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Can spatial reallocation of livestock reduce the impact of GHG emissions?

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ABSTRACT

Historically, concentrated livestock production and, consequently, manure production and management have resulted in considerable environmental impacts in many parts of Europe. The region selected for the current case study was Belgium which is characterized by input-intensive animal production within a geographically concentrated land area. In this study, the effect of a reduction in manure pressure through spatial distribution of CO_2 equivalent emissions was investigated and the impact on the carbon footprint verified through a consequential life cycle approach. This was accomplished by investigating the marginal spatial impact on CO_2 emissions of a decrease in manure pressure. An economic and environmental optimization was conducted using mathematical linear programming and the main differences between both approaches determined. The results of the model simulations show that, while the economic optimum is achieved by maximizing the transport of raw manure until fertilization standards are fulfilled and subsequently processing the excess manure, the environmental optimum, from a carbon footprint point of view, is achieved by separating all manure, as this strategy causes the least CO_2 emissions, mainly due to the limited manure storage time. Moreover, the analyses indicate that rearrangement of the spatial distribution of livestock production in Belgium will not substantially decrease CO_2 emissions. As the study demonstrated that manure storage is the main contributor to the carbon footprint, solutions should instead be sought by changing these storage systems. This article contributes to the methodology of the consequential life cycle approach by linking carbon footprint analysis with an economic model that simulates manure disposal decisions driven by legal constraints and market forces.

1. Introduction

Intensive livestock production is widely regarded as having a detrimental impact on the environment (Sage, 2012; Steinfeld et al., 2006; Meers et al., 2005) due to livestock supply chains requiring significant inputs of feed, energy and water, production of CH_4, NH_3 and other emissions, and pollution risks arising from inefficient waste management practices (McAuliffe et al., 2016). While research into whole-system pig production indicates that feed production generates the greatest environmental pressure, on a localized scale, waste management becomes more problematic, with the main concerns being global warming from greenhouse gas (GHG) emissions, aquatic eutrophication and acidification from ammonia emissions (Lopez-Ridaura et al., 2009; Sandars et al., 2003). More specifically, large amounts of GHG emissions, such as CH_4 and N_2O, relating to manure storage and its application on crop land create a substantial environmental burden (Loyon et al., 2007; Lopez-Ridaura et al., 2009; Rigolot et al., 2010; De Vries et al., 2012). There is a need for a detailed assessment of overall environmental impacts from pig manure management, incorporating available technologies applied at different handling stages in order to reduce the environmental burden (Prapaspongsa et al., 2010).

One tool for assessing the environmental performance of complex systems, such as pig production, is life cycle assessment (LCA). This has often been applied in the case of pig production (McAuliffe et al., 2016). The LCA literature distinguishes two types of LCA: the attributional approach to environmental impact calculation (also called the accounting or descriptive approach) attempts to provide information on the share of global burden that can be associated with a product and its life cycle (Sonnemann and Vigon, 2011), while the consequential approach is designed to generate information on the consequences of actions (Ekvall and Weidema, 2004).

Livestock waste-related, and mostly attributional, LCAs have received widespread attention in the EU in recent years, possibly due to the Water Framework Directive targets in 2015. Waste management is the most localized concern for pig production, due to the N and P content of animal manure and, hence, technologies have been developed to reduce risks associated with traditional manure management techniques, such as anaerobic digestion, biological treatment of manure and manure separation (McAuliffe et al., 2016).

However, the existing literature reports conflicting results for the optimal solutions for pig waste management. According to McAuliffe

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et al. (2016), the general consensus from the research was that treated manure or slurry generated a lesser burden than untreated manure. There were, however, exceptions, such as Lopez-Ridaura et al. (2009), who found that traditional slurry spreading had less impact than aerobic treatment, while Bayo et al. (2012) suggested spreading was preferable to constructed wetlands. Moreover, since these types of studies apply LCAs of GHG emissions for specific areas and animal products, and use different approaches, scopes and functional units, it makes them very hard to compare and draw consistent conclusions (Weiss and Leip, 2012).

In Belgium, the main bottleneck for manure management is the strong geographical concentration of livestock and manure production in the province of West Flanders and the northern part of Antwerp (Van der Straeten and Buysse, 2013). To adhere to targets in the Water Framework and Nitrates Directive, raw manure is currently exported from zones with high manure pressure to zones with low pressure until fertilization standards are fulfilled, to minimize the economic cost, after which the manure surplus is processed. On the one hand, calls have been made to reduce the high manure pressure and related environmental effects by reducing, relocating and more evenly distributing livestock production (Werkgroep voor Rechtvaardige en Verantwoorde Landbouw, 2013). On the other hand, based on the literature, one could argue that a high livestock density increases manure processing and, therefore, reduces the environmental impact of manure management.

In order to come to a clear conclusion on the matter, in this study, we use the concept of consequential LCA (Ekvall and Weidema, 2004) to explore the spatial distribution of CO_2 equivalent (eq.) emissions from pig manure management in Belgium. The consequential LCA is selected in preference to an attributional approach because consequential modeling estimates how flows to and from the environment will change as a result of different potential decisions (Curran et al., 2005; Sonnemann and Vigon, 2011), such as, in this case, the spatial reallocation of livestock production. In this study, however, we do not conduct a complete LCA of all the flows created by manure management. First of all, we limit ourselves to those flows that contribute to the carbon footprint (CF), i.e. the GHG emissions from manure into the atmosphere in the form of CO_2, CH_4, and direct and indirect N_2O and NO_x. Secondly, since in Belgium pig production creates the greatest environmental pressure, only the GHG emissions from concentrated pig production are taken into account.

In this study, we answer the following question: 'Can spatial reallocation of livestock production in Belgium reduce the impact of GHG emissions?' This question is translated into three research objectives: 1) conduct an economic (cost minimization) and environmental (GHG minimization) optimization for three manure management strategies, which are, in this case, pig manure transport, treatment and separation, in Belgium, 2) determine the main differences between both approaches, and 3) determine the consequential CF of a decrease in manure pressure (i.e., wider distribution of pig production). As a basis for our calculations, we use a linear programming model that simulates manure disposal decisions driven by legal constraints and market forces, to which we link CF calculations in order to investigate the impact of spatial reallocation.

2. Methodology

In this section, we first describe the assumptions upon which the LCA calculations are based, followed by a description of the linear programming model in which we insert the LCA data and conduct the consequential LCA.

Fig. 1 provides an overview of the manure management system upon which the life cycle as well as the manure allocation calculations are based and explains how both approaches are combined. We will come back to this figure in the various sections of the methodology.

The basic assumption of the model is that different types of animals produce manure with a different nutrient content. The nutrient content can be altered by managing the manure in different ways, such as manure separation or biological treatment. To apply manure to the field, fertilization standards have to be adhered to. These standards depend on the crop type. The LCA calculations determine the environmental impact of each manure management strategy, focused here on GHG emissions, while the manure allocation model (MAM) allows us to determine the optimal spatial manure allocation depending either on economic optimization (allocation cost minimization) or on environmental optimization (CF minimization). It is important to note that, with regard to the environmental optimization, we only take into account the management of pig manure, while for the economic optimization we consider all existing livestock in Belgium.

2.1. Functional unit and system boundaries

The functional unit is the total amount of pig manure produced on an annual basis in each municipality in Belgium. System boundaries, indicated in Fig. 1 by the black dotted line, are set starting from manure production to the arrival of the (processed) manure at its final destination. The life cycle stages involved are manure production, storage, processing, transport and, finally, application to the land. The life cycle and boundaries of our assessment are presented in Fig. 1, together with a representation of the MAM (see Section 2.4). The system boundaries

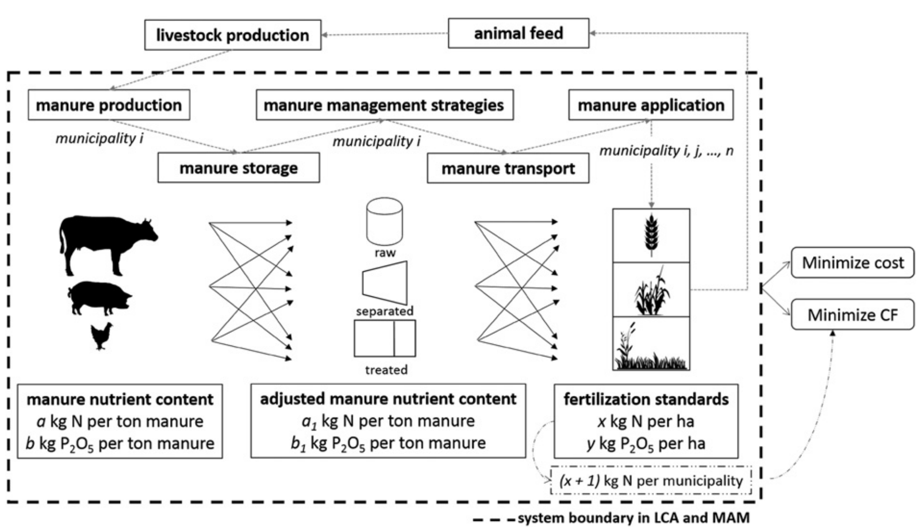

Fig. 1. Manure management system with system boundaries and manure management strategies.

exclude the production of capital goods, such as machinery and equipment, similar to most international studies. The CO_2 emissions from manure storage and processing are not taken into account because these emissions are considered part of the short carbon cycle, i.e. resulting from recent CO_2 uptake by crops. On the other hand, the emission of CO_2 originating from fossil energy use is taken into account.

Typically, animal manure is composed of different constituents, such as nutrients, organic matter, minerals, etc. (Coppens, 2009). These elements all circulate in their own life cycle and are only present in manure in specific parts of that life cycle (represented by the grey dotted arrows). Moreover, not all of these elements cause GHG emissions. Therefore, in our analysis, we limit ourselves to those elements and that part of the life cycle where GHG emissions from manure management occur, in the form of CO_2, CH_4, N_2O and NO_x. Fig. 1 illustrates this research restriction, where the presence of these elements in animal feed and animal production lies outside the system boundaries and is hence not taken into account in this study.

2.2. Data sources

The GHG emissions within the system boundaries of the three manure management strategies are determined for each municipality in Belgium. The emissions for each livestock category are calculated using the '2006 Intergovernmental Panel on Climate Change (IPCC) guidelines for national GHG inventories' (IPCC, 2006a, 2006b), reports on emissions and energy use and available country-specific data from the norms and guidelines of VLM (Vlaamse Landmaatschappij, 2014a), the Belgian National Inventory System (NIS) database and the National Inventory Report for Belgium (VMM et al., 2014).

The MAM is based on Van der Straeten et al. (2010) and Van Der Straeten et al. (2011), and implements the fertilization standards for total and animal-sourced nitrogen and total phosphorous within the Flemish manure legislation (Vlaamse Landmaatschappij, 2014a). The model is based on livestock quantities and crop surfaces at the municipal level from the 2012 Agricultural Survey from the Belgian Federal Public Services Economy, S.M.E.s, Self-employed and Energy (Federal Public Service Economy S.M.E.s Self-employed and Energy, 2012). Technical and economic data on manure management strategies are taken from the study on Best Available Techniques for Manure Processing (Lemmens et al., 2007), and an excerpt from the 'Mestwijzer' from the Belgian Soil Service (Coppens, 2009). Table 1 provides a brief summary of these data. The cost per ton for manure processing changes depending on the management strategy used.

2.3. Life cycle approach

As mentioned before, in this study, we look at selected life cycle stages for certain elements that are responsible for GHG emissions. We focus on the life cycle of those elements with respect to livestock, and hence, manure production.

The first step within this study is the calculation of the costs and environmental impact in terms of GHG emissions for the two traditional manure management strategies applied in Belgium, i.e. raw manure transport and biological treatment, and a third strategy, manure separation. The raw manure transport strategy includes on-farm storage, transport outside the pressure region to the spreading area and application to crop land, substituting mineral fertilizers. In Belgium, manure treatment consists of three important phases: (i) physical separation into a liquid (85%) and a solid (15%) fraction; (ii) composting of the solid fraction into an exportable product and (iii) reduction of the nutrient content in the liquid fraction through biological treatment (Meers et al., 2008). Hence, the manure treatment strategy includes storage, transport to the treatment plant, mechanical separation, biological treatment of the liquid fraction, transport and composting of the solid fraction, and transport and application of the effluent and the compost to crop land. The manure separation strategy includes storage, transport to a manure treatment plant where it is separated, intermediate storage of the separated fractions, transport and application of the liquid fraction to crop land, and composting and export of the solid fraction. To make the distinction between raw manure on the one hand, and treated and separated manure on the other hand, we refer to processed manure to describe manure treatment and separation at the same time.

The following subsections provide a summary of the emission sources relating to the manure management strategies.

2.3.1. Emissions from storage

Pig manure is stored as slurry in a pit under the pig unit and causes CH_4 and N_2O emissions (VMM et al., 2013; Jacobsen et al., 2014; IPCC, 2006a). Direct N_2O emissions are small in comparison to the large quantity of CH_4 emissions from pig slurry storage (Montes et al., 2013).

The calculations for raw manure are based on storage for six months before spreading, since the total storage capacity has to be sufficient to store at least the quantity of manure produced by the animals in the pig unit during a six month period (Lemmens et al., 2007; Vlaamse Regering, 2014). This capacity is necessary, since the time period during which manure can be applied is limited, according to the crop cycles and nutrient requirements. In Belgium, manure and other fertilizers can only be applied from mid-February until the end of August (Vlaamse Landmaatschappij, 2016). Production of CH_4 takes place during manure decomposition under anaerobic conditions (Jacobsen et al., 2014; IPCC, 2006a). The IPCC Tier 2 method is used to calculate the CH_4 output (VMM et al., 2014; IPCC, 2006a). Emissions of N_2O from stored manure are a consequence of nitrification and denitrification processes. Additionally, indirect N_2O emissions from manure are caused by volatilized NH_3 and NO_x which may be deposited at sites downwind from manure handling areas and contribute to indirect N_2O emissions (IPCC, 2006a; VMM et al., 2014).

It is assumed that the liquid fraction is stored for an average of four months before application to crop land, while the solid fraction is transferred to the composting installation almost immediately. Based on experiments and literature reviews, it is assumed that CH_4 emissions from the liquid fraction are twelve times lower in comparison to raw slurry for the same storage period, while the CH_4 emissions from the solid fraction are assumed to be negligible (Mosquera et al., 2010). Nitrous oxide emissions from the liquid fraction are assumed to be negligible due to anaerobic conditions which prevent nitrification (Mosquera et al., 2010; Petersen et al., 2013). The solid fraction, on the other hand, causes higher N_2O emissions, since there are more mineralization and nitrification processes due to the more aerobic environment. However, due to the short storage period, the N_2O emissions from the solid fraction are

Table 1
Summary of technical and economic data on manure management strategies.

Manure type	DM (kg/1000 L)	Thick fraction left after separation (ton)	Separation cost (€·ton^{-1} original manure)	Composting cost (€·ton^{-1} original manure)	Biological treatment cost (€·ton^{-1} original manure)
Calves and cows	85.7[a]	0.204[b]	2[c]	6.92[b]	7.92[b]
Fattening pigs	90[a]	0.220[b]	2[c]	7.50[b]	7.92[b]
Sows and breeding pigs	51.8[a]	0.070[b]	2[c]	2.38[b]	7.92[b]

[a] Coppens (2009).
[b] Calculations based on Lemmens et al. (2007).
[c] Lemmens et al. (2007).

assumed to be zero. Ammonia emissions are assumed to be inhibited by covering the storage tanks.

2.3.2. Emissions from manure processing

Emissions from manure processing consist of emissions from mechanical separation, biological treatment of the liquid fraction and composting of the solid fraction.

It is assumed that a centrifuge is used for the mechanical separation of the slurry since this is the most common technique used in Belgium. Most of the time, separation occurs within a closed device or within the pig unit. Therefore, emissions are expected to be minimal and the quantity of nutrients entering the system should be the same as the quantity leaving it (Lemmens et al., 2007; Melse et al., 2004).

There is considerable uncertainty about the N_2O emissions caused by nitrification and denitrification. Under well-controlled conditions, nitrogen losses of up to 1% of N_2O and 0.01% of NH_3 were measured at a full scale installation of a Trevi plant (Lemmens et al., 2007; Smet and Deboosere, 2007). The low NH_3 emissions are a result of the natural acidification of the activated sludge and the low concentration of NH_3 during the nitrification/denitrification process (Lemmens et al., 2007). The N_2O emissions are even lower than the emissions from raw manure applied to soil (Smet and Deboosere, 2007). Similarly, Loyon et al. (2007) found that less than 1% of the total nitrogen entering the treatment plant was emitted as N_2O. Methane emissions are assumed to be negligible (personal communication, BioArmor 2015). The high residual content of N, P and K in the effluent from the biological treatment plant is still too high to allow discharge in Belgium. However, the effluent can be applied to crop land. In practice, the effluent is applied locally on pasture and cropland as potassium fertilizer, since the amount of potassium is more or less equal to the amount found in raw slurry.

During composting, 28% of the nitrogen from the solid manure is emitted as NH_3-N, 1% as N_2O-N and 1% as N_2, which adds up to a total N-loss of 30% (Basset-Mens et al., 2007). Similarly, Lemmens et al. (2007) and Melse et al. (2004) mention a nitrogen reduction of 30% and 15 to 50% during the composting process respectively. However, manure composting occurs in closed systems (hall or tunnel composting) where the gases are captured and treated, and thus the emissions are considered to be zero. Through composting, 30% of the dry matter is broken down. To calculate the mass balance, only pig manure is taken into account, whereas, in practice, the solid fraction of manure is co-composted with chicken manure. By adding dry organic material, such as chicken manure, the C/N ratio is increased, which is necessary for the composting process.

2.3.3. Emissions from manure application

Direct N_2O emissions and indirect N_2O emissions from NH_3 volatilization from managed soils can be derived using the IPCC equations (IPCC, 2006b).

In comparison to raw manure, separation has no influence on the emission of NH_3 from the liquid fraction or the solid fraction when applied to grassland and cropland respectively. This is also the case for N_2O emissions from the liquid fraction when applied to grassland (Mosquera et al., 2010). Consequently, the same emission factors are used as for the application of raw manure.

2.3.4. Emissions from non-renewable energy use

The use of non-renewable energy includes the energy used for transport and injection of the manure slurry, and for manure processing.

It is assumed that slurry transport occurs by truck with a load of over 20 tons and an emission of 110 g CO_2 eq. $ton^{-1}.km^{-1}$ (Skao et al., 2011). A fuel use of 2.49 and 0.8 L of diesel per m^3 manure is assumed for applying the slurry to the land and injecting it, respectively (Lopez-Ridaura et al., 2009). According to Defra (2012), well-to-wheel greenhouse gas emissions for the combustion of 1 L of diesel equal 3.18 kg CO_2 eq. Furthermore, energy is necessary for separation, biological treatment and composting. The emission factor for electricity in Belgium is 400 kg CO_2.MW h^{-1} (Commissie Benchmarking, 2009). Moreover, we assume that, after manure separation, the thick fraction is routinely transported to a composting facility at an average distance of 50 km, after which the composted, hygienised fraction is transported to the northern part of France, over an average distance of 300 km, in accordance with manure legislation.

With regard to manure processing, we assume the use of a centrifuge for mechanical separation since it is the most common technique used in Belgium. According to Lemmens et al. (2007), the energy consumption of a centrifuge is 2 kW $h.m^{-3}$ slurry, while the energy use for composting on a large scale is assumed to be 50 kW $h.ton^{-1}$, including pre- and post-treatment, conversion and aeration (Lemmens et al., 2007; Melse et al., 2004). In a biological treatment plant where nitrogen is biologically removed from the liquid fraction by nitrification and subsequent denitrification, electrical energy is necessary for aeration, pumping and power. Aeration consumes the highest amount of energy. Registered uses for the two systems found in Belgium are 16 kW h m^{-3} manure (BioArmor system) or 17 kW h m^{-3} (Trevi system). In the BioArmor system, manure is biologically treated in a sequential batch reactor and sedimentation occurs in the SBR or in a regular sedimentation tank, while the Trevi biological treatment system is characterized by a separate nitrification and denitrification basin.

2.3.5. Avoided emissions

The manure slurry applied to crop land also substitutes for synthetic fertilizers. In order to calculate the CF, it is necessary to include (subtract) the impact relating to the production, transport and application of these replaced fertilizers. The production of fertilizers has a high energy demand, and consequently accounts for a large CF. According to Yara (2010), the production of 1 kg nitrogen with the Best Available Techniques emits 3.7 kg CO_2 eq. (kg N)$^{-1}$. The average cradle-to-gate CF for the production of 1 kg phosphorus fertilizer (triple super phosphate) and 1 kg potassium fertilizer (potassium sulphate) in Western Europe equals 0.46 kg CO_2 eq. (kg P_2O_5)$^{-1}$ and 0.29 kg CO_2 eq. (kg K_2O)$^{-1}$ respectively (Kool et al., 2012). For transport, an emission factor of 0.11 kg CO_2 eq. $ton^{-1}.km^{-1}$ is assumed. The mineral fertilizer equivalent (MFE) of nitrogen is based on the system for effective nitrogen and equals 60% for slurry (Sigurnjak et al., 2016; Vaneeckhaute et al., 2014; Vlaamse Landmaatschappij, 2014a). For P_2O_5 and K_2O, the MFE is assumed to be 90% (Coppens, 2008). The use of synthetic fertilizers also results in direct and indirect N_2O emissions through volatilization of NH_3 and NO_x, similar to the emissions related to the application of organic fertilizers.

2.4. Manure allocation model (MAM)

2.4.1. Model development

The second step of the analysis is to develop a model that optimizes manure allocation and builds on the spatial mathematical programming multi-agent simulation approach developed by Van der Straeten et al. (2010), using the optimization software GAMS (General Algebraic Modeling System).

The three strategies for manure management are, as mentioned above, transport of raw manure from nutrient excess to nutrient deficit areas, manure separation, and biological treatment of manure (manure treatment). The MAM minimizes, from a social planner perspective, either the costs or the GHG emissions from manure management in Belgium while respecting the fertilization standards defined by the Flemish Land Agency (Vlaamse Landmaatschappij, 2014a). While cost-efficiency is calculated based on transport distances and the cost of manure separation and treatment, GHG emissions, and hence, CF, are determined based on a consequential LCALCA approach.

As mentioned before, Fig. 1 shows a schematic representation of the MAM, as a part of our specific life cycle. Different types of livestock in municipality *i* produce manure with a different nutrient content, i.e. a specific quantity (in kg) of nitrogen and phosphorus each year (left on the figure). After storage, the manure is applied on the field, where different crops have different fertilization standards, meaning that, per crop type, a specific amount (in kg) of nitrogen and phosphorus can be applied to the land per hectare per year (to the right of the figure). This manure will first be applied to the fields in municipality *i*, and then to the fields in other municipalities *j* to *n*, to minimize transport distance and cost.

Because a manure surplus exists, not all raw manure can be applied on the field. Therefore, this excess manure has to be processed, altering the nutrient content of the different manure streams. This is the middle step in the figure, representing the different manure processing strategies mentioned previously. By adjusting the nutrient content of the different types of manure through processing, more (processed) manure can be applied on the field.

Of course, this manure allocation comes at an economic cost. First of all, there is the cost of transporting the (processed) manure from municipality *i* to municipality *n*. Secondly, there is the cost of spreading the manure on the field itself. Thirdly, processing the excess raw manure also comes at a cost. This cost depends on the type of processing and the manure type. In the cost calculation we also include (subtract) the avoided cost that would have been incurred if mineral fertilizer had been used instead of (processed) manure.

Apart from the economic cost, manure allocation causes GHG emissions. In the MAM, we account for five types of emissions: emissions from manure storage, emissions from manure processing and application, emissions from non-renewable energy use (i.e. transport and injection of manure slurry and manure processing) and avoided emissions from the production, transport and application of mineral fertilizers. These GHG emissions are implemented in the model as calculated values based on the assumptions stated in Section 2.3.

During the optimization, which is either a cost or a GHG minimization, the model allocates, for each municipality, the quantity of manure that should be processed, and transported (in raw or processed form) within the same, or to another, municipality, all while adhering to the fertilization standards.

As this model has been described in detail in previous publications (Van Der Straeten et al., 2011, 2010) we will not repeat the detailed explanation in this section.

2.4.2. Model scenarios

Two scenarios are analyzed; in the first scenario we minimize the economic cost of manure allocation in Belgium (S_COST), while in the second one we minimize the CF of manure allocation in Belgium (S_CF). It is important to note that a 'manure border' exists in Belgium, meaning that manure cannot be transported between Flanders and Wallonia. Hence, we simulate both scenarios for each region at the same time, but only allow transport of manure and derived products within each region.

The model was run for both scenarios and results were obtained relating to cost and CF. Moreover, as the main focus of the model is the CF, more detailed outputs were generated on the different CFs for (i) each type of manure management — raw, treated or separated manure, and (ii) each group of emissions — emissions from storage, transport, treatment, application and avoided emissions. The results of these analyses are presented in the Results section.

2.5. Consequential footprint approach

The effect of the distribution of livestock production can be analyzed both from the supply and the demand side for the surplus manure. The supply side implies an extra quantity of nutrients from animal sources, and it equals the demand-side effect of relaxing fertilization standards, i.e. allowing a marginal quantity of nutrients from additional manure to be put on the field. As, in general, marginal data are used to describe the consequences of a decision (Curran et al., 2005; Ekvall and Weidema, 2004; Sonnemann and Vigon, 2011), we marginally increase the nitrogen standard by 1 kg per municipality in order to understand the consequences of the decision to distribute livestock more evenly. The model then calculates the consequential CO_2 impact of this marginal increase, i.e. how much more or less CO_2 (in CO_2 eq.) is emitted per kg N per municipality. This is indicated in Fig. 1 by the dashed box at the bottom right of the figure.

3. Results

3.1. Overview of the model objective outcome

It must be emphasized that while cost minimization focuses on the optimal allocation of all manure produced in Belgium, CF minimization focuses solely on the optimal allocation of pig manure, as changes in manure policy almost always influence pig manure allocation, while the allocation of other types of manure remains unaffected.

The model simulations show that the total cost of manure management is 183 million and 648 million euro for S_COST and S_CF respectively, while the total emissions from manure management is 1.02 million and 0.67 million tons CO_2 eq. for S_COST and S_CF respectively. The numbers indicate that a lower cost coincides with a higher CF and vice versa. As transport of raw manure is the cheapest way to dispose of excess manure, the model, when minimizing costs, will choose to transport as much raw manure as possible, and excess amounts will be separated or biologically treated. Even though transport of raw manure is the cheapest way to deal with manure allocation, it is also the most polluting one. This largely explains why the CF in S_COST is much higher than in S_CF.

In the cost minimizing scenario S_COST, of the total amount of 20 million tons of pig manure produced annually, 33% is transported as raw manure, while 28% and 39% are treated and separated respectively. Moreover, emissions from raw manure transport, biological manure treatment and manure separation amount to 0.48 million, 0.25 million and 0.29 million tons CO_2 eq. respectively. On the contrary, in the emission-minimizing scenario S_CF, the only emissions originate from manure separation, amounting to 0.67 million tons CO_2 eq. This indicates that manure separation has the lowest CF of all three manure management strategies.

Figs. 2 and 3 allow us to look at these results in more detail.

Fig. 2 provides an overview of the GHG emissions in kg CO_2 eq. per ton of original manure for the three manure management scenarios, for the 'pigs of 20 to 50 kg in weight' category. First of all, it can be seen that raw manure management creates the highest total GHG emission per ton, followed by manure treatment and then manure separation, which has the lowest. Moreover, CH_4 emissions from storage are the highest for raw manure management, lower for separated manure and the lowest for treated manure, due to differences in storage time. This observation has also been made in a large number of environmental studies of pig slurry transfer and treatment (Loyon et al., 2007; Lopez-Ridaura et al., 2009; De Vries et al., 2012; Brockmann et al., 2014; ten Hoeve et al., 2014). Even though the raw slurry can be transported over a considerable distance, the processing strategies will always result in a lower CF since a shorter manure storage period is required. Notwithstanding the liquid fraction of separated manure emitting additional CH_4 during storage after separation, this emission is greatly reduced, by about twelve times. Mosquera et al. (2010) explain this reduction by the fact that CH_4 is formed in liquid slurry in the presence of a vast amount of degradable organic matter which is fermented under anaerobic conditions. However, since most of the organic matter ends up within the solid fraction after separation, less CH_4 is formed in the liquid fraction. Additionally, as manure can be biologically treated throughout the year, it is stored for a shorter time period as opposed to raw manure, where application is restricted in time (from mid-February until the end of August). The N_2O emissions increase with storage time. Moreover, emissions from transport are relatively low for raw manure, at 0.11 kg CO_2 eq. per ton and per km, and higher for processed

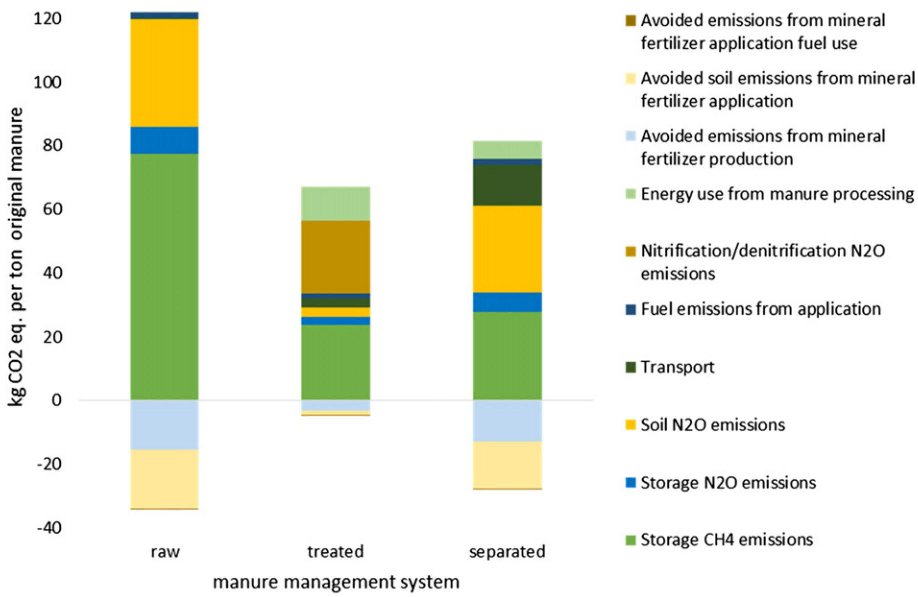

Fig. 2. Graphical overview of GHG emissions per type of manure management strategy for each type of emission source per ton original manure.

manure as this also includes (apart from transporting the liquid fraction over a distance of 1 km in the case of Fig. 2) the transport of the thick fraction to a composting facility and the transport of the resulting compost to France. The soil N_2O emissions from the separated liquid fraction are assumed to be slightly lower than those from the raw manure due to nitrogen losses after the separation process. Furthermore, as the liquid fraction of separated manure is a valuable source of nutrients which can be applied to crop land, it accounts for greater savings in mineral fertilizers compared to treated manure. By applying the liquid fraction of manure slurry instead of biologically treating it, the avoided consumption of synthetic fertilizers and GHG (N_2O) emissions related to the application of such fertilizers results in GHG emission savings.

Fig. 3 provides an overview of the relative size of the different emission types per scenario. Avoided emissions from fuel use are not represented as they are too small to show. The figure shows that when the total cost is minimized (S_COST), over 40% of the total GHG emissions are due to CH_4 emissions from storage. In the S_CF scenario, this emission source also dominates, albeit to a smaller extent. The second largest source of emissions in both scenarios is the soil N_2O emissions after manure application, as also indicated above. Emissions (N_2O) from biological treatment (nitrification/denitrification) are present only in the cost minimizing scenario, as manure treatment takes place here. Another important observation is that manure transport only contributes a small proportion of the total emissions. Finally, the figure indicates that the avoided emissions from the production of mineral fertilizers and the avoided (N_2O) emissions from mineral fertilizer application are similar in size.

3.2. Consequential carbon footprint analysis

As stated earlier, livestock production in Belgium and, thus, the GHG emissions relating to these sources are concentrated in the north and north-western part of the country. It has been suggested that a more equal distribution of livestock production might reduce the CF. By

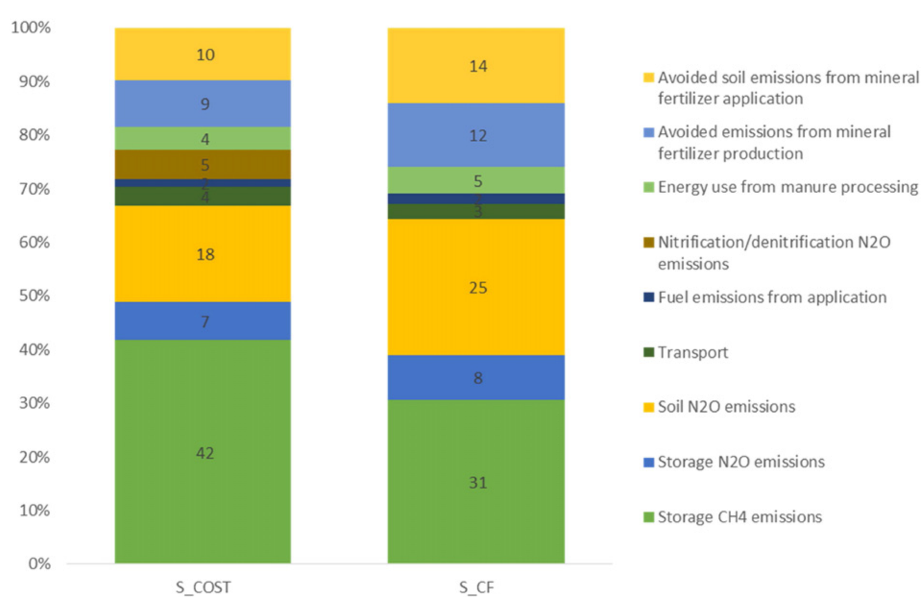

Fig. 3. Graphical overview of the two model scenarios depicting the share of each type of GHG emission source. Scenario S_COST minimizes the cost, while scenario S_CF minimizes the carbon footprint.

means of the consequential CF approach, the GHG emissions at municipality level can be simulated for a more spatially equalized livestock production.

The consequential CF impact of a marginal increase in N fertilization standards is carried out for the GHG emission minimizing scenario S_CF.

Results show that the consequential CO_2 impact is relatively small. The greatest reductions in CF are attained in areas with high livestock densities and carbon emissions, mainly in the North of Antwerp and West Flanders. One of the municipalities with a high CF is Hooglede, in the Western part of Flanders, which also has the highest livestock intensity. When we allow for an extra kg of N to be disposed of in this municipality, the change in nutrient deposition prevents the need for the manure to be transported to a region with non-binding fertilization standards, and, consequently, the CF decreases by 0.5 kg CO_2 eq. More specifically, 500 g CO_2 eq. corresponds to 250 kg of manure being transported over a distance of 20 km, since the emission factor for transport is 110 g CO_2 eq. ton^{-1} km^{-1}.

The consequential CF analysis shows that a high livestock concentration indeed creates a higher environmental burden than a more equally distributed animal production system. However, the difference between animals produced in areas with livestock overpopulation, on the one hand, and a region with lower livestock density, on the other hand, is relatively small. This difference is due to manure transport between regions in a competitive market for manure disposal space. The results show that the CF of this transport is small compared to the other emissions in the manure management and animal production chain.

The GHG emissions from the average storage of 1 ton of manure amount to 95 kg CO_2 eq. (Jacobsen et al., 2014) while the transport of that ton of manure from the surplus to the deficit regions is 0.5 kg CO_2 eq. (i.e. 0.52%), assuming an average transport distance of 50 km. Manure management, which is the consequence of livestock concentration, is responsible for one quarter of the total GHG emissions from pig production. The total CF of 1 kg of pig meat in Flanders is 5.7 kg CO_2 eq., of which 1.37 kg (i.e. 24%) is attributed, on average, to manure management (Jacobsen et al., 2014). Manure management in areas with the highest livestock density, hence, emits 0.13% more CO_2 than manure management in areas with low livestock density. Reducing this impact is more likely to be efficient through the implementation of improved manure management techniques, such as separation or anaerobic digestion, than through relocation of the livestock production itself. From the consequential analysis it can hence be concluded that a spatial rearrangement of pig production in Belgium will not substantially decrease the CF for this agricultural activity.

4. Discussion

The modeling exercise indicated that, out of the three selected manure management strategies, manure separation had the lowest CF and that spatial reallocation of pig production would not substantially decrease the CF for pig manure management. However, both the model and the assumptions we used can be improved.

First of all, the model ignores part of the reality. We assumed that (processed) manure transport takes place solely within the Belgian regions of Flanders and Wallonia. In reality, however, manure transport also takes place from Flanders to Zealandic Flanders in the Netherlands and Northern France. According to the 2015 Manure Report by the Flemish Land Agency (Vlaamse Landmaatschappij, 2015), in 2015, 125.1 million kg N and 60.9 million kg P_2O_5 were produced from manure in Flanders, of which 30% N and 37% P_2O_5 were processed and exported outside Flanders. Of all the nutrients exported from Flanders, 69 and 26% ended up on French and Dutch soil respectively. Allowing transport to those regions in the model would have further decreased the total cost of manure management in the S_COST scenario, while at the same time increasing the CF, as more manure would be transported to these nutrient-poor regions and less manure would be processed, as the need for processing would be lower. Nevertheless, part of this transport cost is already implicitly included in the model. Transport to Zealandic Flanders mainly consists of (unhygienised) raw manure, while transport to Northern France mainly constitutes the hygienised, composted thick fraction. In the MAM, we assume, firstly, that the entire thick fraction of processed manure is transported to a composting facility at an average distance of 50 km from the farm, and then transported to France as a hygienised compost, over an average distance of 300 km. These costs are already included in the model, but were not mentioned explicitly. As the statistics from the Flemish Land Agency indicate that the majority of cross-border manure transport is directed to France, we believe we have, to a large extent, already incorporated these additional costs. There would, however, be no difference in the S_CF scenario, as all manure would be separated by default to minimize the CF.

Secondly, in our choice of three manure management strategies we did not include anaerobic digestion as only about 1% of manure is currently digested in Flanders (De Geest et al., 2014; Vlaamse Landmaatschappij, 2014b). However, Prapaspongsa et al. (2010) found that combining anaerobic digestion with natural crust slurry storage produced the lowest impacts in terms of global warming potential, and McAuliffe et al. (2016) concluded that, when it comes to waste management for pig production, anaerobic digestion has many benefits over manure spreading. Moreover, Clemens et al. (2006) concluded that biogas production is a very efficient way to reduce the GHG emissions, both through the production of renewable energy and the avoidance of uncontrolled GHG emissions into the atmosphere during manure management. Finally, Anderson-Glenna and Morken (2013) observed that CH_4 emissions from digestate are much lower in comparison to raw manure slurry.

Anaerobic digestion could prove a good solution for manure management, as our results indicate that over one third of total emissions are caused by CH_4 emissions from storage (see Fig. 3). The difference in storage emission quantity between both scenarios lies in the fact that we assumed a much shorter storage time for manure processing than for raw manure transport. Changing these assumptions could hence alter the outcome of the exercise. For instance, due to the current pig meat crisis in Europe, Flemish manure processors have noticed that pig farmers store their manure for a longer period before processing to hold off payments for this service.

Besides anaerobic digestion, a number of solutions have been proposed in the literature to reduce the CF during manure storage. Methane production from slurry could be reduced by the addition of inhibiting compounds and acids (Amon et al., 2001; Berg and Pazsiczki, 2006). From the assessments carried out by Petersen et al. (2012) and Hou et al. (2015), it could be derived that slurry acidification significantly lowers CH_4 emission. Slurry acidification is already approved as Best Available Technology and widely applied in Denmark as a cost-effective GHG mitigation measure (Petersen et al., 2012). Slurry cooling can also reduce CH_4 emissions, since lowered indoor temperatures and a reduced air exchange rate reduce CH_4 emissions (Monteny et al., 2001). However, the most effective measure for CH_4 emission inhibition is to prevent formation of bacteria inoculum by frequent and complete slurry removal (Monteny et al., 2001; Osada et al., 1998; Van den Weghe et al., 2005).

Finally, according to the Flemish Environmental Agency, in 2014, animal husbandry was responsible for 63% of the total CH_4 emissions in Flanders and pig manure storage accounted for 64% of all CH_4 emissions from manure storage (Vlaamse Milieumaatschappij, 2014). Therefore, for further research, it might be interesting to look at the economic and environmental effects of imposing a CO_2 tax on manure management.

5. Conclusion

In European regions with concentrated livestock production, manure management creates major environmental problems. As the existing literature reports conflicting results for optimal solutions for pig waste management, this paper investigates the effect of reduced manure pressure through spatial distribution of CO_2 eq. emissions and the impact on the CF, verified through a consequential LCA. While, in the past, transport distance was assumed to be an important parameter in the determination of the total CF, our study shows that rearrangement of the spatial distribution of livestock production in Belgium will not substantially decrease CO_2 emissions. This article contributes to the methodology of consequential LCA by linking the CF analysis with an economic model that simulates manure disposal decisions driven by legal constraints and market forces.

This approach makes possible both an economic and environmental optimization through mathematical linear programming. The main differences between the environmental and economic optima were also determined. The results of the model simulations show that, while the economic optimum is achieved by maximizing the transport of raw manure until fertilization standards are fulfilled and subsequently processing the excess manure, the environmental optimum, from a CF viewpoint, is achieved by separating all manure, as this strategy creates the lowest CO_2 emissions, mainly due to the limited manure storage time. As manure storage is the main contributor to the CF, solutions for GHG reduction from manure management should lie in changing these storage systems, rather than in a spatial reallocation of intensive livestock production.

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References

Amon, B., Amon, T.T., Boxberger, J., Alt, C., 2001. Emissions of NH_3, N_2O and CH_4 from dairy cows housed in a farmyard manure tying stall (housing, manure storage, manure spreading). Nutr. Cycl. Agroecosyst. 60, 103–113. http://dx.doi.org/10.1023/A:1012649028772.

Anderson-Glenna, M., Morken, J., 2013. Greenhouse Gas Emissions From On-farm Digestate Storage Facilities.

Basset-Mens, C., van der Werf, H.M.G., Robin, P., Morvan, T., Hassouna, M., Paillat, J.M., Vertès, F., 2007. Methods and data for the environmental inventory of contrasting pig production systems. J. Clean. Prod. 15, 1395–1405. http://dx.doi.org/10.1016/j.jclepro.2006.03.009.

Bayo, J., Gómez-López, M.D., Faz, a., Caballero, a., 2012. Environmental assessment of pig slurry management after local characterization and normalization. J. Clean. Prod. 32, 227–235. http://dx.doi.org/10.1016/j.jclepro.2012.04.003.

Berg, W., Pazsiczki, I., 2006. Mitigation of methane emissions during manure storage. Int. Congr. Ser. 1293, 213–216. http://dx.doi.org/10.1016/j.ics.2006.02.050.

Brockmann, D., Hanhoun, M., Négri, O., Hélias, A., 2014. Environmental assessment of nutrient recycling from biological pig slurry treatment — impact of fertilizer substitution and field emissions. Bioresour. Technol. 163, 270–279. http://dx.doi.org/10.1016/j.biortech.2014.04.032.

Clemens, J., Trimborn, M., Weiland, P., Amon, B., 2006. Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. Agric. Ecosyst. Environ. 112, 171–177. http://dx.doi.org/10.1016/j.agee.2005.08.016.

Commissie Benchmarking, 2009. Commissie Benchmarking Vlaanderen - Jaarverslag. p. 2008.

Coppens, G., 2008. Dunne fractie inschakelen in de bemesting van derogatieteelten.

Coppens, G., 2009. De mestwegwijzer bundelt vijftien jaar mestanalyses. Landbouw Tech. 37–39.

Curran, M.A., Mann, M., Norris, G., 2005. The international workshop on electricity data for life cycle inventories. J. Clean. Prod. 13, 853–862. http://dx.doi.org/10.1016/j.jclepro.2002.03.001.

De Geest, V., De Mey, J., Vanacker, K., Wynsberghe, T., Meers, E., 2014. Voortgangsrapport 2014 — Anaerobe vergisting in Vlaanderen, stand van zaken werkjaar 2013–2014.

De Vries, J.W., Groenestein, C.M., De Boer, I.J.M., 2012. Environmental consequences of processing manure to produce mineral fertilizer and bio-energy. J. Environ. Manag. 102, 173–183. http://dx.doi.org/10.1016/j.jenvman.2012.02.032.

Defra, 2012. 2012 guidelines to Defra/DECC's GHG conversion factors for company reporting. Methodol. Pap. Emiss. Factors 1–85.

Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. Int. J. Life Cycle Assess. 9, 161–171.

Federal Public Service Economy S.M.E.s Self-employed and Energy, 2012n. Agricultural Survey of May 2012.

ten Hoeve, M., Hutchings, N.J., Peters, G.M., Svanström, M., Jensen, L.S., Bruun, S., 2014. Life cycle assessment of pig slurry treatment technologies for nutrient redistribution in Denmark. J. Environ. Manag. 132, 60–70. http://dx.doi.org/10.1016/j.jenvman.2013.10.023.

Hou, Y., Velthof, G.L., Oenema, O., 2015. Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: a meta-analysis and integrated assessment. Glob. Chang. Biol. 21, 1293–1312. http://dx.doi.org/10.1111/gcb.12767.

IPCC, 2006a. Chapter 10 Emissions from livestock and manure management. Forestry.

IPCC, 2006b. Chapter 11 N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. Agriculture.

Jacobsen, R., Vandermeulen, V., Van Huylenbroeck, G., Gellynck, X., 2014. Carbon footprint of pigmeat in Flanders. Int. J. Agric. Sustain. 12, 54–70. http://dx.doi.org/10.1080/14735903.2013.798896.

Kool, A., Marinussen, M., Blonk, H., 2012. LCI Data for the Calculation Tool Feedprint for Greenhouse Gas Emissions of Feed Production and Utilization Dry Milling Industry. p. 26.

Lemmens, B., Ceulemans, J., Elsander, H., Vanassche, S., Brauns, E., Vrancken, K., 2007. Beste Beschikbare Technieken (BBT) voor mestverwerking.

Lopez-Ridaura, S., Van Der Werf, H., Paillat, J.M., Le Bris, B., 2009. Environmental evaluation of transfer and treatment of excess pig slurry by life cycle assessment. J. Environ. Manag. 90, 1296–1304. http://dx.doi.org/10.1016/j.jenvman.2008.07.008.

Loyon, L., Guiziou, F., Beline, F., Peu, P., 2007. Gaseous emissions (NH_3, N_2O, CH_4 and CO_2) from the aerobic treatment of piggery slurry—comparison with a conventional storage system. Biosyst. Eng. 97, 472–480. http://dx.doi.org/10.1016/j.biosystemseng.2007.03.030.

McAuliffe, G.A., Chapman, D.V., Sage, C.L., 2016. A thematic review of life cycle assessment (LCA) applied to pig production. Environ. Impact Assess. Rev. 56, 12–22. http://dx.doi.org/10.1016/j.eiar.2015.08.008.

Meers, E., Michels, E., Tolpe, I., Tack, F., 2008. Verwerking van mest tot loosbaar/herbruikbaar water. Afvalwaterwetenschap 7, 116–123.

Meers, E., Rousseau, D.P.L., Blomme, N., Lesage, E., Du Laing, G., Tack, F.M.G., Verloo, M.G., 2005. Tertiary treatment of the liquid fraction of pig manure with Phragmites australis. Water. Air. Soil Pollut. 160, 15–26. http://dx.doi.org/10.1007/s11270-005-3428-2.

Melse, R.W., De Buisonjé, F.E., Verdoes, N., Willers, H.C., 2004. Quick scan van be- en verwerkingstechnieken voor dierlijke mest.

Monteny, G.J., Groenestein, C.M., Hilhorst, M.a., 2001. Interactions and coupling between emissions of methane and nitrous oxide from animal husbandry. Nutr. Cycl. Agroecosyst. 60, 123–132. http://dx.doi.org/10.1023/A:1012602911339.

Montes, F., Meinen, R., Dell, C., Rotz, a., Hristov, a.N., Oh, J., Waghorn, G., Gerber, P.J., Henderson, B., Makkar, H.P.S., Dijkstra, J., 2013. SPECIAL TOPICS — Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. J. Anim. Sci. 91, 5070–5094. http://dx.doi.org/10.2527/jas2013-6584.

Mosquera, J., Schils, R., Groenestein, K., Hoeksma, P., Velthof, G., Hummelink, E., 2010. Emissies van lachgas, methaan en ammoniak uit mest na scheiding.

Osada, T., Rom, H.B., Dahl, P., 1998. Continuous measurement of nitrous oxide and methane emission in pig units by infrared photoacoustic detection. Trans. ASAE 41, 1109–1114.

Petersen, S.O., Andersen, A.J., Eriksen, J., 2012. Effects of cattle slurry acidification on ammonia and methane evolution during storage. J. Environ. Qual. 41, 88–94. http://dx.doi.org/10.2134/jeq2011.0184.

Petersen, S.O., Blanchard, M., Chadwick, D., Del Prado, a., Edouard, N., Mosquera, J., Sommer, S.G., 2013. Manure management for greenhouse gas mitigation. Animal 7 (Suppl. 2), 266–282. http://dx.doi.org/10.1017/S1751731113000736.

Prapaspongsa, T., Christensen, P., Schmidt, J.H., Thrane, M., 2010. LCA of comprehensive pig manure management incorporating integrated technology systems. J. Clean. Prod. 18, 1413–1422. http://dx.doi.org/10.1016/j.jclepro.2010.05.015.

Rigolot, C., Espagnol, S., Robin, P., Hassouna, M., Béline, F., Paillat, J.M., Dourmad, J.-Y., 2010. Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part II: effect of animal housing, manure storage and treatment practices. Animal 4, 1413–1424. http://dx.doi.org/10.1017/S1751731110000509.

Sage, C.L., 2012. Environment and Food. Routledge, Oxon.

Sandars, D.L., Audsley, E., Cañete, C., Cumby, T.R., Scotford, I.M., Williams, a.G., 2003. Environmental benefits of livestock manure management practices and technology by life cycle assessment. Biosyst. Eng. 84, 267–281. http://dx.doi.org/10.1016/S1537-5110(02)00278-7.

Sigurnjak, I., Michels, E., Crappé, S., Buysens, S., Ack, F.M.G., Meers, E., 2016. Utilization of derivatives form nutrient recovery processes as alternatives for fossil-based mineral fertilizers in commercial greenhouse production of *Lactuca sativa* L. Sci. Hortic. (Amst.) 198, 267–276.

Skao, Connekt, Stimular, 2011. Lijst emissiefactoren.

Smet, E., Deboosere, S., 2007. Lachgasemissie bij mestverwerking - deel 2. Nieuwsbr. Milieutechnol. 14, 2–5.

Sonnemann, G., Vigon, B., 2011. Global Guidance Principles for Life Cycle Assessment Databases.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., De Haan, C., 2006. Livestock's Long Shadow — Environmental Issues and Options. Food and Agriculture Organization of the United Nations http://dx.doi.org/10.1007/s10666-008-9149-3.

Van den Weghe, H.F.A., Meissner, P., Kiuntke, M., 2005. Flushing litterless fattening houses with biologically treated slurry (SBR plant)—effects of flushing rate, percentage of treated slurry and seasons on gaseous emissions (methane, ammonia, nitrous oxide, carbon dioxide and odour). Proceedings of the Second European Conference on Precision Livestock Farming. Wageningen Academic Publishers, Uppsala, Sweden, pp. 73–80 (ISBN number 9789076998688).

Van der Straeten, B., Buysse, J., 2013. Vraag naar mestverwerking onder Map IV, Beleidsdomein Landbouw en Visserij. afdeling Monitoring en Studie, Brussel.

Van der Straeten, B., Buysse, J., Nolte, S., Lauwers, L., Claeys, D., Van Huylenbroeck, G., 2010. A multi-agent simulation model for spatial optimisation of manure allocation. J. Environ. Plan. Manag. 53, 1011–1030. http://dx.doi.org/10.1080/09640568.2010.495546.

Van Der Straeten, B., Buysse, J., Nolte, S., Lauwers, L., Claeys, D., Van Huylenbroeck, G., 2011. Markets of concentration permits: the case of manure policy. Ecol. Econ. 70, 2098–2104. http://dx.doi.org/10.1016/j.ecolecon.2011.06.007.

Vaneeckhaute, C., Ghekiere, G., Michels, E., Vanrolleghem, P.A., Tack, F.M.G., Meers, E., 2014. Assessing nutrient use efficiency and environmental pressure of macronutrients in biobased mineral fertilizers: a review of recent advances and best practices at field Scale. Adv. Agron. 128, 137–180.

Vlaamse Landmaatschappij, 2014a. Normen en richtwaarden 2014.

Vlaamse Landmaatschappij, 2014b. Voortgangsrapport Mestbank 2014 over de mestproblematiek in Vlaanderen.

Vlaamse Landmaatschappij, 2015. Mestrapport 2015 [WWW document]. URL. https://www.vlm.be/nl/SiteCollectionDocuments/Publicaties/mestbank/Demo Mestrapport 2015.pdf (accessed 2.17.16).

Vlaamse Landmaatschappij, 2016. Uitrijregeling [WWW document]. URL. https://www.vlm.be/nl/themas/Mestbank/bemesting/aanwenden-van-mest/uitrijregeling/uitrijregeling/Paginas/default.aspx (accessed 2.19.16).

Vlaamse Milieumaatschappij, 2014. Lozingen in de lucht 2000–2014.

Vlaamse Regering, 2014. Titel II van het VLAREM: Besluit van de Vlaamse Regering van 1 juni 1995 houdende algemene en sectorale bepalingen inzake milieuhygiëne [WWW Document].

VMM, VITO, AWAC, IBGE-BIM, Health, F.P.S. of, IRCEL-CELINE, Environment, F.C.S., ECONOTEC, 2014. Belgium's Greenhouse Gas Inventory (1990–2012) - National Inventory Report Submitted under the United Nations Framework Convention on Climate Change and the Kyoto Protocol.

VMM, VITO, AWAC, IBGE-BIM, Health, F.P.S. of, IRCEL-CELINE, Food Chain Safety and Environment, ECONOTEC, 2013. Belgium's Greenhouse Gas Inventory (1990–2012) — National Inventory Report Submitted Under the United Nations Framework Convention on Climate Change and the Kyoto Protocol.

Weiss, F., Leip, A., 2012. Greenhouse gas emissions from the EU livestock sector: a life cycle assessment carried out with the CAPRI model. Agric. Ecosyst. Environ. 149, 124–134. http://dx.doi.org/10.1016/j.agee.2011.12.015.

Werkgroep voor Rechtvaardige en Verantwoorde Landbouw, 2013. Economische briefing over de kosten en baten van de Vlaamse veeteelt.

Yara, 2010. CO_2-voetafdruk. Pure Nutr. Info.