigation (Table 6.1).

6.2.5 Life cycle data inventory and assumptions

6.2.5.1 Chemical compositions of manure products

The chemical compositions of the manure products in Ref and NL were based on Menzi (2002) (Table 6.2). To calculate the compositions after excretion, numbers in Table 6.2 were corrected for storage emissions under NW EU conditions using a mass balance approach. The compositions of the urine and feces used in IS were based on De Haan et al. (2003) for cattle and on Aarnink et al. (2007) for pigs (Table 6.3). Because the compositions in IS and Ref were taken from different literature sources, we scaled the urine and feces compositions in IS to the liquid manure composition in Ref, based on the method used in De Vries et al. (2013) (Chapter 4). This ensured that the same amount of nutrients and dry matter entered IS and Ref in accordance with the FU. Following a mass balance, the compositions of all products (Table 6.2 and 6.3) were calculated for each respective stage in the manure management system using emission factors in Table 6.4.

6.2.5.2 Data for in-house management and manure storage

During in-house and outside storage of liquid and solid manure, and of urine and feces, emissions of NH_3, N_2O, NO, N_2, and CH_4 were considered. Emission factors in Ref were based on IPCC and EMEP Corinair guidelines (EMEP, 2009; IPCC, 2006b) (Table 6.4). Emission factors for NL were based on the national inventory data (Groenestein et al., 2012a; Van Bruggen et al., 2011). Emission factors for outside storage in IS were calculated based on the reduction percentages in Table 6.6. We assumed that sealing storages reduced emissions of NH_3, CH_4, and N_2O by 90% (Table 6.6). Emission of NH_3 for in-house management of solid cattle manure was reduced by adding zeolite (50 kg ton^{-1} manure) in IS (Shah et al., 2012a). Indirect emissions of N_2O from (de)nitrification of NH_3, NO_x, and NO_3^- in the environment were included; 1% of NH_3-N + NO_x-N (95% confidence interval (CI) of 0.2 - 5%) and 0.75% of NO_3-N (95% CI of 0.05 - 2.5%) (IPCC, 2006b).

We assumed that 2% of the N present in solid cattle manure leaches as NO_3^- during outside storage(Table 6.3) (EMEP, 2009).

Emissions of CH_4 from storage in Ref and NL were based on data from the Rains model (Klimont & Brink, 2004) and on Dutch data, mostly based on measurements (Groenestein et al., 2012b). Addition of zeolite slightly increased CH_4 emission from solid cattle manure storage (Shah, 2012).

Table 6.1. Overview of the Ref, NL and IS with their combinations of manure products, treatment, storage, application method and timing, and tillage and field trafficking method

	Management	Liquid cattle manure and grass	Solid cattle manure and Grass	Liquid cattle manure and maize	Liquid pig manure and wheat
Ref	Fertilizer product	liquid cattle manure	solid cattle manure	liquid cattle manure	liquid pig manure
	Treatment method	x	x	x	x
	Storage method	inside 5 months	outside 4 months	inside 5 months	inside 6 months
	Application method	broadcast spreading	spreading	broadcast spreading	broadcast spreading
	Application timing*	pgs/egs	pgs	pgs	pgs
	Tillage+traffic method	x + random traffic	x + random traffic	inversion till + random traffic	inversion till + random traffic
NL	Fertilizer product	liquid cattle manure	solid cattle manure	liquid cattle manure	liquid pig manure
	Treatment method	x	x	x	x
	Storage method	inside 4 months	outside-r 4 months	inside 4 months	Inside + outside-c 4 months
	Application method	shallow injection	spreading	deep injection	deep injection
	Application timing*	1egs+3gs	pgs	1pgs	1pgs
	Tillage+traffic method	x + random traffic	x + random traffic	inversion till + random traffic	inversion till + random traffic
IS	Fertilizer product	cattle urine + digested cattle feces	solid cattle manure	cattle urine + digested cattle	pig urine + digested pig feces
	Treatment method	acidification + digestion	zeolite	acidification + digestion	acidification + digestion
	Storage method	outside-s	outside-s	outside-s	outside-s
	Application method	deep injection in pockets (urine and feces)	spreading + water addition	deep injection in pockets (urine) deep injection (5 cm, feces)	deep injection in pockets (urine) deep injection (5 cm, feces)
	Application timing*	egs+5gs	pgs+egs	pgs+5gs	pgs+5gs
	Tillage+traffic method	x + controlled traffic	x + controlled traffic	non inversion till + controlled traffic	non inversion till + controlled traffic

x = no action; * numbers in cells indicate the number of times that the respective fertilizer product is applied at the specific time of the growing season.
Storage methods: inside = inside the animal house, outside-r = outside roofed, outside-c = outside covered, outside-s = outside sealed.
Application timing: gs = during growing season, pgs = prior to growing season/ planting, egs = early growing season/ just after planting.

Table 6.2. Chemical composition and excretion rates of manure in the Ref and NL and their respective management stage

Manure product after excretion	DM[a] kg ton^{-1}	OM kg ton^{-1}	Ash kg ton^{-1}	N_{tot} kg ton^{-1}	N_{min} kg ton^{-1}	N_{org} kg ton^{-1}	P_2O_5 kg ton^{-1}	K_2O kg ton^{-1}	Density[c] kg m^{-3}	kg ap^{-1} year[d]
Liquid cattle manure[b]	74.5	64.5	10.0	5.31	3.42	1.89	1.40	5.10	1005	26000
Solid cattle manure[b]	207	165	42.0	7.68	5.13	2.55	2.40	6.10	900	18000
Liquid pig manure[b]	82.3	68.3	14.0	6.92	5.37	1.56	2.10	3.20	1040	1200
Manure product after in-house storage										
Ref										
Liquid cattle manure	67.0	57.0	10.0	4.52	2.82[e]	1.70	1.40	5.10	1005	
Solid cattle manure	207	165	42.0	5.73	3.43[e]	2.30	2.40	6.10	900	
Liquid pig manure	52.3	38.3	14.0	5.33	3.93[e]	1.40	2.10	3.20	1040	
NL										
Liquid cattle manure	64.3	54.3	10.0	4.90	3.20[e]	1.70	1.40	5.10	1005	
Solid cattle manure	207	165	42.0	6.09	3.79[e]	2.30	2.40	6.10	900	
Liquid pig manure	57.9	43.9	14.0	5.39	3.99[e]	1.40	2.10	3.20	1040	
Manure product after outside storage										
Ref										
Liquid cattle manure	67.0	57.0	10.0	4.00	2.30	1.70	1.40	5.10	1005	
Solid cattle manure	207	165	42.0	5.20	1.40	3.80	2.40	6.10	900	
Liquid pig manure	52.0	38.0	14.0	4.80	3.40	1.40	2.10	3.20	1040	
NL										
Solid cattle manure	207	165	42.0	5.85	3.30[e]	2.55	2.40	6.10	900	
Liquid pig manure	57.6	43.6	14.0	5.28	3.88	1.40	2.10	3.20	1040	

[a] DM = dry matter, OM = organic matter, N_{tot} = total nitrogen, N_{min} = mineral nitrogen (NH_4^+-N), N_{org} = organic nitrogen, ap = animal place.
[b] Menzi (2002) corrected for emissions from in-house and outside storage for NW EU conditions.
[c] KWIN (2009-2010).
[d] CBS (2010) and Menzi et al. (1998) for solid cattle manure.
[e] During in-house storage 10% of the N_{org} was mineralized for all manure products. During outside storage 6.7% of N_{min} in solid cattle manure was immobilized (EMEP, 2009).

Table 6.3. Chemical composition and excretion rates of urine and feces and solid cattle manure in the IS and their respective management stage

Manure product after excretion	DM[a] kg ton^{-1}	OM kg ton^{-1}	Ash kg ton^{-1}	N_{tot} kg ton^{-1}	N_{min} kg ton^{-1}	N_{org} kg ton^{-1}	P_2O_5 kg ton^{-1}	K_2O kg ton^{-1}	Density kg m^{-3}	kg ap^{-1} year
Cattle urine[b]	15.7	12.1	3.58	4.78	4.47	0.31	0.23	5.60	1030	5727
Cattle feces[b]	117	103	14.7	5.69	2.66	3.04	2.25	4.73	1120	7874
Pig urine[c]	13.8	8.03	5.75	6.46	6.09	0.37	0.35	2.82	1010	538
Pig feces[c]	156	133	22.9	7.43	4.59	2.83	3.98	3.61	1120	500
Manure product after in-house management										
Cattle urine	15.7	12.1	3.58	4.51	4.20[e]	0.31	0.23	5.60	1030	
Cattle feces	117	103	14.7	5.63	2.59	3.04	2.25	4.73	1120	
Pig urine	13.8	8.03	5.75	5.62	5.26[e]	0.37	0.35	2.82	1010	
Pig feces	156	133	22.9	6.62	3.78	2.83	3.98	3.61	1120	
Solid cattle manure	207	165	42.0	7.30	5.00[e]	2.30	2.40	6.10	900	
Manure product after digestion and outside storage										
Cattle urine	15.7	12.1	3.58	4.43	4.15	0.28	0.23	5.60	1030	
Digested cattle feces	78.9	63.6[d]	15.3	5.77	3.25[d]	2.53	2.34	4.92	1077	
Pig urine	13.7	7.99	5.75	5.55	5.22	0.33	0.35	2.82	1010	
Digested pig feces	107	82.5[d]	24.1	6.88	4.50[d]	2.39	4.19	3.80	1065	
Solid cattle manure	207	165	42.0	7.17	4.53[e]	2.63	2.40	6.10	900	

[a] DM = dry matter, OM = organic matter, N_{tot} = total nitrogen, N_{min} = mineral nitrogen (NH_4^+-N), N_{org} = organic nitrogen, ap = animal place.
[b] De Haan et al. (2003) scaled to the liquid manure composition in the Ref.
[c] Aarnink et al. (2007) scaled to the liquid manure composition in the Ref.
[d] Assumed that 38% of the OM was decomposed during anaerobic digestion (Timmerman et al., 2005) and 20% of the N_{org} was mineralized to N_{min} (Ovinge, 2008; Schröder et al., 2008).
[e] During in-house storage 10% of the N_{org} was mineralized for urine and solid cattle manure. During outside storage 6.7% of N_{min} in solid cattle manure was immobilized (EMEP, 2009).

Table 6.4. Emission factors for in-house management and outside storage used in the Ref, the NL, and the IS

		N_2O-N % N		NO-N % N		N_2-N % N		NH_3-N % TAN		NO_3-N % N	CH_4 kg t^{-1}	
Scenario	Fertilizer product	in-house	Outside	in-house	outside	in-house	outside	in-house	outside	outside	in-house	outside
Ref	Liquid cattle manure	0.2[a]		0.2[c]		2[c]		20[d] (10-40)	20[d] (10-40)	-	4.02[e]	
	Solid cattle manure	2[a]		2[c]		10[c]		19[d] (10-38)	27[d] (14-54)	2[d]	0.17[e]	
	Liquid pig manure	0.2[a]		0.2[c]		2[c]		28[d] (14-56)	14[d] (7-28)	-	16.1[e]	0.17[g]
NL	Liquid cattle manure	0.1[b]		0.1[c]		1[c]		10[b]	-	-	5.45[f]	-
	Solid cattle manure	2[b]		2[c]		10[c]		10[b]	2% N[b]	2[d]	0.17[e]	
	Liquid pig manure	0.1[b]		0.1[c]		1[c]		27[b]	2% N[b]	-	13.1[f]	0.17[g]
IS	Cattle urine	0.2[a]	-	0.2[c]	-	2[c]	-	0.27[h] kg ton^{-1}	2.0[k]	-	0.09[i]	0.017[k]
	(Digested) cattle feces		0.02[k]		0.02[c]		0.2[c]			-		
	Pig urine	0.07[j] kg ton^{-1}	-	0.07[c] kg ton^{-1}	-	0.67[c] kg ton^{-1}	-	0.83[j] kg ton^{-1}	1.40[k]	-	0.91[j]	0.017[k]
	(Digested) pig feces		0.02[k]		0.02[c]		0.2[c]			-		
	Solid cattle manure	0.18[l]		0.18[c]		0.90[c]		2.47[m]	2.70[k]	2[d]	0.02[l]	

- = not included. [a] IPCC (2006b). 95% confidence interval (CI) N_2O emission: -50%, +100% (EMEP, 2009).
[b] Groenestein et al. (2012a). 95% CI N_2O emission: -50%, +100%; NH_3 emission: ±50% (EMEP, 2009).
[c] Calculated as a ratio of the N_2O-N emission (ratio $N_2O:NO:N_2$ = 1:1:10 for liquid manure and 1:1:5 for solid manure) (Oenema et al., 2000).
[d] EMEP (2009). 95% CI between brackets.
[e] Klimont and Brink (2004). 95% CI CH_4 emission: ±50% (IPCC, 2006a).
[f] Groenestein et al. (2012b) 95% CI CH_4 emission: ±50% (IPCC, 2006a).
[g] Modeled value based on De Mol and Hilhorst (2003) and De Vries et al. (2012a) (Chapter 3). 95% CI CH_4 emission: ±50% (IPCC, 2006a).
[h] In kg per ton of urine and feces. Based on a value of 8.6 kg NH_3 ap^{-1} year for conventional manure management (VROM, 2012) with 49% reduction (average from De Haan et al. (2003) and Swierstra et al. (2001)). 95% CI: 0.23 - 0.34 kg ton^{-1} (Huis in 't Veld & Scholtens, 1998; Swierstra et al., 2001).
[i] In kg per ton of urine and feces (Van Dooren & Smits, 2007).
[j] In kg per ton of urine and feces (Aarnink et al., 2007). 95% CI N_2O emission: 0.012 - 0.12 kg ton^{-1} (Aarnink & Ogink, 2007). 95% CI NH_3 emission: 0.79 - 2.10 kg ton^{-1} (Mosquera et al. (2010) and VROM (2012) in De Vries et al. (2013) (Chapter 4)).
[k] Calculated with 90% reduction from sealed storages (Table 6.6).
[l] Combined reduction calculated with 10% reduction from addition of zeolite and 90% reduction from sealed storages (Table 6.6).
[m] Calculated with 87% reduction from addition of zeolite (Table 6.6).

Table 6.5. Emission factors for field application and nitrogen fertilizer replacement value (NFRV) for the organic N fraction used for the Ref, the NL, and the IS

Scenario	Fertilizer product	Crop	N_2O-N % N	NO-N % N	N_2-N % N	NH_3-N % TAN	NO_3-N (%N)	NFRV N_{org} %
Ref	Liquid cattle manure	Grass	1[a]	0.1[b]	3[b]	55[c]	11[a]	71[d]
	Solid cattle manure	Grass	1[a]	0.1[b]	3[b]	79[c]	11[a]	71[d]
	Liquid cattle manure	Maize	1[a]	0.1[b]	3[b]	55[c]	30[a]	50[d]
	Liquid pig manure	Wheat	1[a]	0.1[b]	3[b]	40[c]	30[a]	38[d]
	Mineral fertilizer	Grass/ Arable	1[a]	0.1[b]	3[b]	2.0[c]	11/ 30[a]	-
NL	Liquid cattle manure	Grass	0.30[e]	0.03[b]	0.90[b]	19.0[f]	11[a]	71[d]
	Solid cattle manure	Grass	0.10[e]	0.01[b]	0.30[b]	74.0[f]	11[a]	71[d]
	Liquid cattle manure	Maize	1.30[e]	0.13[b]	3.90[b]	2.00[f]	30[a]	50[d]
	Liquid pig manure	Wheat	1.30[e]	0.13[b]	3.90[b]	2.00[f]	30[a]	38[d]
	Mineral fertilizer	Grass	0.80[e]	0.08[b]	2.40[b]	2.50[f]	11[a]	-
	Mineral fertilizer	Arable	0.70[e]	0.07[b]	2.10[b]	2.50[f]	30[a]	-
IS	Cattle urine	Grass	0.65	0.07[b]	1.95[b]	3.19[h]	11[a]	71[d]
	Digested cattle feces		0.36[g]	0.04[b]	1.09[b]	3.19[h]	11[a]	71[d]
	Solid cattle manure	Grass	0.65	0.07[b]	1.95[b]	4.42[h]	11[a]	71[d]
	Cattle urine	Maize	0.65	0.07[b]	1.95[b]	3.19[h]	30[a]	50[d]
	Digested cattle feces		0.36[g]	0.04[b]	1.09[b]	3.19[h]	30[a]	50[d]
	Pig urine	Wheat	0.65	0.07[b]	1.95[b]	2.32[h]	30[a]	38[d]
	Digested pig feces		0.36[g]	0.04[b]	1.09[b]	2.32[h]	30[a]	38[d]

- = not included.
[a] IPCC (2006b). 95% CI N_2O emission: 0.3 - 3%; NO_3^- emission: 10 - 80% (IPCC, 2006b). Grassland is 0.36 times arable land (Schröder et al., 2007).
[b] Calculated as a ratio of the N_2O-N emission (ratio N_2O:NO:N_2 = 1:0.1:3). Sim Den model in Hamelin et al. (2011).
[c] EMEP (2009). 95% CI for 55%: 28 - 7%5; for 79%: 40 - 85%; for 40%: 20 - 80%; and for 2%: ±20% (EMEP, 2009).
[d] Average of 72% and 96% of the N_{org} that is mineralized over time multiplied with the extent to which the pattern of mineralization of N_{org} coincides with the N- uptake pattern of a crop (0.85 for grass, 0.6 for maize, and 0.45 for wheat) (Greenwood et al., 1989; Schröder et al., 2008).
[e] Velthof and Mosquera (2010). Same uncertainty assumed as for the Ref.
[f] Van Bruggen et al. (2011). 95% CI NH_3 emission: ±60% for manure products and ±20% for mineral fertilizer (EMEP, 2009).
[g] Combined reduction calculated with 44% reduction from anaerobic digestion and 35% reduction from CTF (Table 6.6).
[h] Combined reduction for urine and feces calculated with 80% reduction from injection and 71% reduction from acidification (Table 6.6). Combined reduction for solid cattle manure calculated with 74% reduction from addition of zeolite and 79% reduction from irrigation or water additions (Table 6.6).

Table 6.6. Reduction percentages assumed for calculating the emission factors used in the IS

Technique/ management	NH_3 reduction	N_2O reduction	NO_3^- reduction	CH_4 reduction	NFRV increase	Energy reduction
Storage						
Acidification	70%[a]	-	-	77%[b]		-
Anaerobic digestion	26%[c]	44%[d]	-	90%[e]		Net production
Zeolite addition	83%[f]	10%[f]	-	+16%[f]		-
Sealed storage	90%[e]	90%[e]	-	90%[e]		-
Field application					Depends on NH_3 emission	
Deep injection	80%[g]	-	-	-		-
Acidification	71%[h]	-	-	-		-
Zeolite + irrigation	74%[i] + 79%[j]	-	-	-		-
Controlled traffic	-	35%[k]	-	factor 5-20[k]		50%[l]
Non-inversion tillage	-	-	-	-		42%[m]
Precision timing	-	-	50%[n]	-	5%[n]	-

- = not included, NFRV = N fertilizer replacement value.
[a] Kai et al. (2008).
[b] Average of 67 and 87% (also the assumed 95% CI) reduction compared to non-acidified liquid manure (Petersen et al., 2012).
[c] Average of 6 and 45% (also the assumed 95% CI) (VanderZaag et al., 2011).
[d] Average of 17 and 71% (also the assumed 95% CI) (VanderZaag et al., 2011).
[e] Assumed reduction due to sealed storage. 80% for covered and 100% reduction for storage bags (the assumed 95% CI: 80 - 100%) (EMEP, 2009).
[f] Shah et al. (2012). 95% CI: 0 - 20%.
[g] EMEP (2009). Assumed 95% CI: ±10%.
[h] Average of 85, 72, and 55% compared to non-acidified manure from surface spreading (assumed 95% CI: 55 - 85%) (Bussink et al., 1994; VanderZaag et al., 2011).
[i] Reduction from addition of zeolite. 95% CI: 60 - 88% (Shah et al., 2012a).
[j] Average of 65 and 92% (also the assumed 95% CI) (Shah et al., 2012b).
[k] Average of 20 - 50% (also the assumed 95% CI) reduction in N_2O emission and a factor of 5-20 increase in soil CH_4 oxidation (Vermeulen & Mosquera, 2009).
[l] Energy reduction with controlled traffic farming (not included) (Lamers et al., 1986).
[m] Based on Knight (2004) and Morris et al. (2010).
[n] VanderZaag et al. (2011). 5% based on this reduction.

6.2.5.3 Data for digestion, acidification and zeolite addition

In IS, cattle and pig feces were anaerobically digested to reduce CH_4 emission and fossil fuel use by production of biogas, a mixture of CH_4 and CO_2. We assumed a continuous stirred tank reactor (CSTR) operating at mesophilic temperature (ca. 35 °C) with a hydraulic retention time of 60 days. Respective CH_4 yields for anaerobic digestion of segregated cattle and pig feces were assumed to be 25 and 37.5 m^3 CH_4 ton^{-1} fresh matter, respectively (Timmerman et al., 2009; Van Dooren, 2010 Unpublished data). The biogas was used in a combined heat and power unit (CHP) to generate electricity and heat. The energetic efficiency of the CHP was 80% and the electric efficiency was 35%. Produced electricity was assumed to avoid electricity on the electricity grid, whereas 50% of the remaining heat after use for the digestion process was assumed to avoid fossil-based heat (De Vries et al., 2012b, Chapter 2). The energy required for anaerobic digestion was 66 MJ electricity per ton of substrate and 166 MJ heat per ton of substrate (Berglund & Börjesson, 2006).

During anaerobic digestion, 1.5 % of the CH_4 produced was assumed to emit from the installation (1%) and the gas engine (0.5%) (De Vries et al., 2012a, Chapter 3).

In IS, urine was acidified during storage and digested feces were acidified just before application with sulfuric acid (H_2SO_4). Production impact of H_2SO_4 was excluded as this was negligible for the end result (Wesnæs et al., 2009). Environmental impact from producing zeolite, added to solid cattle manure, was assumed equal to the impact of stone meal production (EcoinventCentre, 2007).

6.2.5.4 Data for field application and soil tillage

During and after field application of liquid manure, solid cattle manure, urine and feces, emissions of N_2O, NH_3, NO, N_2, and NO_3^- were considered. Changes in CH_4 emission from field traffic were excluded, as the contribution of CH_4 emission from the field to overall impact of climate change from manure management is negligible (Vermeulen & Mosquera, 2009). Emissions in Ref were based on IPCC and EMEP Corinair guidelines, whereas for NL emissions were based on specific national values (Table 6.5). Emission factors of IS were based on Ref values and reductions in Table 6.6. In the situation of combined reductions, a multiplication of reduction factors was applied. For example, the N_2O emission after application of digested cattle feces were reduced due to anaerobic digestion (less available C) and CTF (increased soil aeration) as follows: 1% ´ (1-44%) ´ (1-35%) (Table 6.6).

Leaching and run-off of NO_3^- in Ref and IS was computed as a fraction of the applied N (IPCC, 2006b) (Table 6.5). The applied N was corrected for NH_3 and N_2O emission prior to multiplying with the leaching fraction (Dekker et al., 2009; Langevin et al., 2010). The leaching fraction for grassland was assumed 0.36 of the fraction on arable land (Schröder et al., 2007). The N uptake by each crop was computed using the apparent N recovery (ANR) of that respective crop. The ANR of grass was 75% (range of 60 - 90%), of wheat 70% (range of 60 - 80%), and that of maize 50% (range of 40 - 60%) (Greenwood et al., 1989; Schröder et al., 1998; Ten Berge et al., 2007). The ranges were assumed to be the 95% CIs. We included similar ANRs in IS and in Ref, implying that when more N was available from manure products, more mineral N fertilizer was avoided, meaning an environmental gain.

The amount of mineral N fertilizer avoided by using the N from manure products was calculated by using the N fertilizer replacement value (NFRV). The NFRV of the N_{min} fraction in the manure products was calculated by subtracting the NH_3 emission from the N_{min} fraction. NFRV of the N_{org} fraction in the manure products was calculated by multiplication with the percentages given in Table 6.5. In the IS an increase of the NFRV was assumed as a result of improved synchronization of N application and crop N uptake (Table 6.6). This increase was not taken into account for solid cattle manure because synchronization was similar as in Ref. The fertilizer replacement values for P and K were assumed to be 100%.

In IS, energy use was reduced with 42% by non-inversion tillage (Table 6.6). Energy reduction from controlled field traffic was assumed to be counteracted by increased trafficking from applying urine and feces more often during the growing season.

6.2.5.5 Data for external processes and environmental impact assessment

Emissions and environmental impacts from external processes (Fig. 6.1) were based on the Ecoinvent database (EcoinventCentre, 2007). Mineral fertilizer used, was assumed to be calcium ammonium nitrate for N, triple super phosphate for P, and potassium chloride for K (De Vries et al., 2012a, Chapter 3). The production mix for electricity and heat was based on the Dutch situation; 28% coal-based, 67% natural gas-based, and 5% wind-based electricity; and 79% natural gas-based and 21% light fuel oil-based heat (De Vries et al., 2012a, Chapter 3).

The environmental impact assessment was conducted based on the ReCiPe v.1.04 midpoint impact assessment method (Goedkoop et al., 2009). We considered the following impact categories: climate change (CC) in kg CO_2-eq, including: CO_2, CH_4, and N_2O. Biogenic C emission was excluded from the calculation as it represents C that is taken up by crops earlier (IPCC, 1997). Similarly, C-sequestration was excluded due to its small contribution; terrestrial acidification (TA) in kg SO_2-eq, including: NH_3, NO_x, and SO_2; marine eutrophication (ME) in kg N-eq, including: NH_3, NO_x, and NO_3^-; particulate matter formation (PMF) in kg PM_{10}-eq, including: NH_3, NO_x, and SO_2 as precursors of particulate matter; and fossil fuel depletion (FFD), in kg oil-eq. We calculated NUE as the ratio of N taken up by the crop to the N excreted by the animal, and the P_2O_5 over application rate (POA). The POA represents the amount of P_2O_5 over applied relative to the demanded N, i.e. representing the fittingness of the manure product to the demand of the crop and the addition to the soil P surplus. We computed this by comparing the N/ P_2O_5 ratio of the manure product with the N/ P_2O_5 ratio demanded by the crop. The N/ P_2O_5 ratio demand was 3.7 for cut grassland, 3.3 for maize, and 2.6 for wheat (Schröder, 2005).

6.2.5.6 Uncertainty analysis

A Monte Carlo uncertainty analysis was conducted to assess the effect of variation in emission factors and crop-N uptake on final results and comparisons between Ref, NL and IS. We included the following parameters and defined their 95% CI and distribution functions: N_2O (direct and indirect, and the related NO and N_2 emissions), NH_3, CH_4, NO_3^-, and the ANR (Table 6.4 and 5). Uncertainty in the environmental impact of mineral fertilizer production was included as a result from varying N emissions. Uncertainty in electricity production was excluded.

In IS, emission factors were calculated by using the factors from Ref and additionally applying a 95% CI and distribution function to the reduction percentages (Table 6.6). The distribution function of N_2O was assumed to be lognormal, the others were assumed to be normally distributed (De Vries et al., 2013, Chapter 4; Payraudeau et al., 2007). We ran 1000 Monte Carlo simulations to compute the variation in all final environmental impact categories, the NUE, and POA. Differences between Ref, NL, and IS were indicated for their significance with a two-sided t-test.

6.3 RESULTS

6.3.1 IS applied to Ref

IS reduced the CC impact in all manure product and crop combinations (Fig. 6.2 and Table 6.7). The net CC impact of IS ranged from -43.2 to -77.4 kg CO_2-eq, whereas the net impact of Ref ranged from 71.6 to 417 kg CO_2-eq. Differences between IS and Ref had three main causes: 1. segregation of urine and feces reduced CH_4 emission from in-house management, without increasing other greenhouse gas emissions (7.02 to 58.7 vs. 81.5 to 417 kg CO_2-eq for in-house management in Ref; Supplementary information, Table SI 1 and 2 include all detailed emissions and impacts); 2. bio-energy production from feces reduced the production of fossil electricity and heat, and related GHG emissions, as shown in the outside storage/ processing phase of the manure chain in Fig. 6.2 (132 MJ electricity and 77 MJ heat avoided with digested cattle feces and 180 MJ electricity and 110 MJ heat avoided with pig feces); and 3. reduced N emissions and improved management resulted in more avoided mineral N fertilizer application and production (-71.1 to -96.2 vs. Ref -33.5 to -50.6 kg CO_2-eq). CC impact change from non-inversion tillage instead of conventional tillage was small; maximum of 0.61 kg CO_2-eq.

IS reduced FFD in all manure product and crop combinations (Fig. 6.2 and Table 6.7). The net FFD impact of IS ranged from -347 to -1040 MJ, whereas the net impact of Ref ranged from -89 to -185 MJ. Differences between Ref and IS had two main causes: 1. reduced N emissions resulted in more avoided mineral N fertilizer application and production (336 to 474 vs. 188 to 287 MJ in Ref, Supplementary information, Table SI 1 and 2). 2. bio-energy production from the feces fraction reduced the production of fossil electricity and heat (508 to 710 MJ). FFD for outside storage and processing in the IS with solid cattle manure and grass, slightly increased (38.0 MJ) from the production of zeolite. FFD change from applying non-inversion tillage in the IS, instead of conventional tillage in Ref, was small (at the most 18 MJ).

IS reduced the TA impact in all manure product and crop combinations (Fig. 6.2 and Table 6.7). The net TA impact of IS ranged from 0.69 to 2.40 kg SO_2-eq, whereas the net impact of Ref ranged from 7.22 to 8.75 kg SO_2-eq. The differences between Ref and IS had three main causes: 1. segregation of urine and feces reduced NH_3 emission from housing (0.80 to 2.54 vs. 2.05 to 4.49 kg SO_2-eq in the housing system of Ref, Supplementary information, Table SI 1 and 2); 2. sealed outside storages reduced the emission of NH_3; and 3. application of urine and feces with acidification and injection reduced NH_3 emission from the field (0.33 to 0.34 vs. 3.77 to 4.05 kg SO_2-eq in Ref). TA reduction in IS with solid cattle manure was caused by reduced NH_3 emission from zeolite addition during storage (0.79 kg SO_2-eq for in-house and outside storage vs. 5.84 kg SO_2-eq for the Ref).

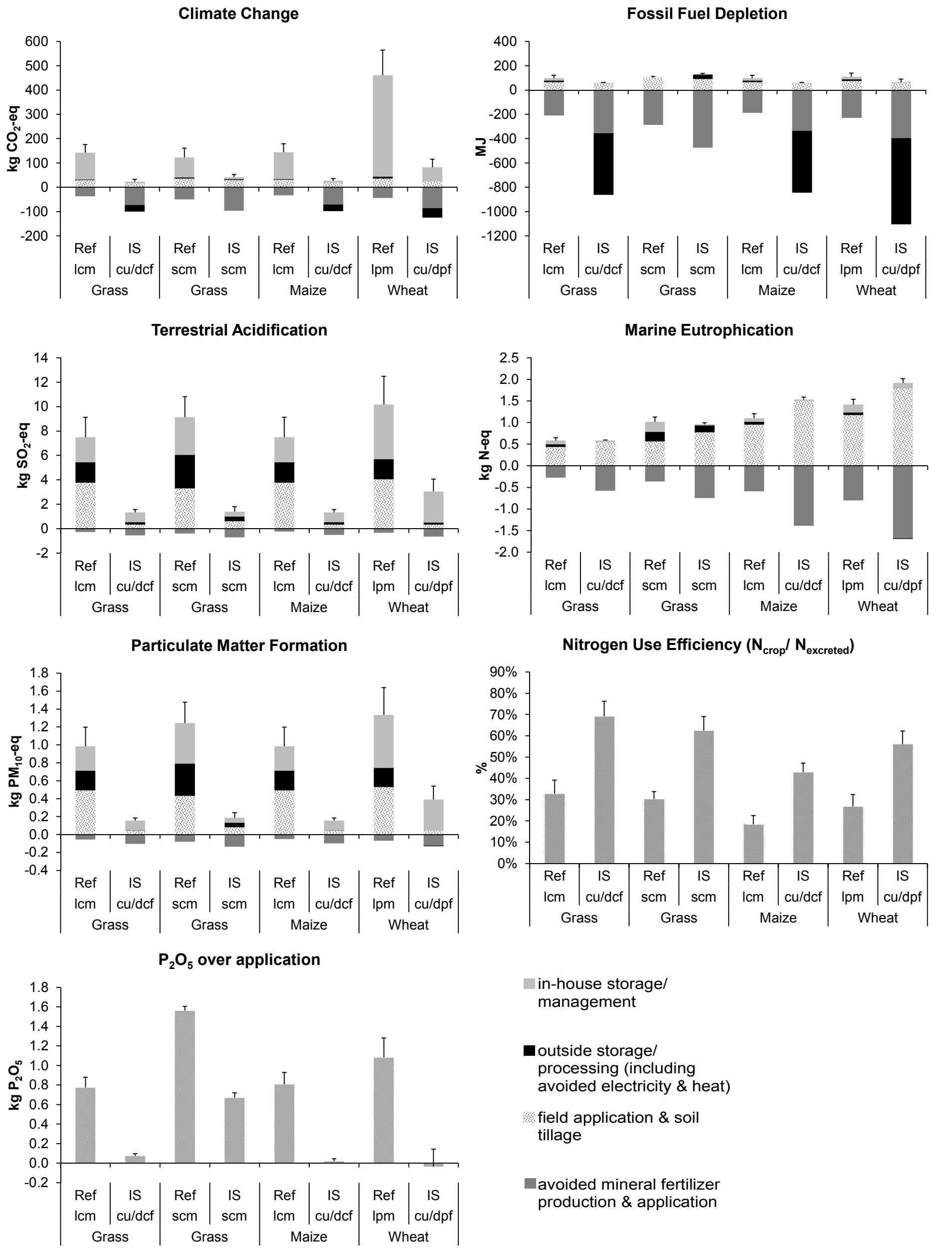

Fig. 6.2. Environmental impact assessment expressed per ton of excreted urine and feces for the Ref and IS with their respective manure types: liquid cattle manure (lcm), solid cattle manure (scm), liquid pig manure (lpm), cattle and pig urine (cu and pu) and digested cattle and pig feces (dcf and dpf), and crops. Error bars represent +SD of the net total. Negative contributions indicate a net reduction in impact.

IS reduced the ME impact in all manure product and crop combinations (Fig. 6.2 and Table 6.7). The net ME impact of IS ranged from 0.01 to 0.22 kg N-eq, whereas the net impact of Ref ranged from 0.31 to 0.65 kg N-eq. The difference between Ref and IS was a result of reduced N emissions and improved management in the manure management system (e.g. NH_3 and NO_x) and consequently more avoided mineral N fertilizer application and leaching of NO_3^-(0.55 to 1.78 vs. 0.44 to 1.17 kg N-eq in the Ref from avoided fertilizer application).

IS reduced the PMF impact in all manure product and crop combinations (Fig. 6.2). The net PMF impact of IS ranged from 0.05 to 0.26 kg PM_{10}-eq, whereas the net impact of Ref ranged from 0.93 to 1.27 kg PM_{10}-eq. Differences between Ref and IS were caused by similar factors as the ones that reduced TA, as NH_3 was the main contributor to this impact category.

IS increased NUE and reduced POA in all manure product and crop combinations (Fig. 6.2 and Table 6.7). The NUE of IS ranged from 43 to 69%, whereas the NUE of Ref ranged from 18 to 33%. POA in IS ranged from -0.03 to 0.67 kg P_2O_5 and in Ref from 0.77 to 1.56 kg P_2O_5. NUE in IS was more than doubled compared to Ref in all manure product and crop combinations. This increase for IS was mainly caused by reduced N emissions and improved management. The POA decreased in IS compared to Ref, also as a result of more retained N in the manure products, i.e. increasing the N/ P_2O_5 ratio of the manure products.

6.3.2 NL compared to Ref

NL had a net CC impact of 57.4 to 322 kg CO_2-eq with the highest impact from liquid pig manure combined with wheat (Fig. 6.2 and Table 6.7). CC impacts did not differ between NL and Ref.

NL had a net FFD impact of -159 to -220 MJ, with the highest impact from solid cattle manure and grass (Fig. 6.2 and Table 6.7). FFD impact for liquid pig and cattle manure was lower in NL than the Ref, mainly because more avoided impact from production of mineral N fertilizer (-301 to -334 MJ for NL vs. max. -287 MJ in Ref). This difference between Ref and NL was due to less N emission after application in NL.

NL had a net TA impact of 0.71 to 8.93 kg SO_2-eq, with the highest impact from solid cattle manure and grass (Fig. 6.2 and Table 6.7). The lower impact for all other manure and crop combinations in NL was mainly due to lower NH_3 emission during in-house and outside storage and field application. In NL, injection of liquid manure is obliged by either shallow or deep injection (0.20 to 1.81 kg SO_2-eq in NL vs. 3.77 to 4.05 kg SO_2-eq in Ref, Supplementary information, Table SI 1 and 2). In NL, covering of outside storages for liquid pig manure is obliged. This further reduces the risk of NH_3 emission during storage compared to NW EU conditions where covering is not always obligatory. The TA impact for solid cattle manure and grass was similar in NL compared to Ref (8.93 vs. 8.75 kg SO_2-eq), but higher emissions of NH_3 occurred in NL during field application, whereas lower NH_3 emission occurred during in-house and outside storage (Supplementary information, Table SI 1 and 2).

NL had a net ME impact of 0.12 to 0.63 kg N-eq, with the highest impact for solid cattle manure and grass (Fig. 6.2 and Table 6.7).

NL had a net PMF impact of 0.07 to 1.19 kg PM_{10}-eq, with the highest impact for solid cattle manure and grass (Fig. 6.2 and Table 6.7). The lower impact for all manure and crop combinations in NL compared to Ref was mainly due to lower NH_3 emission as in the TA impact category.

Table 6.7. Results of the environmental impact assessment of the Ref, IS, and NL

Crop	Fertilizer product		CC[a]	FFD	TA	ME	PMF	POA	NUE
			kg CO_2-eq	MJ	kg SO_2-eq	kg N-eq	kg PM_{10}-eq	kg P_2O_5	%
Grass	Liquid cattle manure	Ref	105	-110	7.22	0.31	0.93	0.77	33%
	Cattle urine and digested feces	IS	-77.4^{***z}	-805^{***z}	0.78^{***y}	0.01^{***y}	0.05^{***y}	0.08^{***z}	$69\%^{***}$
	Liquid cattle manure	NL	100	-205^{***}	2.35^{**}	0.12^{**}	0.28^{**}	0.34^{***}	$56\%^{**}$
Grass	Solid cattle manure	Ref	71.6	-185	8.75	0.65	1.16	1.56	30%
	Solid cattle manure	IS	-54.7^{**y}	-347^{***z}	0.69^{***y}	0.22^{**z}	0.05^{***y}	0.67^{***z}	$63\%^{***y}$
	Solid cattle manure	NL	57.4	-159	8.93	0.63	1.19	1.65	27%
Maize	Liquid cattle manure	Ref	110	-89.0	7.25	0.51	0.93	0.81	18%
	Cattle urine and digested feces	IS	-72.3^{***z}	-786^{***z}	0.82^{***}	0.15^{**}	0.06^{***}	0.02^{***y}	$43\%^{***}$
	Liquid cattle manure	NL	121	-205^{***}	0.71^{***}	0.24^{**}	0.07^{***}	0.15^{***}	$39\%^{***}$
Wheat	Liquid pig manure	Ref	417	-121	9.85	0.61	1.27	1.08	27%
	Pig urine and digested feces	IS	-43.2^{***y}	-1040^{***z}	2.40^{**}	0.22^{**}	0.26^{**}	-0.03^{***}	$56\%^{**}$
	Liquid pig manure	NL	322	-220^{**}	4.31^{**}	0.38	0.54^{**}	0.38^{**}	$45\%^{**}$

[a] Ref = NW EU reference, NL = Dutch state of art, IS = strategies for integrated manure management, CC = climate change, FFD = fossil fuel depletion, TA = terrestrial acidification, ME = marine eutrophication, PMF = particulate matter formation, POA = P_2O_5 over application, NUE = nitrogen use efficiency.
Indication of significance: IS and NL compared to Ref, $^{*}P < 0.05$, $^{**}P < 0.01$, $^{***}P < 0.001$; IS compared to NL, $^{x}P < 0.05$, $^{y}P < 0.01$, $^{z}P < 0.001$.

NL had a NUE of 27 to 56% and a POA of 0.07 to 1.65 kg P_2O_5 (Fig. 6.2 and Table 6.7). In NL, NUE was higher compared to Ref in most manure and crop combinations due to less N losses as indicated earlier. The POA of NL was lower compared to Ref also showing the lower N losses in the manure management system.

6.3.3 Uncertainty analysis
Results of the uncertainty analysis are presented in Fig. 6.2 and Table 6.7, and additionally in Table SI 3 and 4 in the supplementary information. For all manure product and crop combinations, environmental impacts in IS were lower compared to the Ref. For CC, FFD, and some other impacts (e.g. TA, PMF, and POA with liquid and solid cattle manure and grass), IS also reduced the impact compared to NL. Compared to Ref, NL impacts did not differ for CC and FFD. All other impacts for NL were lower, except with solid cattle manure and grass.

The coefficient of variation for CC in Ref (CV, i.e. the SD divided by the mean; Table SI 3) ranged from 24 - 45%. Uncertainty was highest with solid cattle manure and grass as a result of higher N_2O emission that has a relative higher variation. The CV for CC in NL ranged from 28 - 57%. Similar to Ref, uncertainty in NL was highest with solid cattle manure and grass. The CV for CC in IS ranged from 13 - 74% with the greatest uncertainty with pig urine, digested pig feces and wheat, as a result of higher CH_4 emission.

Uncertainty in FFD was related mainly to the varying amount of avoided mineral fertilizer. Hence, variation for FFD was relatively small. The CV for FFD in Ref ranged from 5 - 25%. CVs for NL and IS ranged from 3 - 27% and 1 - 3%, respectively.

The CV for TA in Ref ranged from 19 - 24%. CV for NL ranged from 25 - 37% and of IS from 30 - 61%. In the IS uncertainty was higher as a result of the additional variation in the reduction factors applied to calculate the emission factors (Section 6.2.5 and Table 6.6). The CV for PMF in IS ranged from 55 - 108% being larger than the CV of the Ref that ranged from 20 - 24%.

The CV for ME in Ref, IS, and NL were similar and ranged from 14 - 184%. The CVs for NUE and POA ranged from 10 - 30% and 3 - 1302%, respectively. The CV for POA in IS had the greatest uncertainty with pig urine, digested pig feces and wheat resulting from greater variation in N emissions.

6.4 DISCUSSION

6.4.1 Environmental impacts of Ref, IS and NL
Compared to Ref, IS had lower environmental impacts for all impact categories, and consequently a higher NUE. Moreover, IS yielded bio-energy as a valuable byproduct. The outcomes, show that pollution swapping was avoided in IS as intended. As a result, in most manure product and crop combinations of IS, NUE more than doubled (up to 69%), implying, that the double amount of excreted N was recycled as crop nutrient. Hence, the fraction of N emitted to the environment in IS, i.e. 31 - 57% was reduced compared to current practice, i.e. 67 - 82%. Our values in Ref are within the range of results in the

literature, e.g. Oenema et al. (2007) that reported N loss ranging from 48 - 62% of excreted N (in our case corresponding with 56 - 63% without crop uptake).

Compared to the set requirements in Chapter 5, the potential of the environmental impact reduction of IS was mostly achieved (reduction goal of 100% for CC and FFD, and 75% for the other impacts). Compared to Ref, the reduction potential of IS was up to 176% for CC, up to 92% for TA, up to 95% for PMF, up to >700% for FFD, up to 98% for ME, and up to 103% for POA. Few goals were not achieved, with solid cattle manure and grass for FFD (reduction of 88%) and POA (reduction of 57%), with cattle urine and feces and maize for ME (reduction 71%), and with pig urine and feces and wheat for ME (reduction of 63%). Results, however, were significantly lower than in Ref.

Our study supported the outcomes that adding zeolite to solid cattle manure reduced on-farm N losses (Shah et al., 2013). Shah et al. (2013), however, included on farm emissions and processes only, and, therefore excluded external processes, such as mineral fertilizer production and zeolite production. Our study, therefore, provides a broadened overview of the environmental potential of this solid manure management strategy.

As a result of reduced N losses, IS also had lower POA. These lower rates indicate that less P is provided to soil and crop per unit of N applied. Hence, the P-surplus of the soil is reduced, implying reduced leaching or run-off (Schröder et al., 2011).

Reduced N losses lead to more available N per unit of area in the whole agricultural system, because it is not emitted in another stage of manure management. More available N for application, however, may be a disadvantage in intensive livestock production systems with strict N application limits per unit of area. This ‘excess N’ then requires to be exported. Detailed effects of changing transportation of manure products were considered in another study (De Vries et al., 2012a, Chapter 3).

In NL, TA, PMF and POA were lower compared to Ref in all liquid manure product and crop combinations. On the one hand, this indicates that current policy, aimed at strategies to reduce NH_3 emissions have indeed reduced environmental impacts. On the other hand, where the Kyoto protocol sets a goal to reduce national CC, no policy to reduce emission of greenhouse gases was directed to Dutch agricultural practice. This shows that in NL CC was similar as in Ref for all manure product and crop combinations, implying possibilities for improvement. This is more relevant for liquid cattle manure and grass, as solid cattle manure is a minor product in NL (CBS, 2011). It should be noted that we used the same manure composition for reasons of comparability. But within Europe, compositions of manure vary and with it the dry matter content (Menzi, 2002). In NL, the dry matter content of manure is often higher compared to other countries. A higher dry matter content can lead to higher CH_4 and NH_3 losses. This is mainly relevant when computing absolute emissions for, e.g. national inventories.

6.4.2 Data and assumptions

In this study, we focused on the whole manure management system, from excretion by the animal until crop uptake, including field application and soil tillage (Fig. 6.1). Here, we considered a single crop only, not including the crop rotation. The rotation in which crops are produced largely affects the management of manure products, losses of N, P, and C and finally the environmental impact (e.g. inclusion of legumes or catch crops) (Nemecek et al., 2008). Altering the type of crops in rotation, for example, changes the N/ P_2O_5 ratio demanded by the crop and, therefore, also requires other (combinations of) manure products (Schröder, 2005). As a consequence, the amount of N and P applied will vary and requires adjustment for each succeeding crop in the rotation.

Emissions of N_2O result from (de)nitrification processes during storage and after field application. Production of N_2O, however, occurs via different pathways, including other groups of microbes, such as aerobic denitrification and nitrifier denitrifcation (e.g. Wrage et al., (2001). These processes are influenced by factors, such as pH, available C and oxygen in the soil, and management. Moreover, the water filled pore space that affects the oxygen concentration of the soil alters the N_2O to NO ratio, and, therefore, may contribute also to different environmental impact categories (Oremland & Davidson, 1993). Emissions of N_2O and NO via different pathways are complex, and difficult to assess. This complexity requires more research as the alternative pathways may lead to overestimation or underestimation of N_2O emission under changing management. Reductions assumed in IS were based on best available data.

Nitrate leaching differs between soil types. Generally, clay soils have lower leaching compared to sandy soils (Schröder et al., 2003). In this study, we assumed similar management and leaching for both soils. Specifying to soil type will lead to slightly different results for ME, but will hardly affect comparison between Ref, IS, and NL.

The type of tillage affects the C in the soil (Venterea et al., 2005). Reduced tillage practices, such as non-inversion tillage, can reduce C emissions and hence increase C-sequestration, but also have shown to increase non-CO_2 GHG emissions (Ball et al., 1999; Holland, 2004). We excluded this C-sequestration here, but included C emissions from changed energy use. Furthermore, P run-off may be reduced due to reduced tillage and depends on factors such as the slope of the field and climate conditions. By applying non-inversion tillage in the IS, this will further reduce the loss of P.

6.4.3 Uncertainty analysis

Uncertainty of computed environmental impacts resulted mainly from prediction of N_2O and CH_4 emissions for CC, NH_3 emission for TA and PMF, and ANR assumed in NUE. This is consistent with the literature (De Vries et al., 2013, Chapter 4; Payraudeau et al., 2007). Some other uncertainties were excluded in this assessment, i.e. uncertainty in background emissions from electricity, fuel, and mineral fertilizer production. Based on previous studies however, in which uncertainties were included and tested, no changes in the comparison and significance of the IS and Ref were expected (De Vries et al., 2013, Chapter 4; De Vries et al., 2011). Finally, within Monte Carlo simulation, a correlation matrix of emission factors is often included. Here, we included dependency of factors via calculation where possible, i.e. N_2O, NO, N_2, and NO_3^- emission.

6.5 CONCLUSION

We conclude that newly designed strategies for integrated manure management (IS) avoided pollution swapping as intended in their design goal. Compared to current NW EU practice (Ref), IS simultaneously reduced climate change (CC, up to 176%), fossil fuel depletion (FFD, up to >700%), terrestrial acidification (TA, up to 92%), marine eutrophication (ME, up to 98%), particulate matter formation (PMF, up to 95%), P_2O_5 over application (POA, up to 103%), and more than doubled N use efficiency (NUE) for all manure product and crop combinations. Compared to Ref, current manure management in the Netherlands (NL) reduced TA, PMF, and POA, whereas CC was similar. Compared to Ref, NL had lower environmental impact for TA, PMF, POA, and higher NUE, except for solid cattle manure applied to grass.

This study showed that life cycle assessment was important to provide insight in (prevention of) pollution swapping and to evaluate all environmental consequences of strategies for integrated manure management. The presented outcomes are essential for those tackling environmental issues of manure management as it contributes to informed decision-making.

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APPENDIX A. SUPPLEMENTARY INFORMATION

The supplementary information is provided as an Appendix after the References section.

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Appendix A

// SUPPLEMENTARY INFORMATION: INTEGRATED MANURE MANAGEMENT TO REDUCE ENVIRONMENTAL IMPACT: II. ENVIRONMENTAL IMPACT ASSESSMENT OF STRATEGIES

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Table SI 1. Environmental emissions of the separate components in Ref, NL and IS

				CO_2	CH_4	N_2O_{dir}	NO	N_2	NH_3	NO_3^-	N_2O_{indir}	POA	SO_2	PM_{10}
Scenario	**FP**[a]	**Crop**	**MS**	kg	kg	kg	kg	kg	kg	kg	kg	kg	kg	kg
Ref	liquid	Grass	in-house	1.24	4.02	0.02	0.02	0.21	0.83	-	0.01	0.77	-	-
	cattle		outside	0.88					0.68	-	0.01		-	-
	manure		fapp + till	2.88	-	0.06	0.01	0.24	1.54	1.29	0.02		-	-
			af	-11.6	-0.03	-0.08	-0.06	-0.14	-0.08	-1.07	0.00		-0.06	-0.01
	solid	Grass	in-house	-	0.17	0.24	0.33	1.54	1.18	-	0.02	1.56	-	-
	cattle		outside	-					1.13	0.51	0.02		-	-
	manure		fapp + till	4.9	-	0.08	0.01	0.31	1.34	1.93	0.02		-	-
			af	-16.2	-0.04	-0.11	-0.08	-0.19	-0.10	-1.44	0.00		-0.09	-0.01
	liquid	Maize	in-house	1.24	4.02	0.02	0.02	0.21	0.83	-	0.01	0.81	-	-
	cattle		outside	0.88					0.68	-	0.01		-	-
	manure		fapp + till	2.88	-	0.06	0.01	0.24	1.54	3.58	0.03		-	-
			af	-10.5	-0.02	-0.07	-0.05	-0.12	-0.06	-2.51	-0.01		-0.05	-0.01
	liquid	Wheat	in-house	1.24	16.1	0.02	0.03	0.28	1.83	-	0.02	1.08	-	-
	pig		outside	0.88	0.17				0.67	-	0.01		-	-
	manure		fapp + till	3.26		0.08	0.01	0.29	1.65	4.51	0.03		-	-
			af	-13.0	-0.03	-0.09	-0.07	-0.16	-0.09	-3.41	-0.01		-0.08	-0.01
NL	liquid	Grass	in-house	1.24	5.45	0.01	0.01	0.11	0.42	-	0.01		-	-
	cattle		fapp + till	3.27		0.02	0.00	0.09	0.74	2.05	0.02	0.34	-	-
	manure		af	-16.2	-0.04	-0.12	-0.08	-0.19	-0.15	-1.82	-0.01		-0.07	-0.01
	solid	Grass	in-house	-	0.17	0.24	0.33	1.54	0.62	-	0.01	1.65	-	-
	cattle		outside	-					0.15	0.54	0.00		-	-
	manure		fapp + till	5.21	-	0.01	0.00	0.04	2.96	1.63	0.04		-	-
			af	-15.2	-0.03	-0.09	-0.07	-0.13	-0.11	-1.28	0.00		-0.09	-0.01
	liquid	Maize	in-house	1.24	5.45	0.01	0.01	0.11	0.42	-	0.01		-	-
	cattle		fapp + till	3.68	-	0.10	0.01	0.38	0.08	6.35	0.02	0.15	-	-
	manure		af	-16.7	-0.04	-0.12	-0.09	-0.17	-0.16	-5.31	-0.02		-0.07	-0.01

Table SI 1. continued

				CO_2	CH_4	N_2O_{dir}	NO	N_2	NH_3	NO_3^-	N_2O_{indir}	POA	SO_2	PM_{10}
Scenario	**FP**	**Crop**	**MS**	kg	Kg	kg	kg	kg	kg	kg	kg	kg	kg	kg
NL	liquid	Wheat	in-house	1.24	13.1	0.01	0.01	0.14	1.76	-	0.02	0.38	-	-
	pig		outside	0.88	0.17				0.13	-	0.00		-	-
	manure		fapp + till	3.47	-	0.11	0.01	0.41	0.09	6.83	0.02		-	-
			af	-18.3	-0.04	-0.13	-0.09	-0.19	-0.18	-5.76	-0.02		-0.09	-0.01
IS	cattle	Grass	in-house	0.35	0.09	0.01	0.01	0.13	0.32	-	0.00	0.08	-	-
	urine		out/proc	-30.9	0.12	0.00	-0.04	0.01	0.09	-	0.00		-	-0.02
	digested		fapp + till	2.47	-	0.04	0.01	0.14	0.14	2.36	0.01		-	-
	feces		af	-18.9	-0.04	-0.17	-0.10	-0.29	-0.16	-2.27	-0.01		-0.10	-0.01
	solid	Grass	in-house	-	0.02	0.02	0.03	0.14	0.15	-	0.00	0.67	-	-
	cattle		out/proc	2.82					0.16	0.65	0.00		-	-
	manure		fapp + till	4.46	-	0.07	0.01	0.28	0.24	3.31	0.01		-	-
			af	-25.6	-0.06	-0.22	-0.14	-0.38	-0.21	-2.97	-0.01		-0.11	-0.01
	cattle	Maize	in-house	0.35	0.09	0.01	0.01	0.13	0.32	-	0.00	0.02	-	-
	urine		out/proc	-30.9	0.12	0.00	-0.04	0.01	0.09	-	0.00		-	-0.02
	digested		fapp + till	2.47	-	0.04	0.01	0.14	0.14	6.57	0.02		-	-
	feces		Af	-18.0	-0.04	-0.16	-0.10	-0.27	-0.15	-5.87	-0.02		-0.09	-0.01
	pig	Wheat	in-house	0.35	0.91	0.11	0.11	1.06	1.01	-	0.01	-0.03	-	-
	urine		out/proc	-43.2	0.15	0.00	-0.06	0.01	0.08	-	0.00		-	-0.02
	digested		fapp + till	2.68	-	0.05	0.01	0.18	0.13	7.82	0.02		-	-
	feces		Af	-21.3	-0.04	-0.19	-0.12	-0.33	-0.18	-7.15	-0.02		-0.12	-0.01

[a] FP = fertilizer product, MS = management stage, POA = P_2O_5 over application, fapp = field application, till = tillage, af = avoided mineral fertilizer application and production, out/ proc = outside storage/ processing or treatment, Ref = NW EU reference, NL = Dutch state of art, IS = strategies for integrated manure management.

Table SI 2. Environmental impact assessment per management stage for Ref, NL, and IS

Scenario	FP[a]	Crop	MS	CC kg CO_2-eq	TA kg SO_2-eq	ME kg N-eq	PMF kg PM_{10}-eq	FFD MJ	NUE %	POA kg
Ref	liquid	Grass	in-house	110	2.05	0.09	0.27	20.1	33%	0.77
	cattle		outside	3.52	1.68	0.06	0.22	14.2		
	manure		fapp + till	28.6	3.77	0.44	0.49	64.7		
			Af	-37.3	-0.28	-0.27	-0.06	-209		
	solid	Grass	in-house	81.5	3.08	0.24	0.45	0	30%	1.56
	cattle		outside	4.74	2.76	0.22	0.36	0.0		
	manure		fapp + till	36.0	3.30	0.56	0.43	102		
			Af	-50.6	-0.38	-0.37	-0.08	-287		
	liquid	Maize	in-house	110	2.05	0.09	0.27	20.1	18%	0.81
	cattle		outside	3.52	1.68	0.06	0.22	14.2		
	manure		fapp + till	30.4	3.77	0.95	0.49	64.7		
			Af	-33.5	-0.24	-0.59	-0.05	-188		
	liquid	Wheat	in-house	417	4.49	0.18	0.59	20.1	27%	1.08
	Pig		outside	7.70	1.64	0.06	0.21	14.2		
	manure		fapp + till	35.7	4.05	1.17	0.53	74.7		
			Af	-44.1	-0.33	-0.80	-0.07	-230		
NL	liquid	Grass	in-house	142	1.02	0.04	0.14	20.1	56%	0.34
	cattle		fapp + till	14.6	1.81	0.53	0.24	75.3		
	manure		af	-56.0	-0.49	-0.46	-0.09	-301		
	solid	Grass	in-house	79.3	1.71	0.19	0.27	-		
	cattle		outside	1.00	0.36	0.14	0.05	-	27%	1.65
	manure		fapp + till	20.7	7.26	0.64	0.95	110		
			af	-43.5	-0.39	-0.33	-0.08	-268		
	liquid	Maize	in-house	142	1.02	0.04	0.14	20.1		
	cattle		fapp + till	38.9	0.20	1.45	0.03	86.4	39%	0.15
	manure		af	-59.3	-0.51	-1.25	-0.09	-312		

Table SI 2. continued

Scenario	FP[a]	Crop	MS	CC kg CO_2-eq	TA kg SO_2-eq	ME kg N-eq	PMF kg PM_{10}-eq	FFD kg	NUE %	POA kg
NL	liquid	Wheat	in-house	339	4.32	0.17	0.57	20.1	45%	0.38
	pig		outside	5.63	0.32	0.01	0.04	14.2		
	manure		fapp + till	41.4	0.24	1.56	0.03	80.4		
			af	-64.3	-0.57	-1.35	-0.11	-334		
IS	cattle	Grass	in-house	7.02	0.80	0.04	0.11	5.66	69%	0.08
	urine		out/proc	-27.6	0.19	-0.01	0.00	-508		
	digested		fapp + till	16.0	0.34	0.55	0.05	52.8		
	feces		af	-72.9	-0.55	-0.57	-0.10	-355		
	solid	Grass	in-house	7.62	0.39	0.03	0.06	0	63%	0.67
	cattle		out/proc	3.97	0.40	0.16	0.05	38.0		
	manure		fapp + till	29.9	0.60	0.77	0.08	89.3		
			af	-96.2	-0.70	-0.74	-0.13	-474		
	cattle	Maize	in-house	7.02	0.80	0.04	0.11	5.66	43%	0.02
	urine		out/proc	-27.6	0.19	-0.01	0.00	-508		
	digested		fapp + till	19.4	0.34	1.50	0.05	52.8		
	feces		af	-71.1	-0.52	-1.38	-0.10	-336		
	pig	Wheat	in-house	58.7	2.54	0.13	0.35	5.66	56%	-0.03
	urine		out/proc	-39.4	0.17	-0.01	-0.01	-710		
	digested		fapp + till	23.5	0.33	1.78	0.04	58.8		
	feces		af	-86.0	-0.64	-1.68	-0.12	-395		

[a] FP = fertilizer product, MS = management stage, CC = climate change, TA = terrestrial acidification, ME = marine eutrophication, POA = P_2O_5 over application, PMF = particulate matter formation, FFD = fossil fuel depletion, NUE = nitrogen use efficiency, fapp = field application, till = tillage, af = avoided mineral fertilizer application and production, out/ proc = outside storage/ processing or treatment, Ref = NW EU reference, NL = Dutch state of art, IS = strategies for integrated manure management.

Table SI 3. Results of the uncertainty analysis after a thousand Monte Carlo simulations for the Ref, NL and IS with mean outcomes, the standard deviation (SD), and the coefficient of variation (CV)

				CC	TA	ME	POA	PMF	FFD	NUE
Scenario	**FP**[a]	**Crop**		kg CO_2-eq	kg SO_2-eq	kg N-eq	kg	kg PM_{10}-eq	MJ	%
Ref	liquid	Grass	Mean	111	7.27	0.31	0.78	0.93	-110	32.8%
	cattle		SD	33.5	1.64	0.07	0.11	0.22	22	6.50%
	manure		CV	30%	23%	21%	14%	23%	-20%	20%
	solid	Grass	Mean	85.0	8.80	0.66	1.56	1.17	-185	30.4%
	cattle		SD	38.4	1.69	0.11	0.04	0.23	9.12	3.51%
	manure		CV	45%	19%	16%	3%	20%	-5%	12%
	liquid	Maize	Mean	118	7.31	0.51	0.81	0.94	-89	18.2%
	cattle		SD	34.9	1.64	0.11	0.12	0.22	22.3	4.17%
	manure		CV	30%	22%	21%	15%	23%	-25%	23%
	liquid	Wheat	Mean	432	9.79	0.61	1.07	1.26	-117	27.0%
	pig		SD	104	2.32	0.12	0.20	0.30	29.6	5.63%
	manure		CV	24%	24%	20%	19%	24%	-25%	21%
NL	liquid	Grass	Mean	100	2.36	0.12	0.34	0.28	-205	55.8%
	cattle		SD	35.6	0.62	0.03	0.05	0.08	11.4	6.22%
	manure		CV	36%	26%	25%	16%	29%	-6%	11%
	solid	Grass	Mean	68.2	8.87	0.64	1.66	1.18	-156	26.7%
	cattle		SD	38.7	2.41	0.09	0.20	0.31	42.6	7.91%
	manure		CV	57%	27%	14%	12%	26%	-27%	30%
	liquid	Maize	Mean	125	0.71	0.24	0.15	0.07	-205	38.69%
	cattle		SD	42.0	0.26	0.08	0.03	0.03	5.14	3.91%
	manure		CV	34%	36%	34%	18%	49%	-3%	10%
	liquid	Wheat	Mean	327	4.34	0.38	0.38	0.54	-219	45.1%
	pig		SD	90.0	1.10	0.09	0.14	0.15	20.8	5.10%
	manure		CV	28%	25%	23%	37%	27%	-9%	11%

Table SI 3. continued

Scenario	FP[a]	Crop		CC kg CO_2-eq	TA kg SO_2-eq	ME kg N-eq	POA kg	PMF kg PM_{10}-eq	FFD MJ	NUE %
IS	cattle urine	Grass	Mean	-78.7	0.78	0.01	0.07	0.05	-806	69.6%
	digested		SD	9.96	0.24	0.01	0.02	0.03	4.57	7.01%
	feces		CV	-13%	30%	184%	29%	62%	-1%	10%
	solid	Grass	Mean	-55.0	0.68	0.22	0.67	0.05	-348	62.9%
	cattle		SD	10.9	0.42	0.03	0.05	0.06	10.5	6.50%
	manure		CV	-20%	61%	15%	7%	108%	-3%	10%
	cattle urine	Maize	Mean	-72.9	0.81	0.15	0.02	0.06	-786	42.7%
	digested		SD	9.24	0.24	0.06	0.02	0.03	4.55	4.27%
	feces		CV	-13%	29%	38%	131%	55%	-1%	10%
	pig urine	Wheat	Mean	-45.3	2.41	0.25	-0.01	0.27	-1037	55.5%
	digested		SD	33.4	1.03	0.10	0.18	0.15	26.1	6.21%
	feces		CV	-74%	43%	41%	-1299%	55%	-3%	11%

[a] FP = fertilizer product, CC = climate change, TA = terrestrial acidification, ME = marine eutrophication, POA = P_2O_5 over application, PMF = particulate matter formation, FFD = fossil fuel depletion, NUE = nitrogen use efficiency, Ref = NW EU reference, NL = Dutch state of art, IS = strategies for integrated manure management.

Table SI 4. Calculated z-values for comparison of significance for the Ref, NL, and IS

FP[a]	**Crop**	**Comparison**	CC	TA	ME	POA	PMF	FFD	NUE
		Ref comp to IS	5.44[b]	3.92	4.48	6.48	4.06	30.7	3.85
lcm/ cu+dcf	Grass	Ref comp to NL	0.24	2.81	2.69	3.68	2.82	3.82	2.56
		NL comp to IS	4.83	2.39	3.44	4.50	2.69	49.0	0.00
		Ref comp to IS	3.51	4.67	3.95	13.5	4.67	11.7	4.39
scm	Grass	Ref comp to NL	0.31	-0.02	0.18	-0.49	-0.02	-0.67	0.43
		NL comp to IS	3.06	3.35	4.32	4.77	3.57	4.37	3.53
		Ref comp to IS	5.29	3.93	3.04	6.54	4.06	30.7	4.09
lcm/ cu+dcf	Maize	Ref comp to NL	-0.13	3.98	2.03	5.43	3.98	5.10	3.57
		NL comp to IS	4.61	-0.29	0.93	3.57	0.31	84.7	0.70
		Ref comp to IS	4.36	2.91	2.23	4.06	2.91	23.3	3.40
lpm/ pu+dpf	Wheat	Ref comp to NL	0.76	2.12	1.47	2.81	2.13	2.81	2.38
		NL comp to IS	3.88	1.28	0.98	1.75	1.29	24.5	1.29

[a] FP = fertilizer product (lcm = liquid cattle manure, cu = cattle urine, dcf = digested cattle feces, scm = solid cattle manure, lpm = liquid pig manure, pu = pig urine, dpf = digested pig feces), Ref = NW EU reference, NL = Dutch state of art, IS = strategies for integrated manure management, CC = climate change, TA = terrestrial acidification, ME = marine eutrophication, POA = P_2O_5 over application, PMF = particulate matter formation, FFD = fossil fuel depletion, NUE = nitrogen use efficiency.
[b] z-values were calculated using the following formula: $z = |m_1 - m_2| /$ square root$(CV_1^2 \times m_1^2 + CV_2^2 \times m_2^2)$ where m_1 and m_2 are the means for the to be compared systems and CV_1 and CV_2 are the coefficients of variation for the to be compared systems. Differences were assumed significant: $P < 0.05$ with $z > 1.96$; $P < 0.01$ with $z > 2.005$; and $P < 0.001$ with $z > 3.606$.

Chapter 7

// GENERAL DISCUSSION AND CONCLUSIONS

J.W. De Vries

7.1 INTRODUCTION

The objective of this thesis was to provide knowledge and insight into the environmental consequences of current and future strategies for manure management. In Chapters 2 - 4, we addressed the environmental consequences of current manure processing technologies (Fig. 1.2): anaerobic digestion of liquid manure (mono-digestion) and digestion of liquid manure with co-substrates (co-digestion), high-tech processing of liquid manure, and separation of pig urine and feces immediately after excretion by the animal inside the housing system (referred to as segregation). The knowledge obtained was used to further fathom the current manure management system and design new strategies for integrated pig and dairy cattle manure management (IS) in Chapter 5 (Fig. 1.2). IS were created by a structured design approach that included all emission processes, functions, and interactions to address pollution swapping and reduce environmental impact. In Chapter 6, we quantitatively assessed the potential of these strategies to reduce the environmental impact. This chapter discusses the main findings of these studies and the implications of these findings for future developments in manure management in Europe and the Netherlands.

7.2 ENVIRONMENTAL CONSEQUENCES OF CURRENT AND FUTURE STRATEGIES FOR MANURE MANAGEMENT

7.2.1 Environmental impact of current manure processing technologies

In Chapters 2, 3, and 4, we assessed the environmental consequences of using current available technologies for manure management. Environmental consequences were expressed as changes in environmental impact categories: climate change (CC), terrestrial acidification (TA), marine eutrophication (ME), freshwater eutrophication or P_2O_5 over application (POA), particulate matter formation (PMF), and fossil fuel depletion (FFD) (Table 1.1, Chapter 1). These assessments are summarized in Table 7.1. We found that mono-digestion of liquid pig manure reduced the environmental impact as compared to conventional manure management without digestion, but that the production of bio-energy was low (Chapter 2). Co-digestion with substrates competing with animal feed, like maize silage, beet tails, and wheat yeast concentrate increased bio-energy production >550%[1] compared to mono-digestion, but also increased the environmental impact up to a factor 78. TA, PMF, and FE, however, were reduced up to a factor 22 mainly from avoided production of fossil-based electricity and heat. Because these substrates were used in co-digestion instead as animal feed, a substitute was needed for animal feed. This caused pollution swapping and had a large effect on the total impact, in some impact categories increasing up to roughly 100% of the initial positive contribution. This was partly caused by land use change emissions (LUC) that increased CC and land use. LUC results from the expansion of agricultural land for producing food, feed, fiber, and fuel.

Using additional land for bio-energy induces the expansion of land elsewhere in the

[1] When a system, for example, produces energy or avoids more energy than it uses, the difference with a reference system that uses energy becomes larger than 100%. Similarly, this occurs when more emissions are avoided in the system compared to the reference. Fractions larger than 1000% are given as factors.

Table 7.1 Summary of the environmental consequences of current available technologies for manure management and future strategies for manure management as reported in this thesis.

Thesis section	Environmental impact is reduced by	Environmental impact is increased by
Chapter 2	- Mono-digestion of liquid pig manure. - Co-digestion with residuals & waste products, such as roadside grass.	- Co-digestion with products that compete with animal feed, such as maize silage, beet tails, and wheat yeast concentrate.
Chapter 3	- Mono-digestion of separated solid fraction/ avoided storage - Utilizing excess heat from digestion.	- Liquid manure separation with de-watering of liquid fraction. - Storage of separated fractions.
Chapter 4	- Segregating pig urine and feces directly after excretion.	- Storage and field application of feces with high dry matter content.

world, which leads to conversion of different biomes. Our results support earlier findings that the effect of LUC negates the reduction of CC from biofuels (Plevin et al., 2010; Searchinger et al., 2008; Tonini et al., 2012). The exact extent of LUC, however, is difficult to determine due to the inherent uncertainty (Chapter 2). Using waste products and residues, such as roadside grass instead of ‘high-grade products’, was environmentally most sustainable. These products do not compete with other markets, but represent management with lower environmental impact, here co-digestion instead of composting. Selecting the ‘right’ co-substrates for co-digestion from an environmental perspective, therefore, lies in considering the alternative use of the co-substrates rather than the potential for bio-energy production. The alternative use of co-substrates is determined by market mechanisms and is subject to uncertainty due to economic changes. Addressing these mechanisms in consequential LCA remains essential (Weidema, 2003). Uncertainties can be dealt with by a sensitivity or uncertainty analysis.

When waste products or residues are available as co-substrates, the ‘waste hierarchy’, or ‘Lansink’s ladder’ can be applied. This hierarchy provides a framework for selecting the most environmentally favored waste management option, where waste prevention is most favored and waste disposal is least favored. According to this hierarchy, reuse and recycling of waste is more environmentally preferred compared to energy recovery. This is in agreement with the findings of our study where use of by-products for bio-energy production increased the impact when competing with animal feed. It is also supported by a recent study that suggested to use beet tails as animal feed rather than for bio-energy production as this improves the overall environmental consequences of its use (Van Zanten et al., 2013).

In Chapter 3, we concluded that the processing of liquid manure by separation into a solid and a liquid fraction, and further de-watering of the liquid fraction, increased the environmental impact up to 110%, except for ME and CC with liquid cattle manure, mainly caused by the processing and storage of separated fractions (Table 7.1). Other authors also reported an increase of greenhouse gas emissions with storage of separated fractions (Dinuccio et al., 2008; Petersen & Sørensen, 2008). Similar to results from Chapter 2, anaerobic mono-digestion of the separated solid fraction reduced CC with 117% and FFD with 59%. This effect was stronger when excess heat was utilized to avoid fossil-based heat (Chapter 3). Therefore, unless combined with digestion, separation and de-watering of the liquid fraction provides no environmental benefit compared to current practice where no processing occurs.

In Chapter 4, results showed that segregation of pig urine and feces reduced CC, TA, and PMF with 82%, 49%, and 49%, respectively compared to current liquid manure management. Other benefits of segregation are reduced odor emission and the low energy input needed (Aarnink et al., 2007). Here, we focused on segregation by a V-belt system only, but segregation may be obtained also by other methods, such as tie stall housing and grooved floors with drainage of urine (Swierstra et al., 2001). Developments in methods for segregating pig urine and feces also focus on influencing the excretory behavior of pigs, which may lead to the excretion in the so-called 'pig toilet'. This may provide additional benefits for the animal and environment, such as more walking and lying space for the animal, improved indoor climate, and reduced odor emissions (Van Weeghel et al., 2011). The results in Chapter 4 also showed that storage and field application of feces with a high dry matter content caused higher CC, TA, and PMF impact (up to 94%) than feces with a low dry matter content. This agrees with the outcomes in Chapter 3, which showed that the storage of solid fraction increased environmental impact in the whole system. Overall, segregation provides a sound basis for environmentally sustainable manure management, but attention is needed for storage and field application.

Considering the outcomes from Chapter 2 - 4, pollution swapping appeared an important obstacle to reduce the environmental impact throughout the manure management system. Current technologies hardly prevent pollution swapping, because they often do not consider the underlying processes, as explained in Chapter 1. The knowledge from the assessments was used to design strategies for integrated manure management that prevent pollution swapping, and by doing so, reduce the environmental impact throughout the manure management system.

7.2.2 Environmental impact of strategies for integrated manure management

To prevent pollution swapping and reduce the environmental impact, we applied a structured design approach to create IS (Chapter 5). In Chapter 6, we found that compared to current North Western European (NW EU) practice, the strategies for integrated pig and dairy cattle manure management reduced environmental impact and increased nitrogen use efficiency (NUE) for all manure product and crop combinations. With the designed IS it was possible to simultaneously reduce CC, up to 176%, FFD, >700%, TA, up to 92%, ME, up to 98%, PMF, up to 95%, POA, up to 103%, and more than doubled the NUE, up to 69%. Our figures from NW EU corresponded well with literature (Oenema et al., 2007) and showed that the design process lead to strategies that avoid pollution swapping. At the same time, N losses in the IS were reduced (Fig. 7.1) throughout the manure management system as compared to current practice (Fig. 1.1). In the IS, the fraction of excreted N lost to the environment was 31 - 57% as compared to 67 - 82% in current practice. The estimated reduction in N loss from improved animal feeding and manure management was ~30% and was reported to be one of the key interventions needed to reduce the global emission of reactive N (Galloway et al., 2008). Reductions in N losses in the IS even run up to 54% points, which indicates that the IS provide a sound basis to reduce N loss from manure to the environment. Lower N losses also reduced the need for N from mineral fertilizer. This reduces related production emissions, and POA (Chapter 6).

The integrated strategies designed here should be validated with measurements. Laboratory and field experiments focusing on the different pathways of N and P losses would provide insight into the true reduction potential of the integrated strategies and the nutrient use efficiencies. Increasing the NUE by applying N at the right time and place was highlighted as one of the most stringent actions to reduce environmental impact of food production (Cassman et al., 2002; Spiertz, 2010; Tilman et al., 2002). Using the NUE as an indicator for environmental impact, however, has its limitations. The definition of NUE can vary according to the goal of the study and thus may cover various parts of the production system (Schröder et al., 2003). The NUE can, e.g. incorporate human diets and express the efficiency of producing different food crops (Cassman et al., 2002; Schröder et al., 2003). The type of crop must be considered when using the NUE, because intrinsic crop characteristics determine the efficiency of its N metabolism and thus N uptake (Spiertz, 2010). Apart from this, the NUE does not qualify the environmental impact and does not show pollution swapping, e.g. N_2O emission in relation to CC, NH_3 emission in relation to TA, ME, and PMF, or N_2 emission without adverse impact. The NUE, therefore, should be used with caution, and needs to be complemented with environmental impact categories to provide a more elaborate insight of the environmental consequences. Overall, the results in Chapter 6

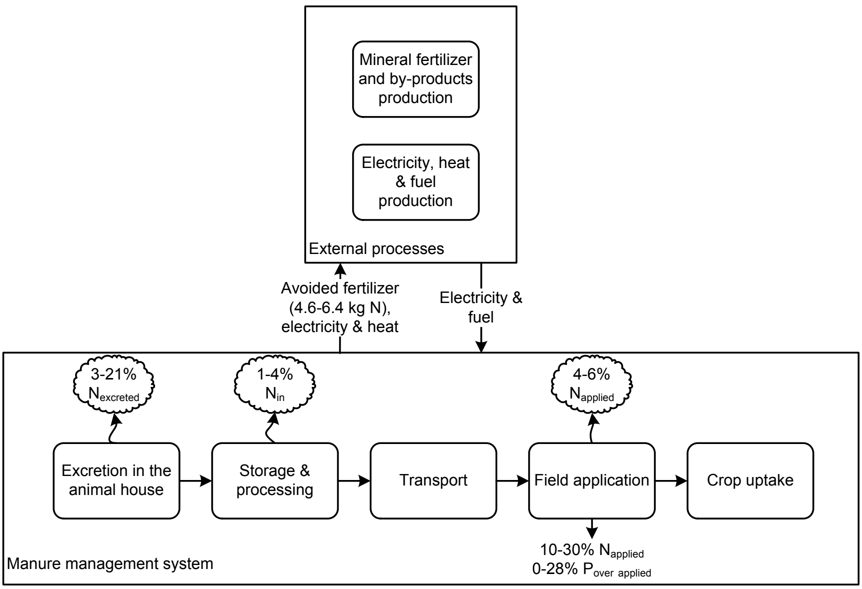

Fig. 7.1 Manure management system, range of N and P losses during management (gaseous losses upwards, leaching losses downwards), and external processes that are affected by changes in manure management.

demonstrate that an important step can be made to increase the NUE and simultaneously reduce environmental impact as part of integrated strategies for manure management.

Greenhouse gas (GHG) emissions and CC impact per ton of manure in the IS were also reduced compared to current practice, with reduction ranging from 110% to 176%. This was mainly caused by reduced CH_4 emission from segregation of urine and feces inside the animal housing system, and by production of bio-energy and avoiding CO_2 from fossil electricity. This is in line with other authors, who reported a reduction up to roughly 125% compared to no digestion (Hamelin et al., 2011; Hristov et al., 2013; Prapaspongsa et al., 2010).

/ 7.3 APPLIED METHODOLOGIES

In Chapter 5, we used a structured design approach based on engineering design to create the IS. This method has proven its value in other projects to design innovative housing systems for laying hens and dairy cows (Bos et al., 2009). Adapting the method to the manure management system was necessary to analyze the current system with its functions and structure the options and solutions needed in the design process. The method, therefore, can be applied in other research areas of biosystems engineering as well. As discussed in Chapter 5, some choice(s) in the construction of the IS have to be made. The knowledge and expertise of the designer plays an important role. The design approach, therefore, differs from natural sciences where repeatability and reproduction of results is essential. In structured design, it is essential that the design steps and decisions taken can be traced and verified rather than exactly reproduced (Eekels, 2000; Eekels & Roozenburg, 1991). In natural sciences the main quality requirement is how results correspond or correlate with reality, whereas the quality requirement of structured design is how the outcomes meet the needs of the stakeholders, as presented in the brief of requirements (Eekels & Roozenburg, 1991). The main goal in Chapter 5, therefore, was to design strategies for integrated manure management that avoid pollution swapping rather than finding the 'one-size-fits-all' or the best obtainable solution. This was achieved in the IS. It is shown that the method provides a usable framework for such a design challenge.

Life cycle assessment (LCA) was used to assess the environmental impact of manure management (ISO-14040, 2006). The method has proven its usefulness in various research fields (Finnveden et al., 2009). The strength of LCA lies in addressing environmental impact throughout the production system, revealing 'environmental hotspots' and pollution swapping. In LCA, environmental emissions are compiled into different environmental impact categories and can also contribute to more than one category, such as NH_3 (Table 1.1.). LCA, therefore, complements other system approaches, such as material flow analysis and multi criteria analysis, where often only a single flow or mass balance is considered or where multiple indicators, next environmental ones, are included for optimizing decisions. These methods answer different questions and can be applied separately or combined depending on the goal to be achieved, e.g. Guinée et al. (2010) and Shields et al. (2011).

As with all tools, LCA results should be interpreted given the defined goal and scope, which includes the system boundaries, of a study. It was shown that it is essential to include the environmental consequences of production of mineral fertilizer as an external process, as it provides a more comprehensive assessment compared to other approaches without LCA, e.g. Velthof et al. (2009). Some environmental impacts are more site specific than others, which requires definition of the location where the study is applied. ME, for example, will change according to the soil type and the local precipitation surplus (Schröder et al., 2011). In this thesis an average soil type was assumed, as we were interested in finding the environmental consequences compared to a reference of current practice regardless of soil type. The LCA as applied here provided a comprehensive overview of the environmental consequences of current and future strategies for manure management.

7.4 IMPLICATIONS FOR POLICY REGULATIONS

Countries have to comply with emission levels in line with several directives or regulations, such as the national emission ceilings (NEC), Nitrates Directive, and the Kyoto protocol (EC, 2013; EU, 2001; EU, 2013). The directives aim to decrease the emissions of NH_3, NO_x, SO_2 and volatile organic compounds, NO_3^- concentrations in water bodies, and GHGs, respectively (Chapter 1). The directives have until now been successful in reducing pollutants including those stemming from manure management. NH_3 emission, for example, has declined by 28% between 1990 and 2010 and GHG emission have declined by 14.9% for the EU-15 in the same time frame (EC, 2013; EC, 2012). However, the regulations also have undesired side effects that are expressed in pollution swapping and unnecessary relocation of nutrients. This is explained in the following paragraphs.

Firstly, regulations are meant to reduce certain pollutants. The NEC, for example, aims at reducing NH_3 emission and thus stimulates the use of low emission application techniques for manure. At the same time N_2O emission is increased from injection, which causes an increase in GHGs that are regulated by the Kyoto protocol. Because there are no existing directives to reduce GHG emission from manure management, this induces unwanted pollution swapping. This is also highlighted in Chapter 6 where CC for NW EU conditions and the situation of the Netherlands is similar for all manure product and crop combinations, but where TA, ME, and PMF, mainly caused by NH_3 emission, are lower in the Netherlands. The Netherlands has strict regulations to comply with the NEC.

Secondly, when the IS, as presented in Chapter 5 and 6, would be implemented, emissions of N would be reduced throughout the manure management system causing increased total N content in the manure products and more efficient application. But because the application of N from manure is limited by the Nitrates Directive (170 kg N ha^{-1}), in a surplus situation the 'saved N emission' needs to be exported at the expense of the farmer (Fig. 7.2). At the same time, the crop in the surplus region remains to have a deficiency of N and requires mineral N fertilizer for optimum growth. For further elaboration also see De Vries et al. (2011). This reveals that under current regulations the efficient use of nutrients from manure in crop production is limited in situations with a manure surplus.

Furthermore, in Chapter 3 and De Vries et al. (2011), it was shown that production and use of mineral concentrates resulted in relocation of nutrients within the agricultural system, but did not lead to more efficient use of N from manure. Mineral concentrates are still considered as animal manure according to the Nitrates Directive. If they were considered as a 'mineral fertilizer' it would mean that in a surplus region more nutrients from animal manure could be applied (Fig. 7.2) (Lesschen et al., 2011), but in that case more mineral fertilizer is required in the region previously receiving this surplus manure (the 'external region', Fig. 7.2) (De Vries et al., 2011). So, under current regulations, distribution of manure and nutrients is altered, but nutrients are not used more efficiently. The efficiency of using nutrients from manure can only be increased when more adequate management measures are taken, such as adjusting storage and the application timing and method, as illustrated in the IS (Chapter 5 and 6 and section 7.2.2). In the IS, the reduced N emissions and improved application timing, place, and method lead to more available N for the crop and more avoided mineral N fertilizer. Alternative processing methods, such as separation and de-watering of liquid fraction may, therefore, be an option under current circumstances and regulations, but do require considerable investment, roughly 9 - 13 Euro per ton of manure (De Hoop et al., 2011).

With anaerobic co-digestion in NL, the digestate by law is considered as animal manure when 50% or more of the added substrates consists of animal manure (DR, 2012). By adding co-substrates, such as maize silage and beet tails, the nutrient concentration in the digestate is increased. Consequently, more nutrients in the digestate need to be applied within the application limits of animal manure (Chapter 2). Thus, the nutrient surplus becomes even larger when co-digestion occurs with nutrient rich co-substrates. Economic feasibility of co-digestion was reported to be low or even negative (Gebrezgabher et al., 2010). Economic feasibility is mainly determined by the subsidies received to produce bio-energy. This stimulates the increased use of high biogas yielding substrates which compete with animal feed and other applications and leads to adverse environmental impacts (Chapter 2).

Over the last 50 - 60 years, production of liquid manure increased labor efficiency in the housing, in transport and at the field application stage compared to solid manure. It can be questioned, however, if liquid manure is the most suitable fertilizer from an environmental and crop production point of view. Firstly, the production of liquid manure mixes N, P, and C, but not always in the best ratio for crop production; N/ P_2O_5 ratios in manure are often lower than 2, whereas the N/ P_2O_5 ratios demanded by the crop are often higher than 2.5 (Schröder, 2005). This results in a higher risk of over or under application of N or P and consequently loss to the environment. Secondly, mixing of urine and feces stimulates enzymatic and microbial processes, such as methanogenis that produces CH_4 and hydrolysis of urea to NH_4^+ and NH_3 (Chapter 1) (Zeeman, 1994). Even more toxic gases, such as hydrogen sulfide are produced during, storage claiming lives of farmers (Donham et al., 1988). The production of liquid manure, therefore, presents drawbacks that also lead to less efficient use of nutrients.

Concluding from this section, the results from the IS clearly show that N losses simultaneously can be reduced and the NUE increased. However, the application in practice is hindered and causes unnecessary relocation of nutrients. Moreover, results

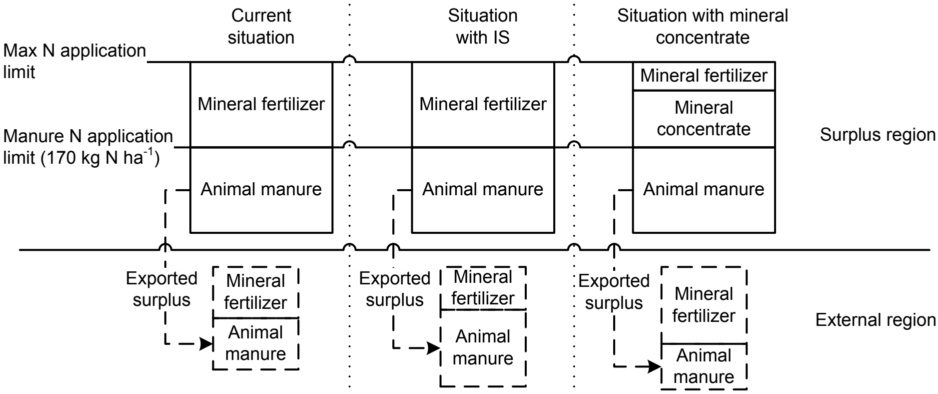

Fig. 7.2 Illustration of distribution of manure nutrients under the current regulations of the Nitrates Directive (170 kg N ha^{-1} in nitrate vulnerable zones) and a situation where the strategies for integrated manure management (IS) and mineral concentrates are implemented. Maximum N application limits are country/ site specific.

of NW European practice show that directives to reduce NH_3 and NO_3^- have been successful, but because no directives for GHG emissions are implemented there remains an opportunity for reducing CC. Finally, current practice is based on developments and thinking from the past 50 - 60 years. With this thinking we have been able to reduce environmental impact from manure management, but new 'integrated thinking' is needed to further address these issues. The greatest challenge for 'sustainable' manure management, therefore, lies in not taking trade-offs too easily for granted, but to rethink and reshape the system.

7.5 GENERAL CONCLUSIONS

Overall, we conclude that current technologies for manure management have potential to reduce environmental consequences, but also induce pollution swapping causing environmental impact to increase elsewhere inside or outside the manure management system. Newly designed strategies for integrated manure management prevent pollution swapping and reduce the environmental consequences throughout the manure management system. Specifically, we conclude that from an environmental perspective:

- Mono-digestion of liquid pig manure reduces the environmental impact compared to conventional manure management, but represents a limited source for bio-energy. Co-digestion with waste and residues, such as roadside grass, increases bio-energy production and further reduces the environmental impact. Co-digestion with substrates that compete with animal feed, such as maize silage, beet tails, and wheat yeast concentrate increases bio-energy, but also the overall environmental impact from producing a substitute for the used co-substrate. Land use change emission is an important factor determining the final environmental consequences of co-digestion.

- Separating liquid manure into liquid and solid fractions and de-watering of the liquid fraction increases the environmental impact as compared to manure management where no processing occurs. When anaerobic mono-digestion of the solid faction is included, this processing method leads to an environmental benefit for climate change and fossil fuel depletion. It is also shown that this processing method relocates nutrients only, rather than increasing the efficient use of the nutrients.

- Segregating fattening pig urine and feces in the housing system reduces climate change, terrestrial acidification, and particulate matter formation and forms a sound basis for environmentally friendly manure management. Storage and application of feces with a high dry matter content increases climate change, terrestrial acidification, and particulate matter formation compared to storage and application of feces with low dry matter content.

- Applying a structured design approach enables the design of new strategies for integrated manure management that prevent pollution swapping and thus reduce environmental impact throughout the manure management system.

- The designed strategies for integrated manure management prevent pollution swapping and reduce the environmental impact throughout the manure management chain by at least 57% and more than double the nitrogen use efficiency for all manure product and crop combinations compared to current North Western European manure management practices.

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/ SUMMARY

Animal manure is the key component that links crop and livestock production. It contains valuable nutrients for the soil and crop, such as nitrogen (N), phosphorus (P), and potassium (K), and carbon (C). Manure is also a source of environmental pollution. In areas with high livestock densities, such as the European Union (EU), management of manure causes major losses of nutrients and C into air, water, and soil. Main pathways and impacts of nutrient losses include: leaching and run-off of nitrate (NO_3^-) and phosphate (PO_4^{3-}) to ground and surface waters, resulting in eutrophication and human health problems; emission of greenhouse gases (GHGs), such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), resulting in climate change; and emissions of ammonia (NH_3) resulting in acidification, eutrophication and particulate matter formation. Management of manure includes several stages from animal to crop: the collection and storage in the animal house, outside storage, processing, transport and field application. So far, technologies to reduce the losses of N, P, and C and the environmental impact of manure management mainly focused on reducing the loss of a single compound, such as NH_3, N_2O, or CH_4, or on a single management stage only, such as storage or field application. These approaches often cause(d) pollution swapping meaning that the targeted loss was reduced, but another loss was increased. Preventing pollution swapping is complex because of interactions between process variables of underlying emission processes and, therefore, requires a structured approach and (re-)design of the whole manure management system that addresses all functions and underlying processes that lead to losses of N, P, and C. From this approach, strategies for integrated manure management can be designed. These strategies, however, require integrated assessment to reveal their potential to reduce environmental impact. Additionally, current technologies for manure management are developed. Such technologies include, separation of liquid manure and de-watering of the liquid fraction, anaerobic co-digestion with various co-substrates, and other separation methods. These technologies also require integrated assessment to consider their environmental consequences. Knowledge obtained from these assessments in turn feeds into the (re-)design of new strategies for manure management. The main objective of this thesis, therefore, was to provide knowledge and insight into the environmental consequences of current and future strategies for manure management. The sub-objectives were:

1 assess the environmental consequences of bio-energy production by means of anaerobic mono and co-digestion of pig manure, while accounting for the production of substitutes for used co-substrates (Chapter 2),

2 assess the environmental consequences of high-tech separation of liquid pig and dairy cattle manure (Chapter 3),

3 assess the environmental consequences of segregating fattening pig urine and feces inside the housing system (Chapter 4),

4 design strategies for integrated manure management that prevent pollution swapping and reduce the environmental impact throughout the manure management system (Chapter 5),

5 assess the environmental consequences of the designed strategies for integrated manure management (Chapter 6).

We used life cycle assessment as a tool to compute the environmental impact of current technologies (Chapter 2 - 4) and future strategies (Chapter 6) for manure management. We quantified the following environmental impacts with contributing compounds: climate change (CC) in kg CO_2-eq (including CO_2, CH_4, and N_2O); terrestrial acidification (TA) in kg SO_2-eq (including NH_3, NO_x, and SO_2); marine eutrophication (ME) in kg N-eq (including NH_3, NO_x, and NO_3^-); Freshwater Eutrophication (FE) in kg P-eq (including PO_4^{3-}); particulate matter formation (PMF) in kg PM_{10}-eq (including: NH_3, NO_x, and SO_2 as precursors of particulate matter); fossil fuel depletion (FFD), in kg oil-eq; and land use, in m^2. Additionally, in Chapter 5, we calculated N use efficiency (NUE) as the ratio of N taken up by the crop to the N excreted by the animal, and the P_2O_5 over application rate (POA) relative to the demanded N by the crop, as an indicator for the soil P surplus.

In Chapter 2, we found that mono-digestion of liquid pig manure reduced the environmental impact as compared to conventional manure management without digestion, but that the production of bio-energy was low. Co-digestion with substrates competing with animal feed, like maize silage, beet tails, and wheat yeast concentrate increased bio-energy production >550%[1] compared to mono-digestion, but also increased the environmental impact up to a factor 78. TA, PMF, and FE, however, were reduced up to a factor 22 mainly from avoided production of fossil-based electricity and heat. Because these substrates were used in co-digestion instead as animal feed, a substitute was needed for animal feed causing pollution swapping. This was partly caused by land use change emissions that increased climate change and land use. Using waste products and residues, such as roadside grass instead of 'high-grade products', resulted in the lowest environmental impact. These products did not compete with other markets, but represented management with lower environmental impact, here co-digestion of roadside grass instead of composting.

In Chapter 3, results showed that the processing of liquid manure by separation into a solid and a liquid fraction, and further de-watering of the liquid fraction, increased the environmental impact up to 110% compared to current practice without processing. This increase was mainly caused by the processing and storage of the solid fraction which increased NH_3 and NO_x emissions. ME was the same and CC with liquid cattle manure was reduced by 67%. Including anaerobic mono-digestion of the separated solid fraction reduced CC with 117% and FFD with 59%. This effect was stronger when excess heat was utilized to avoid fossil-based heat.

In Chapter 4, results showed that separating pig urine and feces directly after excretion (called segregation) reduced CC, TA, and PMF with 82%, 49%, and 49%, respectively compared to current liquid manure management. Storage and field application

[1] When a system, for example, produces energy or avoids more energy than it uses, the difference with a reference system that uses energy becomes larger than 100%. Similarly, this occurs when more emissions are avoided in the system compared to the reference. Fractions larger than 1000% are given as factors.

of feces with a high dry matter content caused higher environmental impact (up to 94%) than feces with a low dry matter content. Segregation was found to provide a sound basis for environmentally sustainable manure management.

In Chapter 5, we applied a structured design approach to create new strategies for integrated manure management that prevent pollution swapping and reduce environmental impact throughout the manure management system. The design approach adapted from engineering design, proved to be useful for structuring the design process and to provide insight into interactions of emission processes and functions in the system and as such to create new strategies.

In Chapter 6, we found that compared to current North Western European practice, the strategies for integrated pig and dairy cattle manure management reduced environmental impact and increased NUE. It was possible to simultaneously reduce CC, up to 176%; FFD, >700%; TA, up to 92%; ME, up to 98%; PMF, up to 95%; POA, up to 103%; and more than double the NUE, up to 69%. In the IS, the fraction of excreted N lost to the environment was reduced to 31 - 57% as compared to 67 - 82% in current practice. Lower N losses also reduced the need for N from mineral fertilizer meaning more efficient use of nutrients from manure.

In Chapter 7, the main findings of the studies and the implications of the findings for future developments in manure management in Europe and the Netherlands were discussed. It was shown that policy regulations, such as the National Emission Ceilings and Nitrates Directives have until now been successful in reducing pollutants, such as NH_3 and NO_3^-, including those stemming from manure management. The regulations, however, also have possible undesired side effects that are expressed in pollution swapping to increased greenhouse gas emissions from, for example, injection of manure. Moreover, it may stimulate unnecessary relocation of nutrients when emissions are reduced and application of N is limited, mainly in areas with a nutrient surplus. In such a case, lower emissions of N lead to increased N concentration in manure products that requires more N to be exported. Current regulations may therefore hinder the implementation of initiatives for more efficient use of nutrients from manure.

CONCLUSIONS

From this thesis, we conclude that:

/ Current technologies for manure management have potential to reduce environmental consequences, but also induce pollution swapping causing environmental impact to increase elsewhere inside or outside the manure management system.

/ Newly designed strategies for integrated manure management prevent pollution swapping and have the potential to reduce the environmental consequences throughout the manure management system.

Specifically, we conclude that:

/ Mono-digestion of liquid manure reduces the environmental impact compared to conventional manure management, but represents a limited source for bio-energy. Co-digestion with waste and residues, such as roadside grass, increases bio-energy production and further reduces the environmental impact. Co-digestion with substrates that compete with animal feed, increases bio-energy, but also the overall environmental impact from producing a substitute for the used co-substrate.

/ Separating liquid manure into liquid and solid fractions and further de-watering of the liquid fraction increases the environmental impact compared to manure management where no processing occurs. When anaerobic mono-digestion of the solid faction is included, this processing method leads to an environmental benefit for climate change and fossil fuel depletion.

/ Segregating fattening pig urine and feces in the housing system reduces climate change, terrestrial acidification, and particulate matter formation and forms a sound basis for environmentally friendly manure management.

/ Applying a structured design approach, enables the design of new strategies for integrated manure management that prevent pollution swapping and thus reduce environmental impact throughout the manure management system.

/ The designed strategies for integrated manure management prevent pollution swapping and reduce the environmental impact throughout the manure management chain by at least 57% and more than double the nitrogen use efficiency compared to current North Western European manure management practices.

/ SAMENVATTING

Dierlijke mest is een belangrijke verbindende schakel tussen dierlijke en plantaardige productie. Mest bevat waardevolle nutriënten voor de bodem en het gewas, zoals stikstof (N), fosfor (P), kalium (K), en koolstof (C). Mest is echter ook een belangrijke bron van milieuvervuiling. Met name in gebieden met een hoge vee-dichtheid, zoals in delen van de Europese Unie, leidt de opslag en de toediening van mest tot verliezen van nutriënten en C naar lucht, bodem en water. Belangrijke verliesposten zijn: uit- en afspoeling van nitraat (NO_3^-) en fosfaat (PO_4^{3-}) naar het grond- en oppervlaktewater, hetgeen leidt tot eutrofiëring en humane gezondheidsproblemen; emissie van broeikasgassen, zoals koolstofdioxide (CO_2), methaan (CH_4) en lachgas (N_2O), hetgeen leidt tot klimaatverandering; en ammoniakemissie (NH_3), hetgeen leidt tot verzuring en eutrofiëring. Mestmanagement omvat een aantal stappen vanaf de uitscheiding van mest door het dier tot en met de opname van nutriënten uit mest door de plant, te weten: het verzamelen en de opslag van mest in de stal, de opslag buiten de stal, de verwerking, het transport en de toediening in het gewas. Technieken die zijn ontwikkeld om emissies naar het milieu tijdens mestmanagement te verlagen hebben zich vaak gericht op één emissie, zoals NH_3, N_2O of $CH_{4,}$ of op één stap in de keten, zoals opslag of toediening. Deze technieken veroorzaken veelal een verschuiving tussen verschillende emissies, een fenomeen dat 'pollution swapping' oftewel afwenteling wordt genoemd. Dit betekent dat één emissie wordt verlaagd, terwijl tegelijkertijd een andere emissie wordt verhoogd. Het vermijden van afwenteling is complex omdat verschillende onderliggende emissieprocessen en interacties tussen die processen een rol spelen. Onderzoek hiernaar vereist een structurele aanpak die alle functies en onderliggende processen, die leiden tot emissie in de hele mestketen, in kaart brengt. Van hieruit kunnen nieuwe strategieën voor geïntegreerd mestmanagement worden ontworpen en kan afwenteling worden voorkomen. Deze ontwerpen dienen vervolgens kwantitatief onderbouwd te worden om inzicht te verschaffen in de daadwerkelijke verlaging van de milieubelasting. Naast deze nieuwe ontwerpen zijn er recentelijk technieken ontwikkeld, zoals mestscheiden en het ontwateren van de dunne fractie, anaerobe vergisting van mest met co-substraten voor het produceren van bio-energie en andere scheidingsmethoden. Het is echter momenteel nog onbekend hoe de milieubelasting verandert door het gebruik van deze technieken en deze technieken dienen daarom geëvalueerd te worden. Kennis en inzicht in de milieubelasting van deze technieken kan vervolgens gebruikt worden voor het (her) ontwerpen van nieuwe strategieën voor mestmanagement. Het doel van dit proefschrift was daarom het verschaffen van kennis en inzicht in de gevolgen voor het milieu van het gebruik van huidige technieken en toekomstige strategieën voor mestmanagement. De deeldoelstellingen waren:

1 het analyseren van de verandering in de milieubelasting door de productie van bio-energie met anaerobe vergisting van varkensmest (mono-vergisting) en co-substraten (co-vergisting) inclusief de veranderingen ten gevolge van het produceren van een substituut voor de co-substraten (Hoofdstuk 2),

2 het analyseren van de verandering in de milieubelasting door high-tech scheiden van varkens- en rundveedrijfmest en het gebruik van de geproduceerde mestproducten als kunstmestvervanger (Hoofdstuk 3),

3 het analyseren van de verandering in de milieubelasting door het gescheiden houden van varkensurine- en feces in de stal (Hoofdstuk 4),

4 het ontwerpen van strategieën voor geïntegreerd mestmanagement die afwenteling voorkomen en de milieubelasting verminderen in de hele mestmanagementketen (Hoofdstuk 5),

5 het analyseren van de verandering in de milieubelasting door de nieuwe ontworpen strategieën voor geïntegreerd mestmanagement (Hoofdstuk 6).

De milieubelasting van huidige en toekomstige strategieën voor mestmanagement zijn doorgerekend met een levenscyclusanalyse (LCA). De volgende milieueffecten zijn daarbij gekwantificeerd: klimaatverandering (KV) in kg CO_2-eq (met CO_2, CH_4 en N_2O); terrestrische verzuring (TV) in kg SO_2-eq (met NH_3, NO_x en SO_2); mariene eutrofiering (ME) in kg N-eq (met NH_3, NO_x en NO_3^-); zoetwater eutrofiering (ZE) in kg P-eq (met PO_4^{3-}); fijn stof vorming (FSV) in kg PM_{10}-eq (met NH_3, NO_x, en SO_2 als precursors van fijn stof); fossiel energie verbruik (FEV), in kg olie-eq; en land gebruik, in m^2. Aanvullend zijn in Hoofdstuk 5 de N gebruiksefficiëntie (NE) en het P_2O_5 overschot (POS) gekwantificeerd. NE is uitgedrukt als fractie van de uitgescheiden N die opgenomen is door het gewas, en POS is gerelateerd aan de N behoefte van het gewas.

In Hoofdstuk 2 bleek dat mono-vergisten van varkensdrijfmest de milieubelasting verlaagde ten opzichte van huidig mestmanagement zonder mestvergisting. Daarentegen was de productie van bio-energie door mono-vergisting laag. Co-vergisting met co-substraten verhoogde de productie van bio-energie >550%[1] vergeleken met mono-vergisting. Het gebruik van co-substraten die ook als veevoeder worden gebruikt, zoals mais-silage, bietenstaartjes en tarwegistconcentraat, verhoogde de milieubelasting met maximaal een factor 78 ten opzichte van mono-vergisting. TV, FSV en ZE werden echter verlaagd met maximaal een factor 22 , met name door de vermeden productie van fossiele energie. De toename van milieubelasting ontstond door het gebruik van cosubstraten voor co-vergisting dat leidde tot de productie van een substituut voor het initiële gebruik van de cosubstraten. De productie van deze substituten zorgde voor afwenteling, onder andere door veranderingen in landgebruik. Veranderingen in landgebruik verhoogde de emissie van broeikasgassen. Het gebruik van afvalproducten of residuen, zoals bermgras als co-substraat, in plaats van hoogwaardige producten, verlaagde de milieubelasting. Dergelijk producten concurreren namelijk niet met toepassingen, zoals veevoeder, maar vertegenwoordigen management met een lagere milieubelasting. In dit geval het co-vergisten in plaats van het composteren van bermgras.

In Hoofdstuk 3 werd aangetoond dat het verwerken van varkens- en rundveedrijfmest door het scheiden in een dikke en dunne fractie en het verder ontwateren van de

[1] Als in een systeem bijvoorbeeld energie wordt geproduceerd of meer wordt vermeden dan wordt verbruikt, dan wordt het verschil met een referentie, die energie verbruikt, groter dan 100%. Op dezelfde wijze kan er in een systeem meer emissie worden vermeden ten opzichte van een referentie. Percentages groter dan 1000% zijn als veelvoud uitgedrukt.

dunne fractie, de milieubelasting verhoogde tot 110% ten opzichte van het huidige mestmanagement. Deze stijging werd met name veroorzaakt door het scheiden van drijfmest en de opslag van de dikke fractie dat leidde tot meer emissie van NH_3 en NO_x. ME bleef gelijk met en zonder verwerking. Bij het verwerken van rundveedrijfmest daalde KV met 67%. Door het toepassen van anaerobe vergisting van de dikke fractie daalden KV en het FEV met respectievelijk 117% en 59%. Dit effect werd groter wanneer het exces aan warmte werd gebruikt om fossiele warmte te vermijden.

In Hoofdstuk 4 werd aangetoond dat het gescheiden houden van varkensurine- en feces de milieubelasting verlaagde vergeleken met huidig mestmanagement. KV, TV en FSV daalden met 82%, 49% en 49%, respectievelijk. De opslag en toediening van feces met een hoog drogestofgehalte verhoogde de milieubelasting (tot 94%) ten opzichte van feces met een laag drogestofgehalte. Het gescheiden houden van urine en feces voorziet in een solide basis voor duurzaam mestmanagement.

In Hoofdstuk 5 hebben we nieuwe strategieën ontworpen voor geïntegreerd mestmanagement die afwenteling voorkomen in de gehele mestmanagementketen. Dit is gedaan met behulp van methodisch ontwerpen. Deze methode werd aangepast voor onze specifieke situatie en bleek zeer bruikbaar voor het structureren van het ontwerpproces. Het gaf tevens inzicht in de interacties van onderliggende emissieprocessen en in de oplossingen die nodig zijn in de nieuwe strategieën.

In Hoofdstuk 6 is de milieubelasting van de nieuwe strategieën voor geïntegreerd mestmanagement gekwantificeerd ten opzichte van de huidige mestmanagement-praktijken in Noord West Europa. Voor alle indicatoren werd de milieubelasting gelijktijdig verlaagd: KV tot 176%, TV tot 92%, ME tot 98%, FSV tot 95% en POS tot 103%. Tegelijk bleek de NE meer dan verdubbeld tot 69%, hetgeen betekent dat minstens twee keer zoveel N werd opgenomen door het gewas. Dientengevolge werd de uitgestoten N naar het milieu teruggebracht naar 31 - 57% ten opzichte van 67 - 82% in de huidige praktijk. De lagere emissie van N leidde ook tot minder behoefte aan N uit kunstmest omdat de nutriënten uit de mest efficiënter werden gebruik in de nieuwe ontworpen strategieën.

In Hoofdstuk 7 zijn de bevindingen uit het proefschrift en de implicaties hiervan bediscussieerd in het kader van toekomstige ontwikkelingen van mestmanagement in Noord West Europa en in Nederland. Er is beargumenteerd dat de huidige wet- en regelgeving, zoals de Nationale Emissieplafonds (NEC) en de Nitraatrichtlijn effectief zijn geweest in het reduceren van onder andere NH_3 en NO_3^- emissies uit het gebruik van mest. Daarentegen heeft deze wet- en regelgeving mogelijk ook ongewenste effecten, hetgeen tot uitdrukking komt in de afwenteling naar broeikasgassen uit mestmanagement door bijvoorbeeld mestinjectie. Maar ook veroorzaakt het onnodige distributie van nutriënten uit mest wanneer emissies in de keten worden beperkt en de toediening van N beperkt is, met name in de situatie van een overschot. In dit geval wordt er door lagere emissies in de keten meer N in de mest gehouden en dient er meer N te worden afgevoerd. Huidige wet en regelgeving kan daardoor belemmerend werken op initiatieven die efficiënter gebruik willen maken van nutriënten uit mest.

CONCLUSIES

Op basis van de resultaten in dit proefschrift wordt geconcludeerd dat:

/ Huidige technieken voor mestmanagement de potentie hebben om de gevolgen voor het milieu te verminderen, maar tegelijk leiden tot afwenteling of pollution swapping. Deze afwenteling kan leiden tot een verhoogde milieubelasting ergens anders binnen of buiten de mestketen.

/ Nieuwe ontworpen strategieën voor geïntegreerd mestmanagement afwenteling voorkomen en de gevolgen voor het milieu verlagen in de gehele mestketen.

Voorts blijkt dat:

/ Mono-vergisten van drijfmest de milieubelasting verlaagt vergeleken met conventioneel mestmanagement. De productie van bio-energie uit mono-vergisten is echter beperkt. Co-vergisten van drijfmest met afvalproducten en residuen, zoals bermgras, verhoogt de productie van bio-energie en verlaagt de milieubelasting. Co-vergisten met cosubstraten, die concurreren met diervoeders, verhoogt weliswaar de productie van bio-energie, maar ook de milieubelasting door de benodigde productie van een substituut voor het gebruikte cosubstraat.

/ Het scheiden van drijfmest in een dikke en dunne fractie en het verder ontwateren van de dunne fractie verhoogt de milieubelasting vergeleken met mestmanagement zonder deze verwerking. Het toevoegen van anaerobe mono-vergisting van de dikke fractie leidt tot een milieuvoordeel voor broeikasgasemissie en fossiel energieverbruik.

/ Het gescheiden houden van varkensurine- en feces in de stal verlaagt de emissie van broeikasgassen, terrestrische verzuring en de vorming van fijn stof. Daarmee voorziet deze techniek in een solide basis voor duurzaam mestmanagement.

/ Het toepassen van methodisch ontwerpen leidt tot nieuwe strategieën voor geïntegreerd mestmanagement welke afwenteling tussen emissies voorkomen en de potentie hebben om de milieubelasting in de gehele mestketen te verlagen.

/ Na kwantificering, de nieuwe ontworpen strategieën voor geïntegreerd mestmanagement daadwerkelijk afwenteling voorkomen en de milieubelasting in de gehele mestketen met minimaal 57% verminderen. Daarnaast stijgt de stikstofgebruiksefficiëntie met meer dan een factor twee vergeleken met de huidige Noord West Europese praktijk van mestmanagement.

/ GEARFETTING

Dierlik dong is in wichtige ferbinende skeakel tusken dierlike en planteftige produksje. Dong befettet weardefolle nutriïnten foar de boaiem en it gewaaks, lykas stikstof (N), fosfor (P), kalium (K), en koalstof (C). Dong is lykwols ek in belangrike boarne fan miljeufersmoarging. Benammen yn gebieden dêr't in protte fee is, lykas yn dielen fan de Europeeske Uny, laat de opslach en de tatsjinning fan dong ta ferliezen fan nutriïnten en C nei loft, boaiem en wetter. Belangrike ferliesposten binne: út- en ôfspieling fan nitraat (NO_3^-) en fosfaat (PO_4^{3-}) nei it grûn- en oerflaktewetter, itjinge laat ta eutrofearring en humane sûnensproblemen; emisje fan broeikasgassen, lykas koalstofdiokside (CO_2), metaan (CH_4) en gniisgas (N_2O), itjinge laat ta klimaatferoaring; en ammoniakemisje (NH_3), itjinge laat ta fersuorring en eutrofearring. Dongmanagement omfettet in oantal stappen fanôf de útskieding fan dong troch it bist oant en mei de opname fan nutriïnten út dong troch de plant, te witten: it sammeljen en de opslach fan dong yn' e stâl, de opslach bûten de stâl, de ferwurking, it transport en de tatsjinning yn it gewaaks. Techniken dy't ûntwikkele binne om emisjes nei it miljeu tidens dongmanagement te ferleegjen binne faak rjochte op ien emisje, lykas NH_3, N_2O of CH_4, of op ien stap yn de keatling, lykas opslach of tatsjinning. Dy techniken feroarsaakje alfolle in ferskowing tusken ferskate emisjes, in fenomeen dat 'pollution swapping' oftewol 'ôfwinteling' neamd wurdt. Dat betsjut dat ien emisje ferlege wurdt, wylst tagelyk in oare emisje ferhege wurdt. It mijen fan ôfwinteling is kompleks om't ferskate ûnderlizzende emisjeprosessen en ynteraksjes tusken dy prosessen in rol spylje. Ûndersyk hjirnei fereasket in strukturele oanpak dat alle funksjes en ûnderlizzende prosessen, dy't late ta emisje yn de hiele dongkeatling, yn kaart bringt. Fan hjirút kinne nije strategyen foar yntegrearre dongmanagement ûntwurpen wurde en ôfwinteling foarkommen wurde. Dy ûntwerpen moatte dêrnei kwantitatyf ûnderboud wurde om ynsicht te krije kinnen yn de feitlike ferleging fan it belêstigjen fan it miljeu. Neist dizze nije ûntwerpen binne der okkerdeis techniken ûntwikkele, lykas dongskieding en it ûntwetterjen fan de floeibere fraksje, anaërobe fergêsting fan dong mei ko-substraten foar it produsearjen fan bio-enerzjy en oare skiedingsmetoaden. It is mominteel noch ûnbekend hoe't it belêstigjen fan it miljeu feroaret troch it brûken fan dy techniken en dy techniken moatte dêrom evaluearre wurde. Kennis en ynsicht yn de miljeubelêsting fan dy techniken kinne dêrnei brûkt wurde foar it (wer)ûntwerpen fan nije strategyen foar dongmanagement. It doel fan dit proefskrift wie dan ek om ynsicht en kennis te krijen yn de gefolgen foar it miljeu fan it gebrûk fan hjoeddeiske techniken en takomstige strategyen foar dongmanagement. De doelstellingen wienen:

1. It analysearjen fan de feroaring yn de miljeubelesting troch de produksje fan bio-enerzjy mei anaërobe fergêsting fan bargedong (mono-fergêsting) en ko-substraten (ko-fergêsting) ynklusyf de feroaringen as gefolch fan it produsearjen fan in substitút foar de ko-substraten (haadstik twa),

2. It analysearjen fan de feroaring yn' e miljeubelêsting troch it high-tech skieden fan barge- en ko-feedriuwdong en it gebrûk fan de produsearre dongprodukten as keunstdongferfanger (haadstik trije),

3. It analysearjen fan de feroaring yn de miljeubelêsting troch it skieden hâlde fan de barge-urine en bargedong yn ’e stal (haadstik fjouwer),

4. It ûntwerpen fan strategyen foar yntegrearre dongmanagement dy’t ôfwinteling foarkomme en de miljeubelêsting ferminderje yn de hiele dongmanagementkeatling (haadstik fiif),

5. It analysearjen fan de feroaring yn de miljeubelêsting troch de nij ûntwurpen strategyen foar yntegrearre dongmanagement (Haadstik seis).

De miljeubelêsting fan hjoeddeiske en takomstige strategyen foar dongmanagement binne trochrekkene mei in libbenssyklus-analyze (LCA). De folgjende miljeu-effekten binne derby kwantifisearre: klimaatferoaring (Kf) yn kg CO_2-eq (mei CO_2, CH_4 en N_2O); terrestryske fersuorring (TF) yn kg SO_2-eq (mei NH_3, NO_x en SO_2); marine eutrofierring (ME) in kg N-eq (mei NH_3, NO_x en NO_3^-); swietwetter eutrofearring (SE) yn kg P-eq (mei PO_4^{3-}); fyn stof foarming (FSF) yn kg PM_{10}-eq (mei NH_3, NO_x, en SO_2 as prekursors fan fyn stof); fossyl enerzjyferbrûk (FEF), in kg oalje-eq; en lângebrûk, yn m^2. Bykommend binne yn haadstik fiif de N gebrûkseffisjinte (NE) en it P_2O_5 oerskot (POS) kwantifisearre. NE is útdrukt as in fraksje fan de útskieding N dy’t opnaam is troch it gewaaks, en POS is ferbûn oan de N-ferlet fan it gewaaks.

Yn haadstik twa bliek dat mono-fergêsting fan bargedriuwdong de miljeubelêsting ferlege oangeande hjoeddeistich dongmanagement sûnder dongfergêsting. Deroer wie de produksje fan bio-enerzjy troch mono-fergêsting leech. Ko-fergêsting mei ko-subtraten ferhege de produksje fan bio-enerzjy >550%[1] ferlike mei mono-fergêsting. It gebrûk fan ko-substraten dy’t ek as feefoer brûkt wurde, lykas mais-silaazje, bytsturten en weetgêstkonsintraat, ferhege de miljeubelesting mei maksimaal in faktor 78 oangeande mono-fergêsting. TF, FSF en SE waarden ferlege mei maksimaal in faktor 22, benammen troch de mijde produksje fan fossile enerzjy. It tanimmen fan de miljeubelêsting is ûntstien troch it gebrûk fan ko-substraten foar ko-fergêsting, dat late ta de produksje fan in substitúsje foar it inisjele gebrûk fan de ko-substraten. De produksje fan dizze substituten soarge foar ôfwinteling, ûnder oaren troch feroaringen yn lângebrûk. Feroaringen yn lângebrûk ferhegen de emisje fan broeikasgassen. It gebrûk fan ôffalprodukten of residuën lykas bermgers as ko-substraat, yn plak fan heechweardige produkten, hat de miljeubelesting ferlege. Soksoartige produkten konkurrearje nammentlik net mei tapassings lykas feefoer, mar fertsjintwurdigje management mei in legere miljeubelêsting. Yn dit gefal it ko-fergêsten yn plak fan it kompostearjen fan bermgers.

Yn haadstik trije waard sjen litten dat it ferwurkjen fan barge- en ko-feedriuwdong troch it skieden yn in floeibere en fêste fraksje en it fierder ûntwetterjen fan de floeibere fraksje, de miljeubelêsting ferhege oant 110% oangeande it hjoeddeiske dongmanagement. Dizze stiging waard benammen feroarsake troch it skieden fan driuwdong en de opslach fan de floeibere fraksje wat laat hat ta mear emisje fan NH_3 en NO_x. ME bleaun itselde mei en sûnder ferwurking. By it ferwurkjen fan ko-feedriuwdong sakke KF mei 67% Troch it tapassen fan anaërobe fergêsting fan de floeibere fraksje sakke

[1] As der yn in systeem bygelyks enerzjy produsearre wurdt, of mear fermijd dan ferbrûkt, dan wurdt it ferskil mei in referinsje, dy’t enerzjy ferbrûkt grutter as 100%. Op deselde wize kin der yn in systeem mear emisjes fermijd wurde oangeande in referinsje. Persintaazjes grutter as 1000% binne as mearfâld útdrukt.

KF en it FEF mei respektivelik 117% en 59%. Dit effekt waard grutter wannear't it ekses oan waarmte brûkt waard om fossile waarmte te omsilen.

Yn haadstik fjouwer waard oantoand dat it skieden hâlde fan barge-urine en bargetrochgong de miljeubelêsting ferlege yn ferliking mei it hjoeddeiske dongmanagement. KF, TF en FSF sakken mei maksimaal 82%, 49% en 49%, respektivelik. De opslach en tatsjinning fan trochgong mei in heech droege stof gehalte ferhege de miljeubelêsting (mei 94%) oangeande trochgong mei in leech droege stof gehalte. It skieden hâlden fan urine en trochgong soarget foar in solide basis foar duorsum dongmanagement.

Yn haadstik fiif hawwe wy nije strategyen ûntwurpen foar yntegrearre dongmanagement dat ôfwinteling foarkomt yn de dongmanagementkeatling. Dit is dien mei behelp fan metoadysk ûntwerpe. Dy metoade waard oanpast foar ús spesifike situaasje en it bliek hiel brûkber te wêzen foar it strukturearjen fan it ûntwerpproses. Ek joech it ynsicht yn de ynteraksjes fan ûnderlizzende emisjeprosessen en yn de oplossingen dy't nedich binne yn de nije strategyen.

Yn haadstik seis is de miljeubelêsting fan de nije strategyen foar yntegrearre dongmanagement kwantifisearre oangeande de hjoeddeiske dongmanagement-praktiken yn Noardwest Europa. Foar alle yndikatoaren waard de miljeubelêsting tagelyk ferlege: KF oant 176%, TF oant 92%, ME oant 98%, FSF oant 95% en POS oant 103%. Tagelyk bliek de NE mear as ferdûbele oant 69%, itjinge betsjut dat op syn minstens twa kear safolle N opnaam waard troch it gewaaks. Fan gefolgen waard de útstjitte N nei it miljeu werombrocht nei 31 - 57% oangeande 67-82% yn de hjoeddeistige praktyk. De legere emisje fan N late ek ta minder ferlet oan N út keunstdong om't de nutriïnten út de dong effisjinter brûkt waarden yn de nij ûntwurpen strategy.

Yn haadstik sân binne de befiningen út it proefskrift en de ymplikaasjes hjirfan bediskusearre yn it ramt fan takomstige ûntwikkelingen fan dongmanagement yn Noardwest Europa en Nederlân. Der is beärgumintearre dat de hjoeddeistige wet- en rigeljouwing, sa't it 'Nationale Emissieplafonds (NEC) en de 'Nitraatrichtlijn' effektyf west binne yn it redusearjen fan ûnder oaren NH_3 en NO_3^- emisjes út it gebrûk fan dong. Dêrfoaroer hawwe dizze wetten en rigels mooglik ek negative effekten, itjinge ta útdrukking komt yn de ôfwinteling nei broeikasgassen út dongmanagement troch bygelyks dongynjeksje. Ek feroarsaket it ûnnedige distribúsje fan nutriïnten út dong wannear't emisjes yn de keatling beheind wurde en de tatsjinning fan N beheind is, benammen yn de situaasje fan in oerskot. Yn dat gefal wurdt der troch legere emisje yn de keatling mear N yn de dong holden en moat der mear N ôffierd wurde. De hjoeddeiske wet- en rigeljouwing kin dêrtroch belemmerjend wurkje op inisjativen dy't effisjinter gebrûk meitsje wolle fan nutriïnten út dong.

KONKLÚZJES

Op basis fan de resultaten yn dit proefskrift wurdt konkludearre dat:

/ Hjoeddeiske techniken foar dongmanagement de potinsje hawwe om de gefolgen foar it miljeu te ferminderjen, mar tagelyk late ta ôfwinteling of 'pollution swapping'. Dy ôfwinteling kin late ta in ferhege miljeubelêsting op in oar plak of bûten de dongkeatling.

/ Nije ûntwurpen strategyen foar yntegrearre dongmanagement foarkomme ôfwinteling en ferleegje de gefolgen fan it miljeu yn de hiele dongkeatling.

Ek blykt dat:

/ Mono-fergêsting fan driuwdong de miljeubelêsting ferleget yn ferliking mei konvinsjoneel dongmanagement. De produksje fan bio-enerzjy út mono-fergêsting is lykwols beheind. Ko-fergêsting fan driuwdong mei ôffalprodukten en residuën, lykas bermgers, ferheget de produksje fan bio-enerzjy en ferleget de miljeubelêsting. Ko-fergêsting mei ko-substraten, dy't konkurrearje mei bistefieding, ferheget wol de produksje fan bio-enerzjy, mar ferheget ek de miljeubelêsting troch de nedige produksje fan in substitút foar it brûkte kobsubstraat.

/ It skieden fan driuwdong yn in floeibere en in fêste fraksje en it fierder ûntwetterjen fan de fêste fraksje ferheget de miljeubelêsting yn ferliking mei dongmanagement sûnder dy ferwurking. It tafoegjen fan anaërobe mono-fergêsting fan de floeibere fraksje laat ta in miljeufoardiel foar broeikasgasemisje en fossyl enerzjyferbrûk.

/ It skieden hâlde fan barge-urine en bargetrochgong yn' e stâl ferleget de emisje fan broeikasgassen, terrestryske fersuorring en foarming fan fyn stof. Dy technyk is in solide basis foar duorsum dongmanagement.

/ It tapassen fan metoadysk ûntwerpe liedt ta nije strategyen foar yntegrearre dongmanagement hokker ôfwinteling tusken emisjes foarkomt en de potinsje hat om de miljeubelêsting yn de hiele dongkeatling te ferleegjen.

/ Nei kwantifisearring foarkomme de nij ûntwurpen strategyen foar yntegrearre dongmanagement ôfwinteling en fermindert de miljeubelêsting yn de hiele dongkeatling mei minimaal 57%. Ek stiget de stikstofgebrûkseffisjinsje mei mear as ien faktor twa yn ferliking mei de hjoeddeiske Noardwest Europeeske praktyk fan dongmanagement.

/ WORDS OF THANKS / DANKWOORD

I will start this section with noting that I never actually had the ambition to obtain a PhD degree or become a doctor. I didn't even know what a PhD was or what this meant and consequently that I would obtain such a degree one day. Nevertheless, I can only say that by discovering and following my heart, I found that I enjoy doing research and thinking about a problem (this still sounds strange to me!). In this way, researching and teaching relates to my being. To me, the title of doctor, therefore, is merely a reflection of what I have already received and this work represented a path to a deeper discovery of my own being. My praise goes to the one who I believe has blessed me with these things, my Father God. Thank you, it is the most wonderful thing to be Loved and Honored by You.

In any case, this thesis reflects the work of a majority of different people, including supervisors, family, and friends. They all contributed in the formation of the thesis in one way or the other. Some did this by challenging me to think about the research and others by supporting me in the process. But most of all, I could only do this by the grace and support of my lovely wife, Johanna. I'm very proud of you and our two great kids, Jonathan and Jaïr! I would also like to mention Guy and Do. Guy, I very much appreciate our talks and discussions. Thank you for being a father, and you and Do for the great support all the time.

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/ Peter: I learned from you to try and keep looking to the unconventional ways and solutions instead of highlighting something I already knew in the back of my head. This was especially important in the final paper about the design of strategies for integrated manure management. Thank you for motivating me to continue and look further than the existing things!

/ Imke: I learned a lot about writing from you and our discussions on the system boundaries of the LCA studies. Also your inspiration in making titles for the papers and the thesis was a delight! I appreciate your support in this. Thank you for all the textual comments.

/ Willem: I learned from you that it is good to stay relaxed under all conditions. With your sympathy and peace you always provided a good basis to fall back upon. Thanks!

- Karin: I learned some pretty nice things from you. For example: separate the means from the objective was one of these important things that helped me to structure my ideas. Another one: don't mention the information excluded from the study in your papers as it is not interesting anyway... Thank you a lot for these tips and tricks and your support during the process of writing and becoming a scientist!

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Jerke de Vries

Wageningen, November 2013.

/ CURRICULUM VITAE

Jerke Wiebe de Vries was born on the 11th of January 1982 in Damwoude, the Netherlands on a dairy farm. Prior to starting his studies at Wageningen University, he went to agricultural high school, joined the EH basisjaar, and obtained a propaedeutic in applied psychology at Saxion University of applied Sciences in Deventer, the Netherlands. In 2003, he started his study in agricultural technology in Wageningen and continued his MSc. in agricultural and bioresource engineering which he completed in 2008. His BSc. as well as his MSc. thesis were awarded with prizes. His two major thesis subjects were on: biogas production from anaerobic digestion and variation in digestate compositions from different manure-crop mixtures and digestion times; and the design, building, and testing of a no-tillage seeder for small-scale vegetable production in Saltillo, Coahuila, Mexico. After his MSc., he shortly worked at Kverneland Group as an engineer and started work as a researcher environment & LCA at Livestock Research in 2009. At the same time he initiated his PhD by writing a research proposal, writing grant proposals, and by finding a supervision committee. His PhD was completed in 2013 and provides insight into the environmental consequences of current and future strategies for manure management; the outcomes as described in this dissertation. Jerke is passionate about research and education and is looking forward to continue to work as a teacher and scientist.

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/ PUBLICATIONS

SCIENTIFIC PUBLICATIONS (PEER REVIEWED)

DE VRIES, J.W., Aarnink, A.J.A., Groot Koerkamp, P.W.G., De Boer, I.J.M., 2013.
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DE VRIES, J.W., Hoogmoed, W.B., Groenestein, C.M., Schröder, J.J., Sukkel, W., De Boer, I.J.M., Groot Koerkamp, P.W.G., submitted.
Integrated manure management to reduce environmental impact: I. Structured design of strategies
Submitted for publication

DE VRIES, J.W., Groenestein, C.M., Schröder, J.J., Hoogmoed, W.B., Sukkel, W., Groot Koerkamp, P.W.G., De Boer, I.J.M., submitted.
Integrated manure management to reduce environmental impact: II. Environmental impact assessment of strategies
Submitted for publication

PABON-PEREIRA, C.P., De Vries, J.W., Slingerland, M.A., Zeeman, G., Van Lier, J.B., in preparation.
Impact of crop-manure ratios and digestion time on the fertilizing characteristics of liquid and solid digestate during codigestion.
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CONFERENCE PROCEEDINGS AND PRESENTATIONS (PEER AND NON-PEER REVIEWED)

DE VRIES, J.W. and De Boer, I.J.M., 2013
Manure management technologies and mitigation of greenhouse gases: opportunities & limitations.
In: Book of abstract of Greenhouse Gases & Animal Agriculture (GGAA) conference, Dublin Ireland, 23-26 June 2013.

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In: A Greener Agriculture for a Bluer Baltic Sea 2012 conference, Copenhagen Denmark, 24-25 October 2012.

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KOOL, A.; Blonk, H.; Ponsioen, T.; Sukkel, W.; Vermeer, H.M.; De Vries, J.W.; Hoste, R. ,2009.
Carbon footprints of conventional and organic pork : assessments of typical production systems in the Netherlands, Denmark, England and GermanyCarbon footprints of conventional and organic pork : assessments of typical production systems in the Netherlands, Denmark, England and Germany.
Blonk Milieu Advies, Wageningen UR Livstock Research, Applied Plant Research, Agricultural Economics Institute, Gouda, Lelystad, The Hague, the Netherlands, p. 90.

/ PE&RC PHD TRAINING CERTIFICATE

With the educational activities listed below the PhD candidate has complied with the educational 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)

- Review for the research proposal - discussed with different groups within WUR: Farm Technology Group, Animal Production Systems Group, Livestock Research and Soil Quality Group

WRITING OF PROJECT PROPOSAL (4.5 ECTS)

- Moving beyond manure: integrating cropping systems and manure products and management to reduce environmental impact (2011)

POST-GRADUATE COURSES (7 ECTS)

- EASEWASTE modelling of solid waste systems with use of life cycle assessment; DTU, Copenhagen, Denmark (2009)
- Advanced course in consequential life cycle assessment; Aalborg University, Denmark (2010)

LABORATORY TRAINING AND WORKING VISITS (1.5 ECTS)

- Greenhouse gases and biogas production in the EU; KTBL, Germany (2009)
- Biogas production in the EU (part of EU Agro Biogas project); Aarhus University (2009)

INVITED REVIEW OF (UNPUBLISHED) JOURNAL MANUSCRIPT (2 ECTS)

- Journal of Environmental Management: biowaste treatment for reuse of nitrogen (2012)
- AMBIO: ecological rationality and manure management (2013)

COMPETENCE STRENGTHENING / SKILLS COURSES (4.5 ECTS)

- Projectmanagement: de weg naar projectsucces; WUR (2009)
- Writing for academic publication; Linda McPhee Consulting (2011)
- Techniques for writing and presenting scientific papers; WGS (2011)
- Expert in sales, basics, pricing and conversely sell; Kenneth Smit training (2011/2012)
- Acklas: trajectory for interacting with higher education from your own research perspective; WUR (2012)
- How to write a convincing research proposal; PE&RC (2013)

PE&RC ANNUAL MEETINGS, SEMINARS AND THE PE&RC WEEKEND (1.5 ECTS)
- PE&RC Weekend (2010)
- PE&RC Day (2011)
- WIAS Science day (2012)

DISCUSSION GROUPS / LOCAL SEMINARS / OTHER SCIENTIFIC MEETINGS (5.3 ECTS)
- LCA Discussion group; Animal Production Systems (2009-2013)
- Seminar; as preparation for proposal; Soil Quality (2010)

INTERNATIONAL SYMPOSIA, WORKSHOPS AND CONFERENCES (9 ECTS)
- LCA Food; poster presentation; Bari, Italy (2010)
- Managing livestock manure for sustainable agriculture; poster presentation; Wageningen, the Netherlands (2010)
- EurAgEng; oral presentation; Valencia, Spain (2012)
- EAAP; invited oral presentation; Bratislava, Slovak Republic (2012)
- Baltic manure conference; invited oral presentation; Copenhagen, Denmark (2012)
- GGAA; oral and poster presentation; financially supported by the HJS Fund; Dublin, Ireland (2013)

LECTURING / SUPERVISION OF PRACTICAL'S/ TUTORIALS; (3 ECTS)
- Livestock technology (2009-2013)
- Global & sustainable animal production in the 21st century (2011-2012)
- Advanced biosystems engineering (2012)

SUPERVISION OF 2 MSC STUDENTS (6 ECTS)
- Environmental impact of anaerobic mono- and co-digestion of pig manure
- Nutrient cycles of future integrated livestock production systems

/ COLOPHON

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