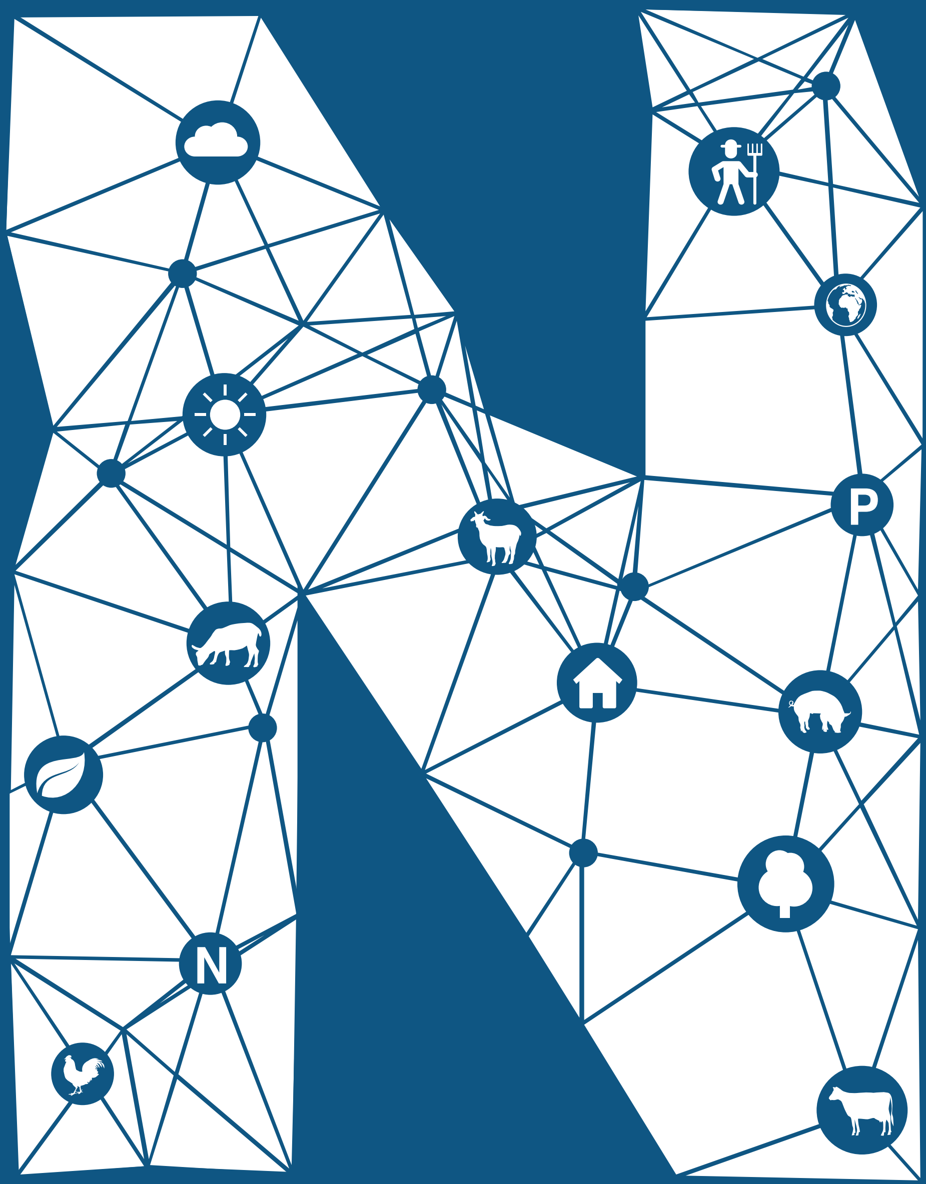
NUTRIENT CHALLENGES IN GLOBAL LIVESTOCK SUPPLY CHAINS.

An assessment of nitrogen use and flows.



Aimable UWIZEYE

Propositions

1. Nitrogen pollution deserves a global convention, with a prominent role of animal scientists and stakeholders.
   (this thesis)

2. Feeding swill to pigs is beneficial for the sustainability of food systems.
   (this thesis)

3. Gene editing using CRISPR-cas9 is vital to the future of humankind.

4. Although farmers acknowledge the need for change towards sustainable food systems, most of them do not translate this into business opportunities.

5. Training PhD candidates in leadership, creative skills and complex project management is essential for their professional success.

6. Open science is indispensable for the acceleration of the fourth industrial revolution.

Propositions belonging to the thesis, entitled

'Nutrient challenges in global livestock supply chains. An assessment of nitrogen use and flows.'

Aimable Uwizeye

Wageningen, 17 May 2019

Nutrient challenges in global livestock supply chains.

An assessment of nitrogen use and flows.

Aimable Uwizeye

Thesis committee

Promotors
Prof. Dr Imke J.M. de Boer
Professor of Animal Production Systems
Wageningen University & Research

Prof. Dr Pierre J. Gerber
Extraordinary Professor, Animal Production Systems Group
Wageningen University & Research
Food and Agriculture Organization of United Nations
World Bank

Co-promotor
Prof. Dr Rogier P.O. Schulte
Professor of Farming Systems Ecology
Wageningen University & Research

Other members
Prof. Dr Carolien Kroeze, Wageningen University & Research
Prof. Dr Jan Willem Erisman, VU University, Amsterdam, The Netherlands
Dr Adrian Leip, European Commission, Joint Research Centre, Italy
Dr David Wall, Agriculture and Food Development Authority (Teagasc), Wexford, Ireland

This research was conducted under the auspices of the Graduate School of
Wageningen Institute of Animal Sciences (WIAS)

Nutrient challenges in global livestock supply chains.

An assessment of nitrogen use and flows.

Aimable Uwizeye

Thesis
submitted in fulfilment of the requirements for the degree of doctor at
Wageningen University
by the authority of the Rector Magnificus,
Prof. Dr A.P.J. Mol,
in the presence of the
Thesis Committee appointed by the Academic Board
to be defended in public
on Friday 17 May 2019
at 1:30 p.m. in the Aula.

Aimable Uwizeye
Nutrient challenges in global livestock supply chains.
An assessment of nitrogen use and flows.
182 pages.

PhD thesis, Wageningen University, Wageningen, the Netherlands (2019)
With references, with summaries in English and French.

DOI : https://doi.org/10.18174/469578.
ISBN: 978-94-6343-582-6

Abstract

The rapid growth of the livestock sector has altered the way the sector influences global nutrient flows and emissions, with repercussions on environmental and public health issues. Designing interventions for better environmental sustainability will require a framework and indicators adapted to the increasingly long, and complex livestock supply chains. To develop such a framework, we reviewed existing studies and found that four methods were used previously to analyse nutrient use in the livestock sector, namely a nutrient balance, nutrient use efficiency (NUE), material flow analysis and life cycle assessment. Among these methods, NUE appeared as a suitable approach to benchmark nutrient management that can be integrated in life cycle approach to compute the supply chain level NUE, which is proposed as a valuable indicator of nutrient management sustainability. To this end, we developed a comprehensive framework of indicators to assess the sustainability of nutrient use. The framework encompassed three indicators, including the life-cycle nutrient use efficiency (life-cycle-NUE), life-cycle net nutrient balance (life-cycle-NNB) and nutrient hotspot index (NHI). It was tested and the indicators proposed were found to be suitable to describe nitrogen (N) and phosphorus dynamics and were all needed. This framework requires detailed data, which are highly variable at global level, resulting in large uncertainties of the results. Focusing on N, we proposed a method, which relies on a global sensitivity, to identify the important inputs parameters that contribute significantly to the variance of the results, using the Global Environmental Assessment Model (GLEAM) dataset. The results showed that uncertainties of a few important input parameters, such as manure deposited and applied could explain most of the variance of N use indicators. Fixing non-important parameters and substituting important parameters in GLEAM for new field survey data, improved the results of N use indicators. Subsequently, we applied the framework to assess N use, flows and emissions, in the global pork supply chains and to evaluate the effects of feeding swill to pigs as a strategy to integrate better livestock in a circular bio-economy. The results showed that N emissions into the environment amount to around 14.7 Tg N y^{-1}, of which 68% is lost to watercourses. These results showed that the efficiency of N use and the magnitude of N losses per unit of area depend chiefly on the region, origin of feed, and manure management. The substitution of swill for grains and soybeans resulted in the improvement of N use indicators and abate N emissions. Implementing swill feeding would require innovative policies to guide the collection, treatment, and usage of swill, and ensure safety and traceability. Applying the framework to global livestock supply chains showed that they are responsible for around one-third of human-induced N emissions (65 Tg N y^{-1}) of which 63% take place in 2 regions (i.e. South Asia and East and Southeast Asia), and 61% at the feed production stage. We found a wide range of values for N use indicators, which indicates that good practices are available and already implemented in parts of the value chains. These findings imply that there is both urgent need to reduce these emissions and the opportunity to design targeted mitigation interventions. The design and implementation of interventions should consider potential trade-offs and synergies with other sustainability dimensions, such as climate change, resource scarcity, trade balance, risk management, public health and food security. Our study suggests that N challenges are global and cannot be tackled without considering the contribution of global livestock supply chains, thus requiring a global convention with a strong representation of stakeholders in the livestock sector.

Contents

Chapter 1

General introdution

1.1 Background

The global livestock sector has contributed, since the origin of agriculture, to livelihoods of millions of rural people and has recently been growing in response to the increasing demand for animal-source food (ASF) (FAO, 2011; Herrero and Thornton, 2013). Since 1961, ruminant meat production has increased two-fold, pork, chicken and egg production have increased four-fold, whereas milk production has increased by an impressive twelve-fold (see Figure 1.1, FAOSTAT (2018)). There are, however, large heterogeneities across regions. In Asia, for instance, cattle meat production has increased eight-fold, whereas growth in Africa has been relatively slow at three-fold. For milk production, the European Union remains the largest producer globally, but its production has been refrained by milk quota between 1984 and 2015 (European Commission, 2015), whereas, milk production has been multiplied by nine in Asia and by four in Africa (FAOSTAT, 2018).

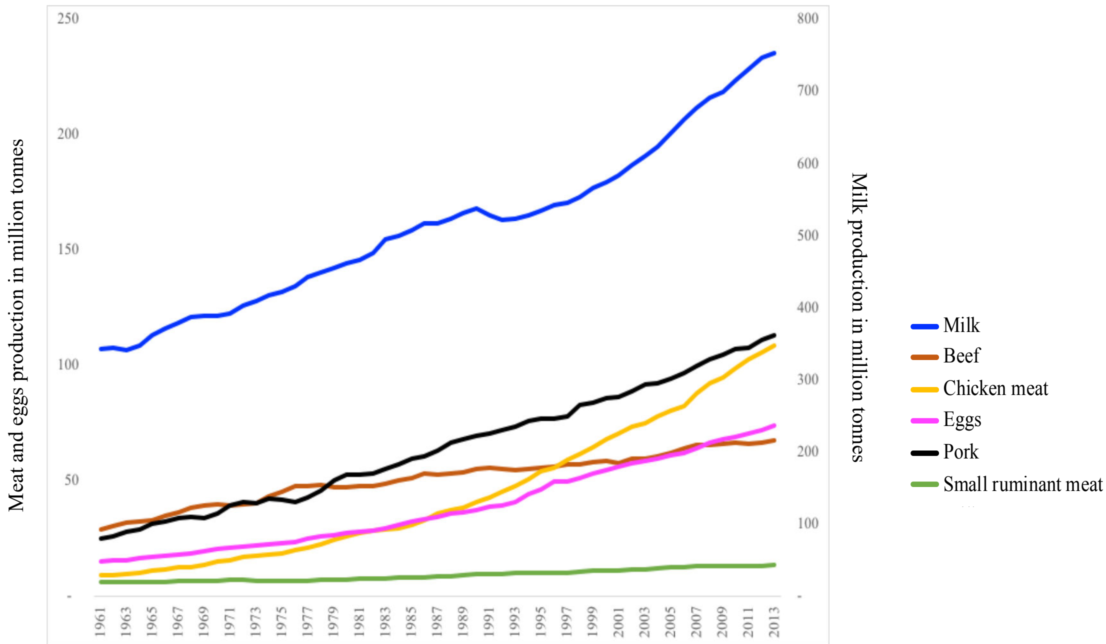


Figure 1.1: Global supply of animal-sourced food from 1961 to 2013 (based on FAOSTAT (2018))

While the speedy growth of the livestock sector has satisfied the growing demand for ASF, the sector has developed a gargantuan appetite for natural resources. Livestock use about 15% of terrestrial land for grazing, while about 40% of the global arable land is used to cultivate livestock feed (Mottet et al., 2017). Livestock consume about 100 Tg nitrogen (N) and 15 Tg phosphorus (P) in the form of grass and feed-crops, of which more than 80% is excreted in manure (Sutton et al., 2013a), leading to N and P emissions into the environment. These losses contribute significantly to environmental issues, such as climate change, eutrophication, acidification, and consequential biodiversity loss (see Box 1.1 and Figure 1.2). The livestock sector contributes about 14.5% of the global anthropogenic emissions of greenhouse gas (GHG)

(Gerber et al., 2013). The contribution of the livestock to global N and P losses, which, overall, have transgressed the planetary boundaries (de Vries et al., 2013; Steffen et al., 2015), hasn't yet been quantified. A recent study found that food production, including livestock, contributed 32% to global terrestrial acidification and 78% to global eutrophication (Poore and Nemecek, 2018). Regional studies have quantified the contribution of livestock to N and P losses in European Union and estimated that the livestock sector is responsible for around 80% of total N emissions to air (NH_3 and NO_x) and 73% of total N and P losses to ground and surface water from the agricultural sector (Leip et al., 2015). Feed production contributes mostly and accounts for 49% of NH_3 emissions and all N and P losses to water bodies.

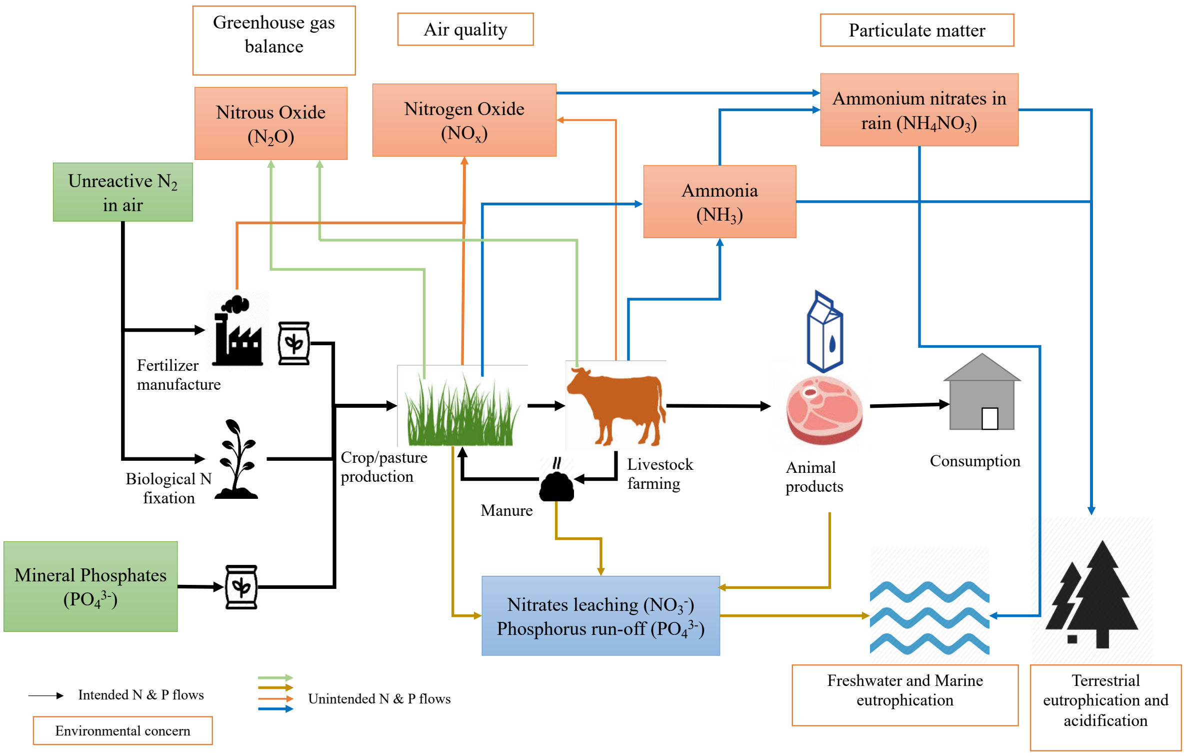


Figure 1.2: A simplified overview of nitrogen (N) and phosphorus (P) flows in livestock supply chains, highlighting anthropogenic sources, the cascade of reactive N forms and associated environmental impacts, adapted from Sutton et al. (2013a).

Given the increasing awareness of society regarding livestock's contribution to environmental issues, a main challenge for stakeholders in the sector is to produce ASF with lower environmental costs. With the rapid growth and transformation of the sector, there is a need to understand where N and P losses in the livestock supply chains occur and which frameworks are most suited to assess them. To address this, I first highlight the changes that have occurred in livestock systems and how these changes have impacted nutrient flows and losses.

Box 1.1 Description of the environmental impacts associated with N and P use and losses

- **Air quality and human health**
Emissions of ammonia (NH_3) and nitrogen oxide (NO_x) from livestock systems contribute to the formation of particulate matter (PM). Moldanova et al. (2011) suggest that PM is the most significant cause of health issues related to air pollution. Globally, air pollution is estimated to cause 9% of lung cancer deaths, 5% of cardiopulmonary fatalities and about 1% of respiratory infection deaths. Outdoor air pollution contributes to 5% of all cardiopulmonary deaths worldwide. NO_x emissions also contribute to photochemical smog, leading to high levels of tropospheric ozone (O_3). NO_x is a mixture of nitric oxide (NO) and nitrogen dioxide (NO_2). The latter is a toxic gas that has adverse health effects due to oxidative stress such as the formation of radicals that are destructive to cell tissues. In areas with high concentrations of animals, emissions of NH_3 and NO_x into the lower atmosphere are abundant and can have implications on human health (Erisman et al., 2007; 2011).
- **Climate change**
Emissions of nitrous oxide (N_2O) from livestock systems contribute to climate change at global level (Galloway et al., 2003). N_2O has one of the longest atmosphere lifetimes lasting for up to 150 years. It has a global warming potential 265 times greater than carbon dioxide (CO_2) without considering carbon cycle feedbacks (Myhre et al., 2013). Climate change is the environmental issue that currently receives most global attention, reinforced since the Paris Climate Agreement (UN, 2018), aiming to limit the global temperature increase below 2 °C and attempting to limit this to 1.5 °C above pre-industrial levels. This agreement requires each country to commit to GHG reduction targets through Nationally Determined Contributions (NDCs) to the United Nations Framework Convention on Climate Change (UNFCCC). The livestock sector is considered in mitigation strategies by 55 countries, through the improvement of manure management, reduction of enteric fermentation, improvement forage quality in grazing and silvopastoral systems or use of biogas digesters (Wilkes et al., 2017).
- **Aquatic and terrestrial eutrophication**
Eutrophication of freshwater ecosystems occurs through the direct inflow of nitrates, organic and particulate N and P via runoff and leaching, and atmospheric deposition (Payen and Ledgard, 2017). Nutrients losses in freshwater or atmospheric deposition can reach the marine ecosystems, causing marine eutrophication (Strokal et al., 2014). Eutrophication is characterised by toxic algal blooms or planktonic growth leading to excessive oxygen depletion in benthic water (Cosme et al., 2017). This hypoxic environment can cause several damages to exposed species, such as fish kills and other loss of biodiversity (CML, 2003). It also can have a negative impact on water usages for drinking or recreation (Schulte et al., 2006). Leaching of nutrients from synthetic fertilizer and manure into rivers, for instance, are the primary sources of eutrophication in the Yellow and South Chinese seas (Strokal et al., 2014). Eutrophication of terrestrial ecosystems occurs via the atmospheric deposition of NH_3

causing biodiversity loss through the reduction of plant species richness (Stevens et al., 2004).

- **Aquatic and terrestrial acidification**
  Livestock systems release large quantities of NH_3 that are transported in the atmosphere and are deposited to other ecosystems (Costanza et al., 2008). Besides the aquatic uptake of CO_2, N deposition plays a role in coastal waters because half of NH_3 emissions from agriculture is deposited in oceans. The aquatic acidification reduces ocean alkalinity and threatens the aquatic ecosystems such as coral reefs and coastal benthic and planktonic foodwebs (Doney et al., 2007). The terrestrial acidification occurs when ammonium ions increase levels of the acidity of the rainwater causing damages to the soils or natural ecosystems such as a forest. The terrestrial acidification occurs when ${NH_4}^+$ is formed from NH_3 in the air, and is subsequently deposited on the soils through rainwater. Bacteria subsequently nitrify ${NH_4}^+$ into nitrate, and during this process, H^+ is formed. This H^+ can be buffered by the soil, depending on the availability of base cations (${Ca_2}^+$, ${Mg_2}^+$, K^+ and Na^+). Once these cations are exhausted, ${Al_3}^+$ and ${Fe_3}^+$ are mobilised from soils. High Al and Fe concentrations are toxic to soil biota and are characteristic of acidified soil (Lu et al., 2014).

1.2 Structural changes in the livestock sector and their impacts on nutrient cycles

Traditionally, livestock systems are local, small-scale and based on family farms, where they produce ASF from locally available feed resources, and for local markets (FAO, 2011). N and P management has been at the centre of these systems, connecting feed and animal production stages in one location. Nutrients recycling and balances are the critical elements to N and P management and soil fertility. At present, nutrient cycles are often disrupted, because of decoupling of crop and animal production, massive use of artificial fertilizer and leakage of valuable nutrients. For example, massive nutrient losses occur in countries where subsidies on synthetic fertilizer have resulted in high application rates per unit of land, such as in China and India (Sutton et al., 2013a). In other regions, such as in Sub-Saharan Africa, the fertilisation of feed crops and pastures has found to be lower than plant nutrient requirements, which leads to nutrient depletion and soil degradation (Lassaletta et al., 2014; Reis et al., 2016).

Significant structural changes, driven by economic and institutional trends are transforming the nature of livestock systems in many parts of the world, away from small-scale mixed crop-livestock systems. Globalisation and liberalisation of trade have boosted international transfers of capital, labour, commodities and technologies. The resulting internationalisation of markets, combined with a growing demand for ASF, has led to a shift from a predominantly supply driven activity providing local needs, to global and demand-driven activity (De Haan et al., 2010; Freeman et al., 2006). Economies of scale in production and processing combined with standards and regulations related to market access have resulted in the emergence of larger commercial farms. Producers have taken advantages of better and cheaper transport facilities and cold chains, in some countries, to relocate production activity where costs are low and rely on transport to connect to upstream and downstream markets (De Haan et al., 2010; Gerber et al., 2010; MacDonald and McBride, 2009). These developments have resulted in more extended, and frequently transcontinental supply chains in which feed and animal production stages are disconnected with implications on global nutrient flows and losses. These implications are related to the change from circular systems, where most of the nutrients in residues are reused, to linear systems, where excessive nutrients from feed and animal production stages are lost into the environment.

These developments have also resulted in a geographical concentration of livestock farms leading to the concentration of large quantities of manure that exceed the absorptive capacities of the available agricultural land and environment (MacDonald and McBride, 2009; Strokal et al., 2014). The transport of manure to agricultural land in many regions is limited by the increasing cost of transport and spreading (Fealy and Schröder, 2008; Hendriks et al., 2016). This situation results in the excessive application rate of manure (Costanza et al., 2008) or manure dumping into surface water (Bai et al., 2016; Gerber and Menzi, 2006; Schaffner et al., 2009; Strokal et al., 2014), contributing to water pollution and eutrophication. For instance, Bai et al. (2014) reported that 45% to 70% of manure excreted by pig production in China is discharged into the environment. Moreover, the concentration of animals in limited areas exacerbates air and water pollution, harming human health and biodiversity (Sutton et al., 2013a).

Consequently, feed production has expanded rapidly, mainly in Latin America and North America, taking advantage of land availability, low-cost synthetic fertilizer, mechanisation, and energy (De Haan et al., 2010; Gibbs et al., 2015; de Oliveira and Schneider, 2016). The intensive use of synthetic fertilizer and associated nutrient losses has exacerbated large-scale environmental impacts, such as eutrophication of rivers and oceans (Hamilton et al., 2018). These trends are well illustrated by Galloway et al. (2007), who estimated that the Japanese pork consumption is associated with 220 Gg N losses, taking place in feed exporting countries and 70 Gg N, which are released directly in Japan's environment due to the domestic pig production. Nutrient losses in exporting countries are often ignored in nutrient balance approaches of consuming countries, which focus on the local animal production units (Godinot et al., 2014; Mu et al., 2016). Ignoring these losses can misguide the design of improvement pathways by policy and decision-makers. Given that most of livestock supply chains are long and globalised, it is essential to assess nutrient flows and losses at a chain level by considering the current trends of the sector and going beyond the mere assessment at one production stage.

1.3 Knowledge gaps

1.3.1 Methodological challenges

To reduce nutrient losses in livestock systems, there is a need for methods and indicators that determine these losses or the other way around, determine the nutrient use efficiency (NUE). A method used is a nutrient balance approach (NB), which assesses the difference between nutrients entering and leaving a farm, and yields NUE indicators, such as the amount of nutrients in outputs over nutrients in inputs. An NB is a highly relevant method to understand nutrient losses at local, small-scale farming systems, in which plant and animal production are still connected (Gourley et al., 2012b; Mu et al., 2016; Powell et al., 2010). An NB has been used effectively to identify best practices in nutrient management at farm-scale and to support livestock farmers to reduce nutrient losses into the environment (Oenema, 2006; Schröder et al., 2003). Besides the farm level, NB approaches are also relevant at the watershed or regional level, where they provide insights about nutrient loads into the environment and support the formulation of nutrient management policy (Schulte et al., 2010; Gerber et al., 2005; Hutchings et al., 2014).

For current livestock supply chains, which run across national and continental boundaries, however, NB approaches over-look nutrient losses associated with off-farm activities, such as the production of feed (Godinot et al., 2014; Mu et al., 2016). To confront with this issue, some studies have attempted to improve NB by not only considering the local inflows and outflows, but also import and export nutrients flows, e.g. Erisman et al. (2018), or by including upstream nutrient losses, e.g. Godinot et al. (2014). These approaches, however, focused on country and system level and did not consider the entire supply chain to identify hotspots of nutrient losses.

Other researchers have used life cycle assessments (LCA), a process-based approach at chain level, to report environmental impacts (possibly taking place in various locations) per unit of ASF (De Vries and De Boer, 2010; Leip et al., 2013). LCA is a holistic and standardised approach that focusses on potential environmental impacts, such as eutrophication or acidification, related to production, usage and end-of-life (ISO 14040, 2006). The LCA framework, however, involves several modelling choices, leading to differences in results (Curran, 2014). When it comes to impact assessment methods, the differences in results are more significant due to a lack of scientific consensus. It is also not confident that an LCA is needed to inform policy and decision-making since any continuous loss of nutrient is likely to cause an environment and resource management issue. In this case, nutrient use indicators computed at a supply chain level can provide insights into the location of hotspots of nutrient losses as an entry-point for nutrient management interventions. To assess the nutrient losses and NUE, few studies have developed a chain level approach, from farm to regional or country (Suh and Yee, 2011; Wu et al., 2014a). These studies have given insights into NUE for food systems, but their scopes have been limited to the regional level (Suh and Yee, 2011; Wu et al., 2014a) and they did not consider the effect of nutrient recycling and stock changes on NUE (Sutton et al., 2013a). These studies also did not identify hotspots of nutrient loss, which is required to support targeted nutrient improvement pathways towards sustainable nutrient use.

Considering the geographical nature of global livestock supply chains, a framework to assess nutrient flows and losses at chain level is lacking. Such a framework would help to quantify best practices at various stages of the supply chain. This thesis focusses on the development and application of such an assessment framework. The results will support policy and decision-making with evidence to reduce environmental impacts across global livestock supply chains.

1.3.2 Data gaps

To quantify nutrient flows and losses along global livestock supply chains, not only an indicator framework is needed, but also high-quality data. Global datasets on livestock production, however, are scarce, often outdated or do not reflect the spatial and temporal variability of the production systems (Henderson et al., 2017). Given the limited availability of good-quality data, regional averages, default parameters and expert knowledge are often used to quantify nutrient flows at the global level (Billen et al., 2014; Sutton et al., 2013a), despite high uncertainty and potential biases associated with them (Oenema et al., 2015). The results of the analysis performed with such datasets are informative at a global scale, but they are hardly used at a country level to inform evidence-based policy and decision-making, limiting their relevancy where most policy decisions are made. To make these results relevant at a local level, all input parameters can be addressed with high-quality data, but this process is onerous and expensive. An alternative approach is to understand which input parameters contribute mostly to the variance of the results and address them with high-quality data while assessing nutrient flows and losses. A comprehensive method to identify and improve the quality and completeness of environmental indicators computed from global datasets, nevertheless, is lacking.

1.3.3 Livestock's contribution to the disruption of global N cycles

The indicator mentioned above framework and data can be used to assess the contribution of the livestock sector to the disruption of global N cycles, which is lacking. At the regional level, Leip et al. (2015) have provided insights into the contribution of the livestock sector to different forms of N losses in the European Union. These insights are essential to shape a sustainable development of the livestock sector and inform policy-makers with evidence for better mitigation strategies. Such a contribution can provide a detailed analysis of the magnitude, sources and pathways of N to stimulate international efforts to design improvement pathways and inform policy dialogue. It can help to understand how the livestock contribute to the transgression of the planetary boundary for biochemical flows of N (Steffen et al., 2015) and support evidence-based decisions to shape the future of livestock sector. It can also feed into existing international initiatives aiming to improve N management such as International Nitrogen Management Systems (http://www.inms.international/).

1.3.4 Potential of new circular livestock supply chains

To understand the effectiveness of the improvement pathways proposed above, it is essential to test novel ways of re-using locally available resources at a global scale, especially for globalised livestock supply chains, where feed and animal production stages are increasingly disconnected. Among those, feeding food waste and losses to livestock has been shown to result in positive effects such as the reduction on reliance on human edible feed, reduction of land-use (zu Ermgassen et al., 2016; van Zanten et al., 2018), reduction of GHG emissions (Papargyropoulou et al., 2014), and reduction of waste streams (Makkar, 2017; Papargyropoulou et al., 2014). Feeding food wastes is a common practice in smallholder' systems, but is seldom done in industrial pigs, except in South Korea and Japan (zu Ermgassen et al., 2016). This intervention is part of the broader circular bio-economy context, defines as: "*economic system that replaces the 'end-of-life' concept with reducing, reusing, recycling and recovering materials in production/distribution and consumption processes*" (Kirchherr et al., 2017). An overview of the effects of large-scale feeding of food wastes to global pigs would have on NUE_N and N losses, is lacking. Such an analysis would foster discussion amongst livestock stakeholders and farmers at international level on the acceptability of the use of food wastes and losses as animal feed.

1.4 The aim of this thesis

The objectives of the thesis, therefore, are to:

- develop a framework of indicators to assess nutrient flows and emissions along global livestock supply chains, while identifying data, which can be improved to enhance the accuracy of the final results;
- assess the impacts of the global livestock supply chains on the nitrogen cycle, while exploring the improvement options.

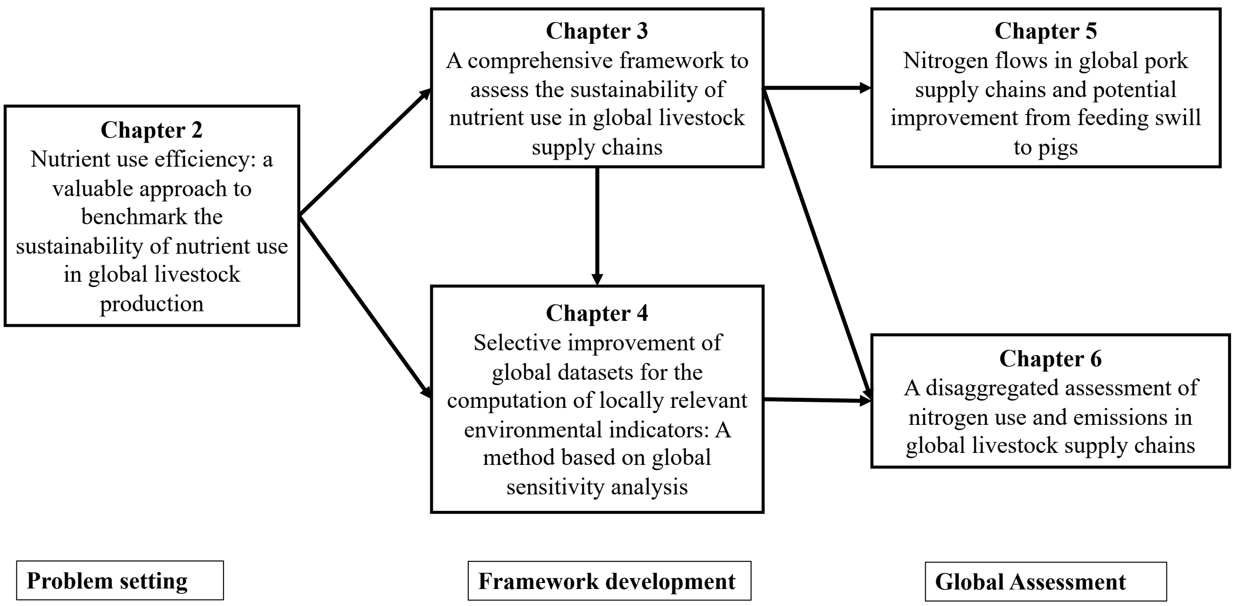


Figure 1.3: Overview of the chapters and structure of the thesis.

1.5 Outline of the thesis

The thesis is divided into three parts: definition of nutrient use efficiency at supply chain level; development of methodological frameworks and; global assessment of nitrogen use in livestock supply chains. The overview of the chapters and structure of this thesis are shown in Figure 1.3.

Chapter 2 defines NUE from a life cycle perspective as a valuable indicator of sustainable nitrogen and phosphorus management. It also provides a comparative overview of existing frameworks for the assessment of nitrogen and phosphorus use in livestock supply chains.

Chapter 3 proposes a framework to support the monitoring of options to improve the sustainability of nitrogen and phosphorus use in global livestock supply chains. The framework integrates three nutrient use indicators that describe complementarily aspects of nitrogen and phosphorus management. The framework is tested for the case of mixed dairy supply chains in Europe.

Chapter 4 provides an innovative method to improve global datasets for the computation of local-level environmental indicators. The method relies on the global sensitivity analysis of the model output. Input parameters that can be derived from global datasets are distinguished from those that require local data collection. The method is tested for two case studies: mixed dairy supply chains in The Netherlands and Rwanda.

Chapter 5 assesses N flows and N use indicators for global pork supply chains: backyard, intermediate and industrial. Given the importance of industrial system in global pork supply and its potential to absorb a part of food losses and wastes (swill), the chapter explores potential improvement from swill feeding as an option to increase NUE and reduce nitrogen losses. The chapter provides a comprehensive assessment of the magnitude of N flows and N use indicators. The results are aggregated at a global, regional and system level.

Chapter 6 assesses the impacts of global livestock supply chains on nitrogen flows. The chapter provides a detailed analysis of the magnitude, sources and pathways of N by species and production systems and proposes macro-level mitigation options to stimulate policy dialogue for the improvement of the global sustainability of N management.

Chapter 2

Nutrient use efficiency: a valuable approach to benchmark the sustainability of nutrient use in global livestock production?

Pierre J. Gerber[1,2], Aimable Uwizeye[1,2], Rogier P.O. Schulte[3], Carolyn I. Opio[1] and Imke J.M. de Boer[2]

[1]Animal Production and Health Division, Food and Agriculture Organization of the United Nations, Viale delle Terme di Caracalla, 00153 Rome, Italy
[2]Animal Production Systems group, Wageningen University, PO Box 338, Wageningen, The Netherlands
[3]Teagasc Johnstown Castle, Wexford, Ireland

Current Opinion in Environmental Sustainability, 9, 122-130

Abstract

Livestock have a large impact on nutrient cycles, with repercussions on environmental and public health issues. Designing interventions for better environmental sustainability will require indicators adapted to the increasingly long and complex supply chains. Nutrient use efficiency is a well know approach to benchmark nutrient management at the animal level, and to some extent at the farm level. Integrating the life cycle approach into NUE allows for the computation of supply chain level NUE, which is proposed as a valuable indicator of nutrient management sustainability. It characterizes the use of finite nutrient and energy resources and the losses of nutrients per unit of product, likely to have impacts on the environment and public health. Further research is required to harmonize life-cycle-NUE and test its validity as an indicator of nutrient management sustainability.

2.1 Introduction

2.1.1 The internationalization of livestock supply chains

The management of nutrient flows is central to agriculture and food supply chains. It has driven the development of agricultural practices since their origins: flood control, crop rotation, manure recycling and crop residues management are examples of techniques aimed at, inter alia, harnessing nutrients and maintaining them in agricultural systems. Animals have always played an important role in agricultural nutrient cycles, whether we think about their capacity to "harvest" nutrients in natural rangelands, or about intensive systems were they are fed nutrient dense rations specifically cropped for this purpose.

The manipulation of nutrient cycles and amplification of nutrient flows have dramatically increased since the industrialization of agriculture, with a positive effect on the volume and stability of food production, but also with associated risks of detrimental effects on ecosystems and public health (Steinfeld et al., 2006). Several studies have highlighted these environmental impacts and developed analytical frameworks for their assessment (see for example Bouwman et al. (2013); Godinot et al. (2014); Halberg et al. (2005a); Schröder et al. (2004)).

At the same time, livestock supply chains are increasingly internationalized, spanning across borders and continents. Fertilizers and feed have been internationally traded in large quantities over the past decades, but livestock products, too, are increasingly exchanged on global markets: today, it is estimated that 16% of poultry meat, 12% of pig meat and 15% of beef are traded on international markets, compared to a 35% of soybean cakes, 28% of nitrogen (N) fertilizers and 31% of phosphorus (P) fertilizers (FAO statistics (FAOSTAT, 2018)). These figures indicate that livestock supply chains are increasingly global, diverting from a pre-industrial situation where domestic animals essentially converted locally available feed resources (nutrient) into products for local consumption.

The management of nutrient use in livestock supply chains, therefore, needs to take these recent trends into consideration. New indicators and approaches are required that are applicable to longer and more complex supply chains and can assist in improving the performance of the entire livestock systems, going beyond the mere assessment of performance at animal or production unit level.

2.1.2 Management of natural resources along the supply chain

In this context, the management of natural resources, and nutrient flows in particular is increasingly focusing on the concept of efficiencies along the supply chain. Used in different fields, efficiency is a measurement of performance, relating the result of a process to the mix of inputs mobilized for its delivery. The concept has gained growing importance in the sustainability debate, playing a pivotal role in addressing the basic conflict between continuous growth in the consumption of material goods and the finite natural resources of the planet. It is of particularly high relevance to the agricultural sector, which needs to deliver on the dual objective of food security (output growth) and environmental sustainability (reduced natural resource mobilization) (Gerber et al., 2013; Rai et al., 2011; Sutton et al., 2013a).

Natural resource use efficiency, defined as the amount of natural resources engaged per unit of product is of particular relevance to livestock supply chains, which are typically longer than crop supply chains; involve more biophysical processes and often include the recycling and reuse of by-products from other sectors. The livestock sector is a major user of natural resources, such as land and water, using about 35% of total land and representing about 8% of total water withdrawals, mostly for feed production (Steinfeld et al., 2006).

Improving natural resource use efficiency is an objective of global relevance. It applies to the most affluent areas of the globe, where the sector is requested to minimize its environmental impact (Schröder et al., 2011), and to emerging economies, where livestock production expands rapidly in a context of relatively weak environmental policies and often wastes natural resources (FAO, 2009). It is equally relevant to the poorest regions of the world, but from an opposite perspective: here, there is a need for maximizing production out of limited resources (Brouwer and Powell, 1998).

2.1.3 The need for harmonized metrics of resource use efficiency

As a consequence, improved efficiency is increasingly proposed as the panacea to environmental sustainability, possibly overlooking some of its limitations. At times the term is misused and confused with a range of other metrics, in the plethora of indicators that have been developed to assess nutrient use in agriculture. Furthermore, the quantification of efficiency can pose challenges in the context of data availability and comparability, given the sparse information available in some regions and production systems and discrepancies in the data collection procedures. In addition, the increasing role played by private sector organizations in improving the sustainability of livestock supply chains calls for the development of metrics that can easily be communicated to producers and thus possibly based on concept and data used for the computation of other production management indicators.

The main objectives of this paper are thus (i) to clarify concepts and definitions regarding nutrient use in livestock systems, (ii) to review recent work on Nutrient Use Efficiency (NUE) and (iii) to discuss the relevance, comparative advantages and development opportunities of NUE indicators in the context of sustainable livestock development. While relevant to all nutrients, the paper focuses on nitrogen and phosphorus because of the higher environmental impacts associated to these nutrients in animal production systems (Steinfeld et al., 2006; Godinot et al., 2014; Bouwman et al., 2013; Halberg et al., 2005b; Schröder et al., 2004; Rai et al., 2011; Gerber et al., 2013; Sutton et al., 2013a; Schröder et al., 2011).

2.2 “New” versus recycled N and P resources, implications of these differences

Galloway et al. (2003) introduced the N cascade as a global pattern of reactive N [1] (N_r) circulation in Earth's atmosphere, hydrosphere and biosphere. The authors describe how each new atom of N_r flows through the cascade, impacting ecosystems and human health along the way, involving processes such as tropospheric ozone and aerosol formation, deposition on natural habitats, acidification, eutrophication and climate change.

According to Sutton et al. (2013a), three major human activities involve the transformation of N_2 into new N_r entering the cascade: synthetic N fertilizer production based on the Haber-Bosh reaction (120 Tg N year^{-1}), cultivation of legumes and other crops capable of converting N_2 into N_r through biological N fixation (60 Tg N year^{-1}) and fossil fuel combustion which converts atmospheric N_2 and fossil N into N_r (40 Tg N year^{-1}). Livestock supply chains play an important role in each of these processes: they are estimated to consume about 47.9 Tg of N-fertilizer and 14.3 Tg of N from legumes (Gerber et al., 2013), and while no figure is available for fuel consumption by livestock, the food sector in general is estimated to account for about 30% of the energy consumption worldwide (FAO, 2011a).

The cascade of impact is limited by the capacity of certain systems to accumulate N_r, e.g. forests and unmanaged grasslands storing N_r in soil and biomass, or to host denitrification processes that convert N_r back into N_2, e.g. wetlands, streams and marine coastal regions (Galloway et al., 2003). Despite remaining uncertainties regarding N flows in agro ecosystems (Oenema et al., 2008; Smil, 1999), the low to moderate potential of agricultural systems to act as N_r sink and to produce N_2 indicates that any N_r that does not exit the agricultural system in the form of agricultural products is highly likely to enter the N_r cascade (Galloway et al., 2003).

Furthermore, the high energy requirement of the Haber-Bosh reaction makes N_r generated through this process economically costly and is associated with the environmental impacts of energy production and consumption. It is estimated that industrial N fixation uses about 2% of world energy supply (Sutton et al., 2013a), mostly in the form of natural gas (Steinfeld et al., 2006).

At aggregate level, the efficiency with which new N_r is used in agricultural systems, and livestock supply chains in particular, is thus a pertinent indicator of the potential environmental impact. First, because at aggregated level, only new N_r can be considered to be brought into the system (Figure 2.1). Second, because any inefficiency will generally result in additional N entering the N_r cascade and in fossil fuel use inefficiency, both associated with negative environmental effects.

A similar observation can be drawn for P, although with partially different considerations. In this case, it is mostly the finite nature of P resources that drives the need for efficient use. Rock P reserves are indeed estimated at 71,000 Tg P_2O_5, and given current levels of extraction and

[1] Refers to all forms of N, except for di-nitrogen gas. The metabolism of most organisms cannot use N_2 and thus entirely depend on N_r.

technology, it is projected that these resources will be exhausted at some point in the medium-term future, albeit that projections vary widely from 50-100 years (Cordell et al., 2009) to 370 years (Sutton et al., 2013a). NUE thus appears as the key strategy to extend this period until either new resources or extraction techniques are discovered or systems that tend to fully recycle P are developed. The fact that rock P reserves are also concentrated in few countries accentuates this issue by posing the problem of global access to P resources: almost 90% of P_2O_5 reserves are concentrated in only five countries (Sutton et al., 2013a). The migration of P losses from agricultural sources into the ecosystem also causes environmental and public health concerns, although P is less mobile than N and, contrary to N, accumulates in agricultural soils (Schulte et al., 2010; Schulte and Herlihy, 2007).

2.3 Frameworks for the assessment of nutrient use in livestock systems

Several frameworks have been developed for the assessment of nutrient use in livestock system, tailored to the objectives and scope of the analyses. We can broadly classify them into four categories: nutrient balance (NB), NUE, material flow analysis (MFA), and life cycle assessment (LCA).

2.3.1 Nutrient balance

NB (or nutrient budget) is computed as the difference between aggregated inputs and outputs of a production process, usually expressed in kg nutrients (Nut) year^{-1}. This simple approach is being widely used as an indicator to raise awareness and advise farmers and policy makers on issues related to fertiliser use, manure management and water quality protection (Bassanino et al., 2007; Halberg et al., 2005b; Öborn et al., 2003; Oenema, 2006; Ondersteijn et al., 2002).

NB can be computed at a range of levels in the food supply chain, from unit processes (Aillery et al., 2006; Arriaga et al., 2009; Gourley et al., 2012a), to farms (Godinot et al., 2014), or entire food supply chains (Bouwman et al., 2013; Sutton et al., 2013a) cf. Figure 2.1. NB can also be computed for spatial units integrating several agricultural activities (Gerber et al., 2005).

The relative data parsimony of NB is an advantage of the approach, although aggregated data on regional or system level can be sparse and lead to important uncertainties and biases. Limitations of NB are related to the simplicity of the approach: only inputs and outputs of the analysed production process are quantified, the difference being an estimate of surplus or deficit that aggregates losses, mining and stock changes in a unique figure. This may result in a black-box effect, especially when the analysis is carried out at aggregated level and sub-systems are not analyzed separately. The spatial resolution at which the analysis is carried out also matter: the redistribution of P and N is likely to be heterogeneous, so while N and P balances may be fine at the farm level, within farm (i.e., field or sub-field scale) surpluses and deficits may occur.

Furthermore, although a surplus can generally be interpreted as an indicator of environmental pressure, no information is provided on the nature or the likelihood of the environmental impact.

Different metrics have been used to express the NB in an intensity form, either relating surplus or deficit to a surface area, e.g.: kg Nut ha^{-1} $year^{-1}$ (Halberg et al., 2005b) or to units of output, e.g. kg Nut $year^{-1}$ kg^{-1} output (Leach et al., 2012).

2.3.2 Nutrient use efficiency

Drawing on the same data as NB, NUE is a dimensionless indicator computed as the ratio between the aggregated amount of nutrients in the outputs and in the inputs. As NB, NUE can be computed for different systems, e.g. a field with crop cultivation, a herd, or an entire chain, supporting decision making at various levels (Figure 2.1). Similar to the NB approach, NUE does not provide direct information on environmental impacts. However, performed at an aggregated level encompassing all processes in the supply chain, NUE interestingly informs on the efficiency with which new nutrients are used. Knowledge of the overall efficiency of a chain is however insufficient to identify and guide decision making at the process level.

Several shortcomings of NUE are described by Godinot et al. (2014) who provide technical adjustments, including the extension of system boundaries to account for the life cycle of products, the clarified definition of end products, and the computation of net flows and changes in stock. These developments allow for a greater comparability of results and prevent bias related to "purchase-release" effect (NUE is altered if the same value is added to input and output) and to unaccounted stock changes, e.g. in soil organic matter. Despite these refinements, some of the more intrinsic limitations of NUE remain, including the fact that it does not provide information on overall pressure nor on the environmental impacts and that, when calculated at supply chain level, it does not inform on the distribution of inefficiencies in the chain, or on their location.

In animal production, NUE has been evaluated at animal level (Powell et al., 2010), farm level (Nevens et al., 2006; Powell et al., 2010; Gourley et al., 2012a) or food system level e.g. (Suh and Yee, 2011), cf. following section.

Derivatives of NUE include the crop recovery efficiency, which calculates the nutrients in harvested products as a proportion of nutrient fertilization (Conant et al., 2013); or the partial factor productivity of applied nutrients, which accounts for quantity of harvested crop product per quantity of nutrient applied (Kg output kg^{-1} Nut).

2.3.3 Material flow analysis

MFA, or substance flow analysis, is used to map and quantify flows of selected elements through production systems (Brunner and Ma, 2009; Hashimoto and Moriguchi, 2004; Liu et al., 2008; Senthilkumar et al., 2012). This approach has been used in agriculture to assess nutrient fluxes, accumulation of hazardous substances (heavy metals) and sources of environmental pressure. MFA is built on input-output models applied to each unit-process along the supply chain, and

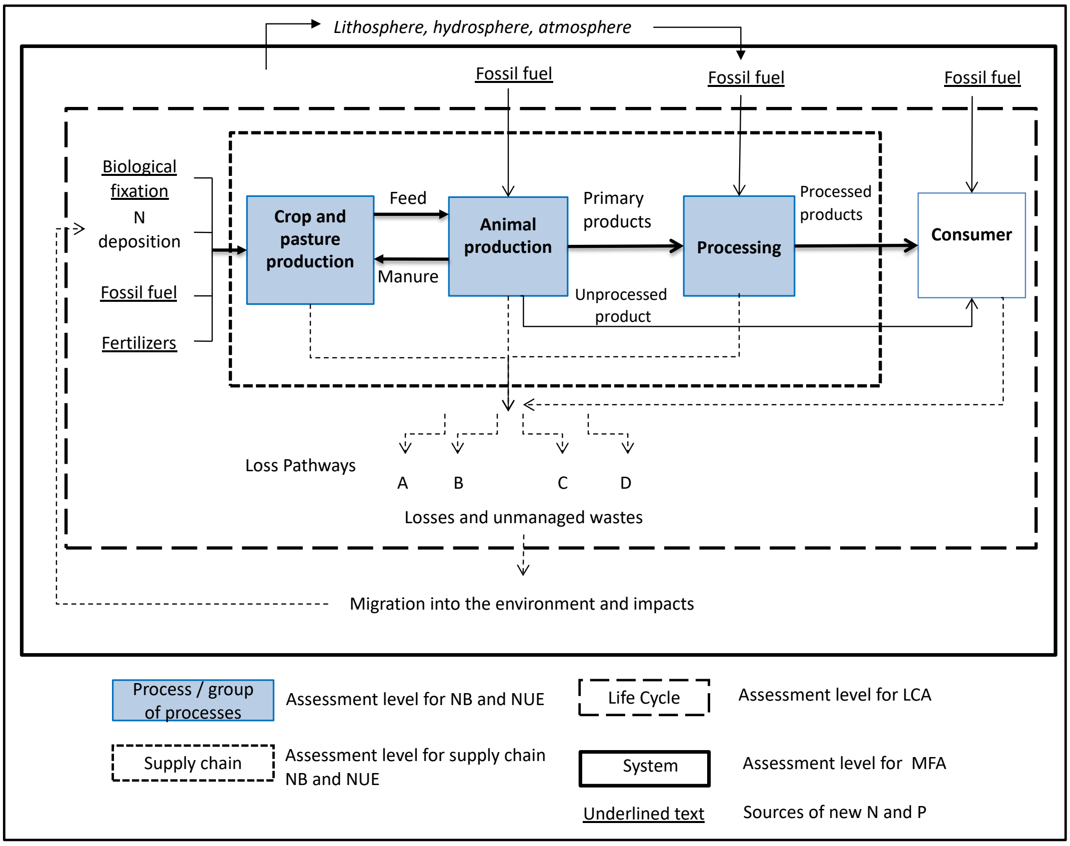


Figure 2.1: Main flows of nutrients and levels for the analysis of nutrient use in livestock systems.

connected to each other (Cooper and Carliell-Marquet, 2013). Several studies applied MFA to map P flows throughout regional (Cooper and Carliell-Marquet, 2013; Senthilkumar et al., 2012; Suh and Yee, 2011; Wu et al., 2014a) or global food production systems (Cordell et al., 2009; Liu et al., 2008).

Starting with one input or chemical element in inputs, e.g. N_r from biological N fixation, MFA maps and quantifies the flow of this element throughout the entire system or through a defined sub-system. MFA informs on the forms of losses (pressure) but not on their impacts on the environment, e.g. on N_r migration to surface water (eutrophication) but not on the impact this may have on biodiversity.

Results from MFA can be used to compute several indicators. For example, Suh and Yee (2011) compute NUE at different stages and levels of food supply, based on an MFA of the US food system.

Whereas MFA is an effective approach to better understand *hotspots* of nutrient losses and to develop mitigation strategies (Cordell et al., 2009), it requires large amounts of high-resolution data, which is often not available on a regional to global scale.

2.3.4 Life cycle assessment

LCA takes a reverse perspective to MFA, taking a unit of product as reference and looking at all upstream (and downstream) activities and related environmental impacts. It is a holistic accounting approach that captures environmental pressure related to the production, usage and disposal (life cycle) of a product or a service (Guinée et al., 2002). Characterization factors are used to estimate the related environmental impacts, reported to a unit of product (Impact kg^{-1} output). Originally developed for industrial processes, LCA has been extended to agricultural studies. A growing number of studies have used LCA to assess the environmental impacts associated with livestock commodities. While the reference unit is always a unit of product, LCA assessment can be computed at different levels of aggregation, from a single supplier to a production system or entire region (Cederberg and Mattsson, 2000; De Boer, 2003; De Vries and De Boer, 2010; MacLeod et al., 2013; Opio et al., 2013). Most of the recent studies focus on climate change, land use, fossil energy use, eutrophication and acidification (De Vries and De Boer, 2010; Xue and Landis, 2010).

Relying on consolidated procedures and international standards (ISO 14040, 2006), LCA is commonly accepted as a valuable environmental management tool for decision-makers (Curran, 2013). As MFA, LCA is however a data intensive approach, which can represent a considerable constraint to its use, especially at high levels of aggregation.

2.4 Nutrient Use Efficiency: relative advantage and applications

NUE demonstrates a number of relative advantages compared to the other approaches reviewed. NUE and NB are basically two different expressions of the same information. Compared to NB, NUE however has the advantage of expressing the potential impact in an intensity form (per unit of product), and therefore cancelling the effect of the size of the activity (cf. Table 2.1). Although NUE does not provide direct information about the environmental impacts of agriculture, at aggregated level, it informs on the performance with which new N and P are used. This provides a critical insight into potential environmental and resource issues related to nutrient use in livestock production.

NUE can be calculated at different scales, embracing one, several or all of the supply chain involved in the supply chain and thus supporting performance improvement at various levels. Computing supply chain NUE as the multiplication of NUE in each process or group of processes avoids the "black box" effect but is obviously much more data-intensive.

NUE can be coupled with other indicators. NB is a natural complement, using the same information but providing an overall estimate of losses and stock change. Losses and stock changes however do not have the same environmental implications and should thus be possibly differentiated in the computation (Godinot et al., 2014). This is especially the case for P, which

Table 2.1: Overview of main approaches used to assess nutrient use in livestock supply chains

Approach	Definition	Scale	Environmental impacts	Data requirements
Nutrient balance	Difference between nutrients inputs and nutrients outputs.	From process to entire supply chain. Can be applied to geographical area including several processes.	Partial information: computes an aggregated amount of surplus, losses or mining.	+
Nutrient use efficiency	Ratio between nutrients outputs and nutrients inputs.	From process to entire supply chain.	Partial information as NB. At Chain level, informs on the efficiency of nutrient use.	+
Life Cycle Assessment	Environmental impact per unit of product.	Supply chain	Aggregate environmental impact per unit of output.	+++
Material Flow Analysis	Map of quantified material flows in the system.	System	Characterizes fluxes of environmental pressure.	+++

has a greater potential of retention within soils than N. Furthermore, long time positive/negative balances or inefficiencies may potentially lead to accumulation/depletion of nutrients (P in particular) in soils (Gourley et al., 2012b). Attention should therefore be paid either to the timeframe of the analysis (a long timeframe would capture accumulation/depletion), or to the existing soil P status in the case of short term analysis. Gourley et al. (2012b) thus argue that while farm-level N NB and NUE can greatly assist management decision, P NB and NUE can't, unless they are combined with soil fertility levels and accumulation/depletion trends. They also note the spatial and temporal heterogeneity of P NB and NUE, further complicating their use for decision support.

The other two reviewed approaches provide more information on environmental pressure (MFA) and impacts (LCA), with however the major drawback of being much more data-hungry. It is also not certain whether the full mapping of flows or the quantification of environmental impacts is actually necessary to inform decision making at the supply chain level. Since any loss of nutrient is likely to cause negative environmental impacts and poses issues with regard to the management of finite resources, one may argue that benchmarking supply chain level NUE provides relevant information to guide nutrient management: any improvement of NUE is likely to generate environmental benefits. The quantification of these benefits would however require implementing analyses such as MFA or LCA.

2.4.1 Nutrient Use Efficiency in the literature

NUE has been used by a large number of authors to assess nutrient use in livestock systems, at animal, farm and supply chain level. Table 2.2 provides a non-exhaustive summary of this literature, reporting results by species and level of analysis. Only N and P were considered in this review, given the scarcity of references addressing other nutrients.

Many studies have assessed animal level NUE, generally computing NUE as the percentage of nutrient in feed that is recovered in edible products. Results are therefore comparable and differences between studies mostly related to species and management practices, e.g. feed rations, climate, animal health, etc. Computations at this level are quite common and the results presented in table 2.2 are only a fraction of existing literature. They nevertheless tend to confirm a pattern of decreasing efficiencies for N and P, as we move from poultry to pig, dairy cattle, and beef cattle. Results found for all species are lower than could be expected when considering those for single species. This may be related to bias in the geographic coverage of assessments at specie level – predominantly addressing industrialized countries – whereas studies addressing all species are global.

NUE assessments carried out at farm and system level encompass livestock rearing as well as other activities on the farm, such as crop and pasture management. The comparability of these studies is more challenging than for animal level NUE.

First, because of the many potential methodological discrepancies. An important source of inconsistencies is the selection of the flows that are considered in the calculation, e.g. the inclusion/exclusion of manure and other non edible products in the outputs, and the inclusion/exclusion of non-purchased nutrients in the inputs. In table 2.2, results are given separately for studies including or excluding non edible products from the outputs. Regarding dairy and beef cattle, for which several published research could be gathered, NUE computed including non edible product is generally higher than NUE excluding non edible products. This is a logical result given the NUE computation formula. Ranges of results however greatly overlap and this relation is not observed in other species, for which limited literature is available. The lack of harmonized data regarding the nutrient content of the various inputs and outputs, as well as differences in the temporal scale of the analysis are further sources of discrepancy. Differences in approach are well explained by diverse purposes and users of single analyses but represent a major issue when producing aggregated assessments of the sector and providing guidance to producers and policy makers (Oenema et al., 2003; Gourley et al., 2012b). In addition to these methodological issues, Oenema et al. (2003) describe several sources of biases and errors that can occur during the analysis and can affect results.

The second major challenge regarding comparability of results at farm level is the great diversity of activities and processes that can be combined in a farm and that have an effect on the overall NUE. This issue calls for a disaggregated NUE analysis or for completing the farm level NUE analysis with other assessments of single processes.

As for animal level, farm level P NUE tends to be higher than N NUE. This trend was observed by previous authors (Domburg et al., 2000) and is to be related to the many pathways of N losses, in gaseous or liquid form.

No studies were found to assess NUE at supply chain level (life-cycle-NUE, cf. Figure 2.1), including fertilizers production and post-farm gate processing.

Table 2.2: Overview of main approaches used to assess nutrient use in livestock supply chains

	Animal level NUE (%)				Farm and system level NUE (%)			
	N		P		N		P	
	Range	References	Range	References	Range	References	Range	References
Dairy cattle	15 to 35[a]	[1,2,3,4,5,6,7, 8,9,10]	19 to 60[a]	[1,4,12,13]	15 to 41 [c] 15 to 55[d]	[4,8,14,15,16, 17,18,19,20,21] [15,17,19,20,21]	31-48[c] 56 to 74[d]	[4,2,22,23] [20,21,24]
Beef cattle	4 to 8[a]	[4,24]	14 to 28[a]	[4]	7 to 38[c] 26-34[d]	16,25] [15]	21-44[d]	[4,25]
Pig	10 to 44[a]	[4,20,21]	34[1]	[4]	50[c] 41-45[d]	[4] [26,27]	37[d]	[4]
Poultry	25 to 62[a]	[4,20,21, 25,28,29]	34 to 58[a]	[4,28]	39[c] 35 to 48[d]	[29] [27]	61[d]	[29]
All species combined	7.1 to 10.5[a] 74.1[b]	[30,31]	4 to 19[a] 36[b]	[30,32]	5 to 45[d]	[11,27]		

Note: when several studies are referenced, the range make reference to mean values provided in the studies; when one study only is referenced, the range or the single value provided in the study are reported in the table.
[a]Calculated as the percentage of nutrient in feed that is recovered in edible products.
[b]Calculated as the percentage of nutrient in feed that is recovered in edible and non-edible products, including recycled manure.
[c]Calculated as percentage of total nutrient input (including deposition and biological fixation) recovered in edible outputs.
[d]Calculated as percentage of total nutrient input (including deposition and biological fixation) recovered in edible and non-edible outputs.
References: 1: Arriaga et al. (2009), 2: Gourley et al. (2012a), 3: Powell et al. (2010), 4: Domburg et al. (2000), 5: Jonker et al. (2002), 6: Nadeau et al. (2007), 7: Ryan et al. (2011), 8: Oenema et al. (2012), 9: Kohn et al. (1997), 10: Powell et al. (2013), 11: Simon et al. (2000), 12: Bannink et al. (2010), 13: Klop et al. (2013), 14: Godinot et al. (2014), 15: Bassanino et al. (2007), 16: Oenema (2006), 17: Nevens et al. (2006), 18: Segato et al. (2010), 19: Schröder et al. (2003), 20: Spears et al. (2003a), 21: Hristov et al. (2006), 22: Spears et al. (2003b), 23: Plaizier et al. (2014), 24: Oenema and Tamminga (2005), 25: Watson et al. (2002), 26: Schulte and Herlihy (2007), 27: Öborn et al. (2003), 28: Kratz et al. (2004), 29: D'Haene et al. (2007), 30: Sutton et al. (2013a), 31: Van der Hoek (1998), 32: Suh and Yee (2011).

2.5 Conclusion

Drawing on relatively simple and generally accessible information, NUE can inform the efficiency of animal production systems and is confirmed as a valuable indicator to guide decision making and improve sustainability of nutrient management (Schröder et al., 2011; Sutton et al., 2013a). Computed at supply chain level, it notably characterizes (i) the use of finite resources (rock P and fossil fuel for N_r production), and (ii) the losses of nutrients per unit of product, likely to have impacts on the environment and public health.

Relating nutrient use to a unit of product, NUE can be considered a production oriented indicator that producers and the private sector can use to benchmark activities and monitor progress. This is likely to ease its adoption into existing monitoring and reporting systems. Relying on the same data, NB is a complementary indicator that may be used in parallel to NUE.

Recent assessments have improved NUE calculation but further methodological developments are required to deliver effective support to decision making. These include (i) the full incorporation of the life cycle thinking in NUE assessment and the identification of the major causes of inefficiency along the supply chains, to better address longer and increasingly complex food systems, (ii) the harmonization of definitions regarding system boundaries, inputs/outputs and timeframes that can be consistently applied, to ensure comparability of results and (iii) the development of common datasets on nutrient content in feed and other nutrient sources, to overcome current data shortage and shortcomings. Several initiatives address these needs, among which the Livestock Environmental Assessment and Performance (LEAP) multi-stakeholder partnership ([2]). Initiated in 2012, LEAP focuses on the development of approaches, metrics and databases to guide environmental sustainability decisions in the livestock sector.

[2]http://www.fao.org/partnerships/leap/en/

These developments could be the initial steps towards a better understanding of the variability in NUE within systems and the related gap between production and processing units operating at relatively high levels of efficiency and those which on the contrary are less efficient. Quantifying the efficiency gap and understanding underlying drivers is key in helping the sector to make relatively quick and cost effective sustainability gains, by generalizing the adoption of best practices (Gerber et al., 2013). Further investigations are also required to test the hypothesis that, at aggregated level, the efficiency with which new nutrients are used is indeed a good proxy for environmental and public health impacts related to nutrient management.

Chapter 3

A comprehensive framework to assess the sustainability of nutrient use in global livestock supply chains

Aimable Uwizeye[1,2,3], Pierre J. Gerber[1,2], Rogier P.O. Schulte[3], Imke J.M. de Boer[1]

[1]Animal Production Systems group, Wageningen University, PO Box 338, 6700 AH, Wageningen, the Netherlands
[2]Food and Agriculture Organization of the United Nations, Animal Production and Health Division, Viale delle Terme di Caracalla, 00153 Rome, Italy
[3]Teagasc – Crops, Environment and Land Use Programme, Johnstown Castle, Wexford, Ireland

Journal of Cleaner Production, 129, 647-658.

Abstract

The assessment of the performance of nutrient use along livestock supply chains can help to identify targeted nutrient management interventions, with a goal to benchmark and to monitor the improvement of production practices. It is necessary, therefore, to develop indicators that are capable to describe all nutrient dynamics and management along the chain. This paper proposed a comprehensive framework, based on life-cycle approach, to assess the sustainability of nitrogen and phosphorus use. The proposed framework represents nutrient flows in typical livestock supply chain from the "*cradle-to-primary-processing-gate*", including crop/pasture production, animal production and primary processing stage as well as the transportation of feed materials, live-animals or animal products. In addition, three indicators, including the life-cycle nutrient use efficiency (Life-cycle-NUE), life-cycle net nutrient balance (Life-cycle-NNB) and nutrient hotspot index (NHI) were proposed, and tested in a case study of mixed dairy supply chains in Europe. Proposed indicators were found to be suitable to describe different aspects of nitrogen and phosphorus dynamics and, therefore, were all needed. Moreover, the disaggregation of Life-cycle-NUE and Life-cycle-NNB has been investigated and the uncertainties related to the choice of the method used to estimate changes in nutrient soil stock have been discussed. Given these uncertainties, the choice of method to compute the proposed indicators is determined by data availability and by the goal and scope of the exercise.

3.1 Introduction

The management of nutrients in agricultural systems has contributed to a tremendous growth of plant and livestock production (Galloway et al., 2003; Steinfeld et al., 2006). With the future increase of world population, livestock production will continue to play a key role in sustaining food production, and economic growth (Alexandratos and Bruinsma, 2012). Livestock production has, and will continue to have, an important role in nitrogen (N) and phosphorus (P) cycles from a global perspective (Bouwman et al., 2013) because livestock ingest more than 80% of total harvested N and P and deliver only 20% of N and P in edible products for human consumption (Sutton et al., 2013a).

Losses to the environment are inherent to these biochemical cycles, for example, through leaching and runoff from fertilised soils and manure storage; through soil erosion, or through emissions to the atmosphere, such as ammonia (see e.g. Galloway et al. (2003)). These losses potentially threaten water, soil and air quality, but also climate, biodiversity and human health (Gerber et al., 2013; Schulte et al., 2010; Sutton et al., 2013a). These losses also relate to the use of fossil resources, such as fossil fuel and P rock (Cordell et al., 2009). Achieving better nutrient management is thus an important aspect to improve environmental performance in the livestock sector.

Improving the efficiency of N and P use has been identified as a main strategy to achieve global food security and sustainability (Meena et al., 2015; Oenema et al., 2014; Sutton et al., 2013a). Several studies described the quantification of nutrient use efficiency (NUE) as a relevant approach of nutrient management (Gerber et al., 2014; Sutton et al., 2013a) or as an indicator of nutrient pressure in agro-environmental policy (Halberg et al., 2005b; Powell et al., 2010).

The assessment of NUE, however, commonly suffers from several shortfalls, as identified by Gerber et al. (2014). These shortfalls relate to the fact that nutrient data are aggregated at regional or system level which can lead to high uncertainties and biases associated with methodological considerations related to nutrient flows considered, or modelling approaches (Godinot et al., 2014; Oenema et al., 2003). For example, the simplistic quantification of inputs and outputs flows results in black-box effect that aggregates nutrient losses, mining and stock change in unique figure of nutrient surplus. So far, NUE assessments have been conducted at farm, regional or global scales considering either crop production or livestock production. However, livestock systems are globalised, necessitating a chain analysis. Only few studies assessed NUE at chain level to understand the overall performance of delivering animal products at regional or global level. Exceptions include Suh and Yee (2011) and Wu et al. (2014a), but these studies focus on food system. These studies do not identify hotspots of nutrient loss, which is required to support more targeted nutrient management decisions and improve sustainability.

To assess the sustainability of increasingly globalised livestock systems and guide continuous improvement, it is necessary to develop indicators that are relevant at regional scale and integrate complex interactions along the supply chain (Dolman et al., 2014; Gerber et al., 2014). The objective of this paper, therefore, is to propose a comprehensive framework to assess the sustainability of nutrient use in global livestock supply chains to support benchmarking and monitoring of production practices. We propose a set of three indicators that we test in the case study of mixed dairy systems in Europe. We discuss their relevance as well as methodological uncertainties related to their computation.

3.2 Materials and Methods

3.2.1 Nutrient flows and system boundaries

Figure 3.1 summarises nutrient flows in a typical livestock supply chain. The system boundary includes three interconnected stages: crop and pasture production, animal production, and primary processing of livestock products as illustrated in Figure 3.1. Each stage has multiple nutrient inputs and outputs. These inputs and outputs might be produced in different regions and might be transported to the farm or processing plant for their utilisation. We consider, therefore, transport as part of the system boundary; the associated nutrient flows are thus included in the framework. Fossil fuel consumption is also considered at other stages of production, such as field operations and processing.

The system boundary represents the "*cradle-to-primary-processing-stage*" of the life-cycle of livestock commodities, including the cradle-to-farm gate, transportation of animals or animal products to the primary processing, and then through the primary processing gate. For instance, the primary processing stage is limited to the primary milk-processing factory or animal slaughter for meat processing. We assume that all nutrient flows occur during the same year and that the system operates in a fixed state, with the exception of soil stock change, to avoid the allocation between processes over years. Nutrient stocks and changes thereof take place at each stage of production (Özbek and Leip, 2015; Suh and Yee, 2011), whereas nutrients in products include all co-products delivered at each stage. So far, several studies don't account for changes in nutrient stocks, while this flow may be essential for a low input system or for P management strategy. The description of nutrient flows accounted for is illustrated in table 3.1.

3.2.2 The proposed framework

The proposed framework includes three indicators: Life-cycle-NUE[1], life-cycle net nutrient balance (Life-cycle-NNB)[2] and nutrient hotspot index (NHI)[3]. Life-cycle-NUE is a dimensionless indicator, which is independent of units, and it defines the efficiency upon which nutrient inputs are recovered in end-products, and it considers nutrient mobilisation, use, change in nutrient stocks and recycling. It is similar to "entire chain NUE" suggested by Suh and Yee (2011), which refers to linear and multidirectional processes which takes into account three processes (crop, animal and processing), changes in nutrient stocks and recycling (e.g. manure, crop residues), but different from a "full-chain NUE", which refers to the linear and unidirectional input – output approach, which is estimated as a product of NUE of each stage of the supply chain (Sutton et al.,

[1] NUE_N: *Nitrogen use efficiency*; NUE_P: *Phosphorus use efficiency*
[2] NNB_N: *Net nitrogen balance*; NNB_P: *Net phosphorus balance*
[3] NHI_N: *Nitrogen hotspot index*; NHI_P: *Phosphorus hotspot index*

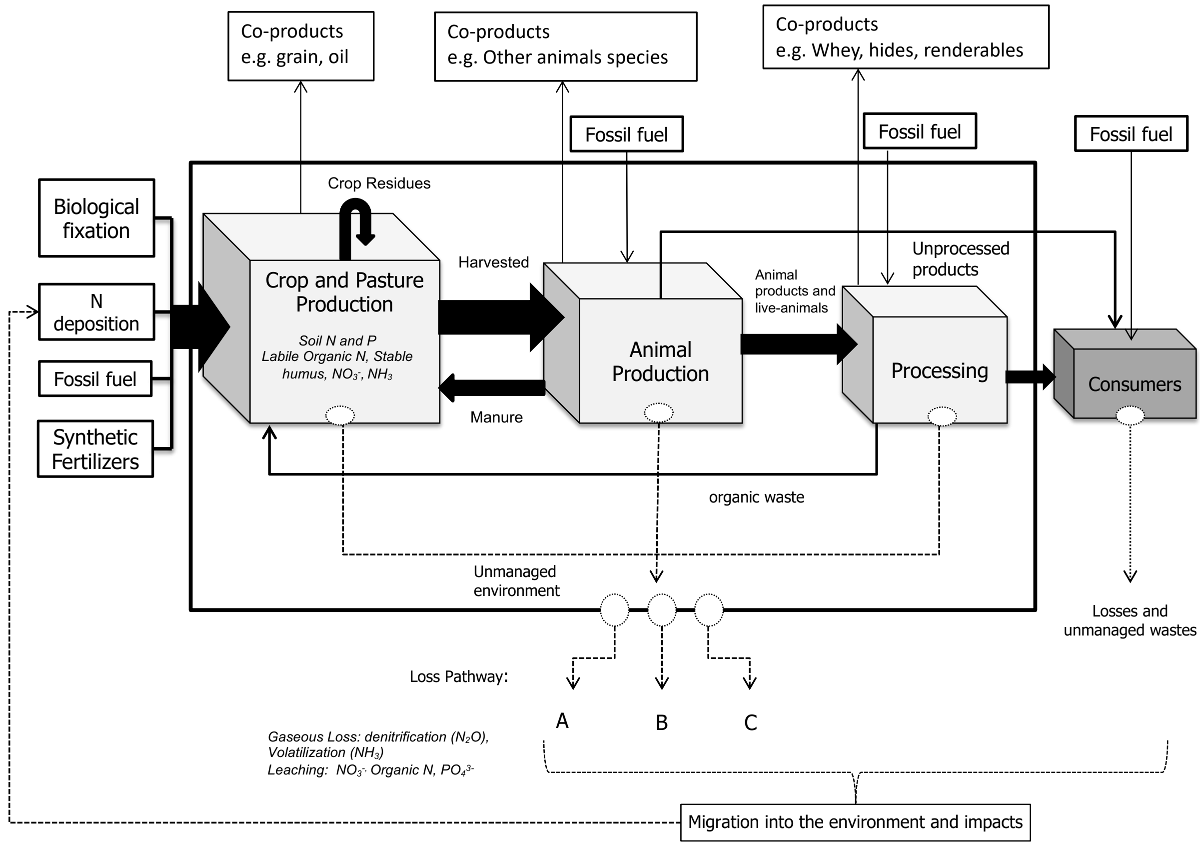


Figure 3.1: System boundary and nutrient flows in livestock supply chain (adapter from Gerber et al. (2014))

2013a). Life-cycle-NNB expresses the number of nutrients that is not used for either products or the build-up of soil fertility regardless of the actual location where they occur in the chain. In this paper, for N, NNB refers to losses to the environment, including emissions into the soil, water and air, whereas, for P, it refers to the sum of losses and no useful stock change. The NNB is reported per unit of land used. This indicator is different to the “nutrient footprint”, which is computed per unit of product or per capita. The third indicator, NHI, is defined as the relative distribution of nutrient balances in the chain. It quantifies the evenness of hotspots of nutrient balances and thus gives information to refine nutrient management strategies. The combination of these three indicators is proposed to concisely provide relevant and complementary information on nutrient management performance, and provides direct information on the efficiency of nutrient use, nutrient balance per ha, and distribution of nutrient pressures along the chain.

Furthermore, this framework adopts the life-cycle approach approach and is consistent with biophysical relationships among supply chain stages. Each stage is characterised by inputs, outputs and loops to different unit processes. The indicators are calculated based on a model which operates in two steps. First, it estimates aggregated nutrient flows at the unit process,

Table 3.1: Description on nutrient flows for livestock supply chains e.g. dairy cattle

Stage	Type of flow	Flow name	Flow description
Crop and pasture production	Inputs	Fuel	Quantity of fuel x emissions factors
		Synthetic fertilizer	Quantity of N and P applied to crop or pasture
		Manure	Quantity of manure applied or deposited to the land x N or P content
		Crop residues	Quantity of biomass in aboveground and below ground x N or P content
		Biological N fixation	Quantity of N fixed by microorganisms
		Atmosphere deposition	Quantity of N or P deposited throughout rainfall
	Outputs	Harvested crop or forage / grazed pasture	Quantity of DM harvested x N or P content of crop
		Net nutrient balance	N or P loss through volatilisation, runoff and leaching
		Soil stock change	(see *section 2.3*)
Animal production	Inputs	Feed	Quantity of feed component x N or P content per feed component
		Fuel	Quantity of fuel x emissions factors
		Feed additives	Quantity of feed additives x N or P content
	Outputs	Milk	Quantity of milk x N or P retention in milk
		Calves	Number of calves x N or P retention in tissues
		Culled cows	Number of culled cows x N or P retention in tissues
		Manure recycled/used	(Quantity of excreted manure per animal category + litter and bedding - Quantity of manure loss or disposed of) x N or P content
		Net nutrient balance	Nutrient loss via volatilisation, leaching and surface runoff
Processing	Inputs	Milk	Quantity of milk to be processed x N or P content of milk
		Live-animals	Number of culling cows
	Output	Milk	Quantity of milk processed x N or P content
		Carcass	Quantity of meat produced (55% of live animals) equivalent to N or P retention in edible tissues
		Other non-edible products	Quantity of non-edible products (% of live weight) x N or P content
		Net nutrient balance	Loss associated with organic waste or waste water

before considering the entire chain in a calculation matrix (Suh and Yee, 2011), see A.1. Second, it estimates the three indicators based on equations described below.

Nutrient use efficiency

Nutrient use efficiency at each production stage

NUE at each stage of the supply chain (NUE_i) is estimated as a vector of processes, using the following formulae (Eq. 3.1):

$$\mathbf{NUE}_i = \frac{\mathbf{PROD}_i + \mathbf{SC}_i}{\mathbf{PROC}'_i + \mathbf{IMP}'_i + \mathbf{RES}_i} \tag{3.1}$$

where $\mathbf{PROD}_i$ denotes the product output from each process of supply chain i, $\mathbf{SC}_i$ denotes the amount of stock change within each process of supply chain i, $\mathbf{INP}_i$ denotes the internal amount of product input to each process of supply chain i, $\mathbf{IMP}_i$ denotes the amount of import to supply chain i and $\mathbf{RES}_i$ denotes the amount of "new" nutrient input to each process of supply chain i from either nature (e.g. biological N fixation), industrial process (e.g. synthetic fertilizer) or other agricultural activities (e.g. recycled manure from other livestock species).

Life-cycle nutrient use efficiency

Life-cycle-NUE is expressed as one unit of nutrient in end-products, divided by the amount of "new" nutrient mobilised in the supply chain to produce it. These amounts of "new" nutrient mobilised $\mathbf{RES}_i$ at each stage are estimated as follows (Eq. 3.2):

$$\mathbf{RES}_i^* = \mathbf{RES}_i.(\mathbf{PROD}_i + \mathbf{INP}_i - \mathbf{IMP}_i + \widehat{\mathbf{SC}_i})^{-1} \tag{3.2}$$

Life-cycle-NUE, therefore, was calculated as an inverse of the 3rd element of the matrix $\mathbf{RES}_\mathrm{i}^*$, which corresponded to the amount of nutrient mobilised to produce 1 kg of nutrient in end-products at processing stage (Suh and Yee, 2011).

Net nutrient balance

The NNB is defined at each stage and supply chain levels. It is calculated as nutrient input minus nutrient output and stock change. For entire supply chain, NNB is also defined as "Life-cycle-NNB" and is calculated as follows (Eq. 3.3):

$$\text{Life-cycle-NNB} = \frac{\sum \mathbf{NNB}_i.AF_i}{A} \tag{3.3}$$

where $\mathbf{NNB}_\mathrm{i}$ refers to nutrient losses at stage i, $\mathrm{AF_i}$ refers to the biophysical allocation factor between products at stage i, and A refers to total land used at supply chain. Allocation is required to split nutrient losses between the product(s) used in the livestock supply chain and the product(s) exported out of the supply chain (e.g. grain). Allocation factors were calculated based on the relative mass of a nutrient in a specific product compared to the total mass of that nutrient in all products.

Nutrient hotspot Index

The NHI is calculated as the standard deviation of NNB divided by the average of NNB of all stages of the supply chain. A high NHI implies the occurrence of one or major hotspots of nutrient balance in the chain, whereas a low NHI implies that nutrient balance is evenly distributed along the chain, and nutrient management should target each production stage. NHI is estimated as follows (Eq. 3.4):

$$NHI = \frac{\sigma_{(NNB_i)}}{\mu_{(NNB_i)}} \tag{3.4}$$

where σ is the standard deviation of NNB for all stages of a supply chain, and μ is the corresponding average of NNB for all stages of a supply chain.

3.2.3 Nutrient modelling

Nutrient modelling at each stage of the supply chain requires specific methods to estimate potential nutrient losses and stock change.

Crop and pasture production

Nitrogen

Nutrient modelling in crop and pasture production requires information on all nutrient flows, including inputs, soil stock change, losses and removal in harvested biomass. However, no dataset contains direct or full measurement of soil stock changes and losses. Existing models, therefore, attempt to model either soil stock change or losses (or, alternatively, assume a value), and then deduct the other variable from a mass balance. We identified three methodological approaches to estimate soil stock change and N-losses in the literature. Method 1 (M1) assesses N losses using a field balance and assuming that soil stock change is 0, (Eurostat, 2013). Method 2 (M2) estimates N losses following nutrient dynamics at different unit processes based on several assumptions (Velthof et al., 2009). Method 3 (M3) assumes minimum and maximum NUE and applies them to an empirical equation to estimate stock change (Özbek and Leip, 2015). For the case study, M2 approach was used to illustrate the framework.

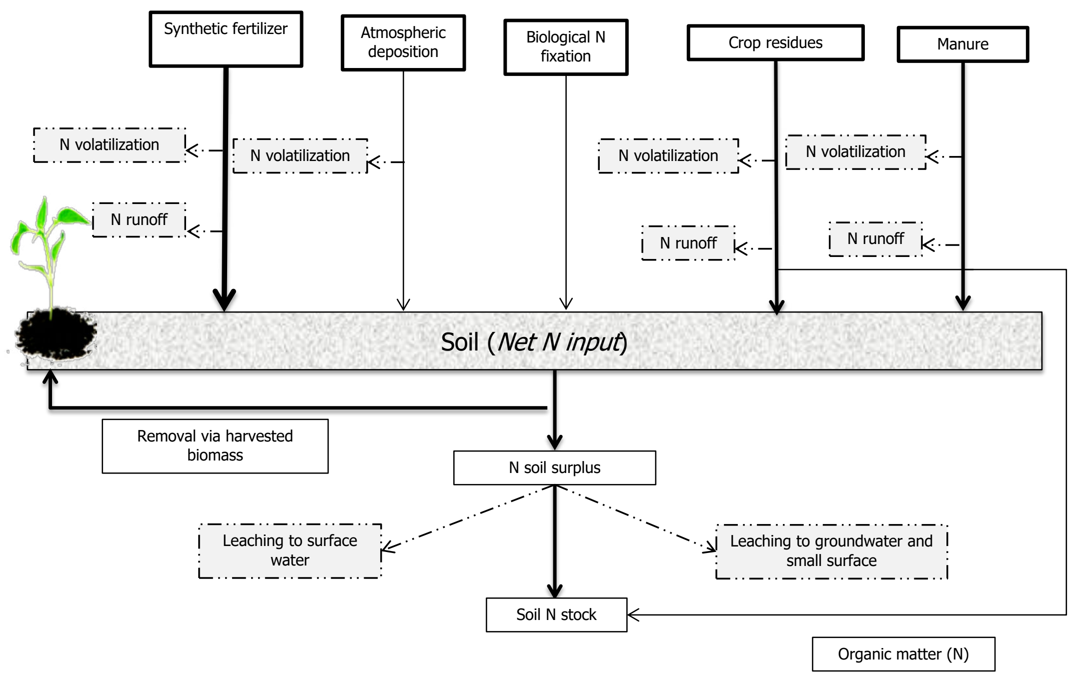


Figure 3.2: N application transfer into the soil (adapted from Velthof et al. (2009))

The approach for NNB_N estimations is described in Figure 3.2. NNB_N, refers to all emissions of polluting N compounds into the soil, water and air (OECD/EUROSTAT, 2007) resulting from N inputs including synthetic fertilizer, manure, crop residues, biological N fixation and atmospheric deposition. The NNB_N was estimated as follows (Eq. 3.4:

$$NNB_N = Atm_N + Runoff_N + Leaching_N \tag{3.5}$$

where Atm_N refers to N emissions to the air (e.g. NH_3, N_2O, NO_x) estimated based on IPCC method (IPCC, 2006), whereas RunoffN refers to surface N runoff, estimated using country-specific runoff fractions (Velthof et al., 2009) and LeachingN refers to N leaching estimated as the fraction of N soil surplus leached to ground and surface water (Velthof et al., 2009). The N stock change was estimated as follows (Eq. 3.6):

$$Stockchange_N = (Organic_N + SoilSurplus_N) - Leaching_N \tag{3.6}$$

where $Organic_N$ refers to a part of organic N that is not mineralised and directly enters soil stock (Dollé and Smati, 2005; Velthof et al., 2009) and $SoilSurplus_N$ refers to the sum of N soil surplus, and $Leaching_N$ refers to N leaching. N stock change can be negative when net N input is less than the N removal via harvested biomass.

Phosphorus

The general approach for NNB_P estimation is described in Figure 3.3. To estimate NNB_P , agricultural soils are divided into two categories: deficient/optimum fertility and excess fertility soils, using soil P profiles of European soils for cropland and grassland (Tóth et al., 2013b). We define deficient/optimum soils for soil P content less than 50 mg P kg^{-1} of soil, and excess fertility soils at soil P content greater or equal to 50 mg P kg^{-1} of soil (do Carmo Horta and Torrent, 2007; Tóth et al., 2014). For deficient/optimum soils, soil P surplus is considered a sustainable build-up (Schulte et al., 2010; Ulén et al., 2007), because it is needed to increase plants P uptake. A sustainable build-up is defined as the amount of P required to overcome a possible P deficiency or to maintain soil P levels (Batjes, 2011). For excess fertility soils, soil P surplus is divided into a sustainable build-up and an unsustainable build-up. An unsustainable build-up is defined as the undesirable accumulation of P into soils that cannot be recovered by plants unless soil management changes over time and it may increase the risk of P leaching/runoff. Total P inputs to labile and stable pools include synthetic fertilizer, manure, crop residues, sewage, atmospheric deposition and weathering. The NNB_P was estimated as follows (Eq. 3.7):

$$NNB_P = (Erosion_P + Runoff_P) + Unsustainable_P \tag{3.7}$$

where ErosionP refers to P losses via soil erosion, RunoffP refers to surface P runoff assumed at 5% of P inputs for grass and 10% of P inputs for crop (Sattari et al., 2012; Hart et al., 2004), whereas UnsustainableP refers to a fraction of soil P accumulation which is not needed for soil P build-up (Batjes, 2011; Jordan et al., 2005; Schulte et al., 2010), and is estimated using P recovery fractions, which correspond to potential P retentions (Batjes, 2011).

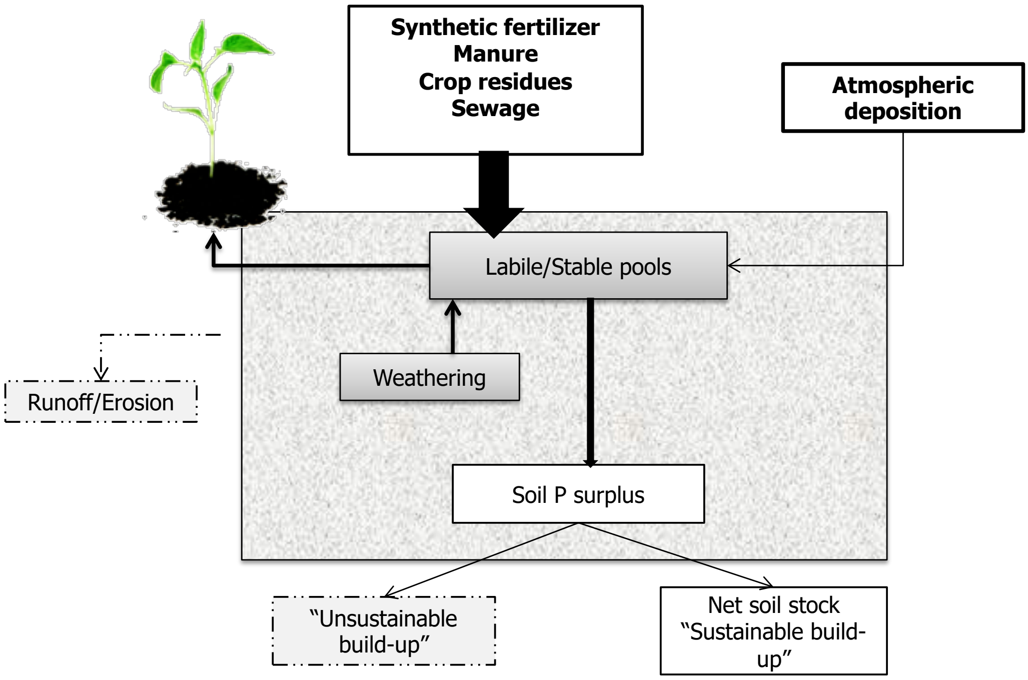


Figure 3.3: P application transfer into the soil (adapted from Sattari et al. (2012) and Batjes (2011))

Animal production

The calculations of nutrient flows at the animal production stage are performed to consider the proportion of animals in each cohort, and the transfer rate of animals between cohorts. Characteristics of animals include the average weight, growth rate, mortality, fertility and replacement rate. Feed nutrient intake is divided into nutrients retained into edible and non-edible products. Nutrient excretion is estimated by subtracting nutrient retention from the nutrient intake. The nutrient losses related to manure management are estimated using specific emissions factors for the volatilisation, denitrification and leaching of nutrients. Moreover, recycled manure is considered a valuable output for its fertilizer value and is therefore included in the animal output (Gerber et al., 2013).

Processing stage

After farm gate, live animals and animal products are processed, providing edible and non-edible products. The edible products are either sold to market or sent to further processing, whereas the non-edible products are inputs into industrial processes (e.g. hides, tallow, renderable, and blood). Nutrient losses may occur through losses of solids and wastewater which, if not treated, may be considered as harmful to the environment, see Figure 3.1.

3.2.4 Case study

System definition

To illustrate the proposed framework, we applied it to European mixed dairy systems. Mixed dairy systems are defined as systems combining dairy farming with other associated agricultural activities, such as crop production or other animal species husbandry, and are characterised by an intensive exchange of products and services between these different activities (Oomen et al., 1998; Robinson et al., 2011). At least 10% of dry matter used as the feed comes from crop production. These systems account for around 80% of the dairy cows population and 84% of milk in Western Europe (Gerber et al., 2013).

Data sources for the case study

Data on N inputs were compiled from Global Livestock Environmental Assessment Model (GLEAM) database. GLEAM was developed to assess the environmental performance of livestock supply chains at global level for the reference year of 2005 (Gerber et al., 2013). The GLEAM database contains detailed country-specific data on crop-specific synthetic fertilizers, crop residues, manure applied, manure deposited to grazing areas, feed materials, feed rations, feed intake, animal numbers, feed and animal parameters, and processing yields. However, additional data were obtained on synthetic N fertilizer (Conant et al., 2013), atmospheric N deposition (Dentener, 2006), and biological N fixation (Herridge et al., 2008).

Data on synthetic P fertilizer were obtained from FAOSTAT (2017). P contents in manure, crop and crop residues were estimated based on N/P ratio extracted from Bouwman et al. (2013) and Feedpedia (2012), whereas P contents in forages and animal products were obtained from Jongbloed (2011, personal communication), Bouwman et al. (2013) and Wu et al. (2014b). Moreover, data were collected on atmospheric P-deposition (Mahowald et al., 2008), P-erosion (Sattari et al., 2012), soil P profile for cropland and grassland (Tóth et al., 2013b), and soil P recovery and retention potential (Batjes, 2011). However, data on N and P inputs for imported crops were estimated using FAOSTAT trade matrix considering the main exporter countries to each country. For grazing animals, the percentage of manure loss on non-agricultural areas was assumed to be 2-5% of the total manure produced. At milk processing plant, the rate of nutrient losses was estimated to 2.3% for N and 0.15% for P (Verheijen et al., 1996). For meat processing at a slaughterhouse, 12% of N and P in non-edible products were assumed lost through organic waste and wastewater (Verheijen et al., 1996). A summary of aggregated N and P data is provided in the appendix (Table A.1). A descriptive statistical analysis and calculation model were carried out using R software (R Core Team, 2013) to estimate N and P flows and indicators. The correlation analysis was used to select valuable indicators, and a t-test was used to compare average NUE and NNB values obtained for different methods.

3.3 Results and discussion

3.3.1 Life-cycle nutrient use efficiency, life-cycle net nutrient balance, and nutrient hotspot index

Indicators are displayed in Figure 3.4 for N and in Figure 3.5 for P. There was no correlation across the three indicators (Life-cycle-NUE$_N$ vs. Life-cycle-NNB$_N$: $R^2 = 0.01$; Life-cycle-NUE$_N$ vs. NHI$_N$: $R^2 = 0.02$; Life-cycle-NNB$_N$ vs. NHI$_N$: $R^2 = 0.25$). Life-cycle-NUE$_N$ ranged from 27% in the Republic of Serbia to 48% in Lithuania. Life-cycle-NNB$_N$ ranged from 35 kg N ha^{-1} in Lithuania to 207 kg N ha^{-1} in the Netherlands. In addition, NHI$_N$ ranged from 86% in Estonia to 133% in the Netherlands. High values of NHI$_N$ indicate the presence of hotspots of losses, despite high Life-cycle-NUE$_N$ that may be associated with it. For instance, France had a high Life-cycle-NUE$_N$ (44%), combined with a high Life-cycle-NNB$_N$ (105 kg N ha^{-1}) and a high NHI$_N$ (123%). Schröder et al. (2011) described a similar situation where high NUE was associated with high losses but did not define any indicator to measure it.

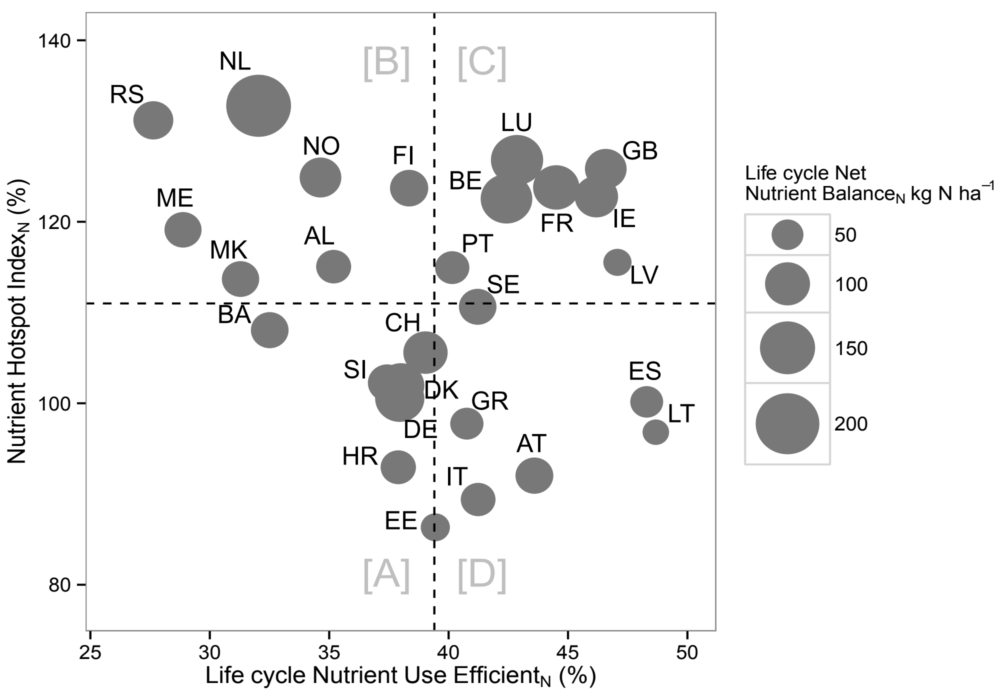


Figure 3.4: Relationships among Life-cycle-NUE$_N$, Life-cycle-NNB$_N$ and NHI$_N$. The horizontal and vertical dotted lines in the plot indicate the average for country nutrient hotspot index (NHI$_N$) and Life-cycle-NUE$_N$, respectively. A, B, C, and D represent fours sectors for nutrient management. Countries are indicated in the plot by their country-acronym: AL: Albania, AT: Austria, BE: Belgium, BA: Bosnia and Herzegovina, CH: Switzerland, DE: Germany, DK: Denmark, EE: Estonia, ES: Spain, FI: Finland, FR: France, GB: United Kingdom (of Great Britain and Northern Ireland), GR: Greece, HR: Croatia, IE: Ireland, IT: Italy, LT: Lithuania, LU: Luxembourg, LV: Latvia, MK: Macedonia, NO: Norway, NL: Netherlands, PT: Portugal, ME: Montenegro, MK: The former Yugoslav Republic of Macedonia, RS: Republic of Serbia, SE: Sweden, SI: Slovenia.

For P, the Netherlands had lowest values for Life-cycle-NUE$_P$ (46%) and highest values for Life-cycle-NNB$_P$ (36 kg P ha^{-1}) and for NHI$_P$ (161%). Latvia had the highest Life-cycle-NUE$_P$ (85%), associated with lowest values for Life-cycle-NNB$_P$ (2 kg P ha^{-1}) and for NHI$_P$ (97%). There was a correlation across the three indicators (Life-cycle-NUE$_P$ vs. Life-cycle-NNB$_P$: $R^2 = 0.68$; Life-cycle-NUE$_P$ vs. NHI$_P$: $R^2 = 0.49$; Life-cycle-NNB$_P$ vs. NHI$_P$: $R^2 = 0.38$).

These results for Life-cycle-NUE are different from those of Sutton et al. (2013a). Those authors found that, global "full-chain NUE", defined as the nutrients in final products divided by nutrient inputs, was 8% for N and ranged from 12% to 20% for P. Higher efficiencies in our study can be related to the multidirectional approach used, which accounts for internal loops, recycling, stock changes, and losses. For P, we considered the sustainable P build-up. Those are not the case in the study by Sutton et al. (2013a), which uses a unidirectional method based on simplistic input-output approach, focusing on two stages of the chain. Moreover, we focused on dairy systems in Europe from cradle-to-primary-processing, whereas Sutton et al. (2013a) considered the global agro-food system.

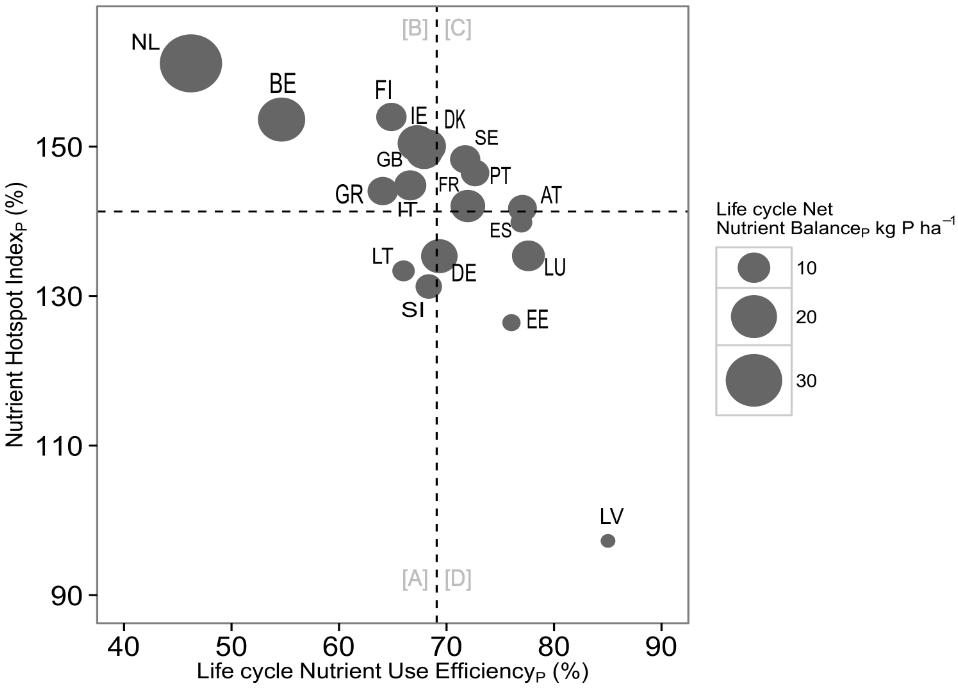


Figure 3.5: Relationships among Life-cycle-NUE$_P$, Life-cycle-NNB$_P$ and NHI$_P$. The horizontal and vertical dotted lines in the plot indicate the average for country nutrient hotspot index (NHI$_P$) and Life-cycle-NUE$_P$, respectively. A, B, C, and D represent fours sectors for nutrient management. Countries are indicated in the plot by their country-acronym: AT: Austria, BE: Belgium, DE: Germany, DK: Denmark, EE: Estonia, ES: Spain, FI: Finland, FR: France, GB: United Kingdom (of Great Britain and Northern Ireland), GR: Greece, IE: Ireland, IT: Italy, LT: Lithuania, LU: Luxembourg, LV: Latvia, NL: Netherlands, PT: Portugal, SE: Sweden, SI: Slovenia.

The plots in Figure 3.4 and Figure 3.5 are divided in four sectors, namely A, B, C and D, based on overall averages of Life-cycle-NUE and NHI for European countries investigated. Sector A is characterised by a below average Life-cycle-NUE and NHI. Life-cycle-NNB$_N$ is generally low to medium for all countries. Countries in this sector may present a potential to improve Life-cycle-NUE while maintaining the low level of hotspot and Life-cycle-NNB. The improvement interventions would focus on the reduction of nutrient inputs, the increase of nutrient uptake by plants, and the reduction of N emissions during manure management. By contrast, sector B includes the highest Life-cycle-NNB for both N and P associated with high NHI. Thus, higher NHI values than in sector A indicate that interventions would focus pre-eminently on one stage of the chain: crop and pasture production. Moreover, sector C is characterised by high NHI, high Life-cycle-NUE, and medium Life-cycle-NNB. From a nutrient management perspective, there is a potential to reduce nutrient balance and hotspots while maintaining high levels of Life-cycle-NUE. Interventions in sectors B and C would focus on the reduction of nutrient inputs and nutrient losses at all stages of the chain. Sector D presents the low NHI, high Life-cycle-NUE, and low Life-cycle-NNB. While this Sector can be seen as optimum we identify some caveats. For example, high levels of Life-cycle-NUE_N may be a reflection of low N input levels, with Lithuania as a case in point compared to other countries in the same sector. Thus, the depletion of nutrients in the soil, for instance, might be negative for NUE in the long-term. For P, there is an apparent synergy between P status into soil and Life-cycle-NUE. For instance, Latvia benefited from a build-up of P in its deficient fertility soils that resulted in the highest Life-cycle-NUE_P. The high Life-cycle-NUE_P suggests that soil P concentrations should be monitored in the long-term to maintain this level of efficiency after that the soil P profile reaches the optimum level.

3.3.2 Variability related to nutrient use efficiency and net nutrient balance

Disaggregating nutrient use efficiency along the chain

Table 3.2 shows the results for disaggregated NUE_N and NUE_P. Both NUE_N and NUE_P differed depending on the stage in the supply chain and they decreased in the order of: processing > animal production > crop and pasture production, except for Croatia, Germany, Greece, and Italy, where NUE_N decreased in the order of: processing > crop and pasture production > animal production.

Processing stage had high NUE for all countries at around 95% for N and 98% for P. There were no difference between N and P at this stage, because the amounts of N and P losses via organic waste and wastewater are comparable. At animal production, NUE values ranged from 69% to 84% for N and from 95% to 99% for P. These results are higher compared to those reported in literature (range 15% - 36% for NUE_N and 19% - 60% for NUE_P) (Gerber et al., 2014). The reason is that most of previous studies focused on the efficiency of recovering nutrient into edible animal products, thus excluding manure and non-edible products as a valuable output (Godinot et al., 2014; Gourley et al., 2012b; Powell et al., 2010). In this study, we included manure as an output for its fertilizer value.

Low values of NUE were found in crop and pasture production, ranging from 59% to 81% for N and from 55% to 92% for P. The difference between N and P are related to the biochemical processes of both elements into soil. N is labile and mineralised after a short period, whereas, P is stable and tends to accumulate for long-term and may contribute to build-up P into soil.

Table 3.2: Results of dissagregated nitrogen and phosphorus use efficiency at each stage of the dairy supply chains

Country	Crop/pasture production		Animal Production		Processing	
	NUE_N (%)	NUE_P (%)	NUE_N (%)	NUE_P (%)	NUE_N (%)	NUE_P (%)
AL	70	-	75	-	95	-
AT	80	83	74	99	96	85
BE	71	66	81	97	96	85
BA	70	-	71	-	95	-
HR	76	-	72	-	95	-
DK	73	76	70	97	96	88
EE	80	86	69	97	96	87
FI	69	75	77	97	96	88
FR	72	80	83	97	96	86
DE	73	79	70	95	96	87
GR	77	74	74	97	95	82
IE	74	75	84	97	96	85
IT	78	77	71	97	96	86
LV	76	92	82	98	96	84
LT	81	79	81	97	96	84
LU	70	84	82	97	96	87
ME	64	-	70	-	95	-
NL	60	55	72	95	96	87
NO	65	-	74	-	96	-
PT	72	81	76	97	96	87
RS	60	-	70	-	95	-
SI	73	81	72	97	96	86
ES	77	83	81	97	96	87
SE	74	80	75	97	96	88
CH	73	-	73	-	96	-
MK	67	-	71	-	95	-
GB	73	76	84	97	96	88

Countries are indicated in the plot by their country-acronym: AL: Albania, AT: Austria, BE: Belgium, BA: Bosnia and Herzegovina, CH: Switzerland, DE: Germany, DK: Denmark, EE: Estonia, ES: Spain, FI: Finland, FR: France, GB: United Kingdom (of Great Britain and Northern Ireland), GR: Greece, HR: Croatia, IE: Ireland, IT: Italy, LT: Lithuania, LU: Luxembourg,LV: Latvia, MK: Macedonia, NO: Norway, NL: Netherlands, PT: Portugal, ME: Montenegro, MK: The former Yugoslav Republic of Macedonia, RS: Republic of Serbia, SE: Sweden, SI: Slovenia.

Results of NUE_N differ from those of Lassaletta et al. (2014) (range 30% - 75%) and OECD (2008) (range 41% - 78%), whereas, results of NUE_P differ from those of OECD (2008) (range 37% - 86%%). The differences may be related to data inputs or method used. Lassaletta et al. (2014) and OECD (2008) used a field balance, with stock changes assumed to equal 0, whereas, in our study, the stock change was considered as part of the production. Moreover, for P, we considered the build-up of P into soil sustainable for deficient fertility soils. However, in excess fertility soils, the accumulation of P is unsustainable and may present a high risk of P loss in long-term (Jordan et al., 2005; Sharpley, 1999), therefore, it was considered a loss. The role of build-up of P in soil to enhance plant uptake was discussed previously by several authors (Cordell et al., 2009; Jordan et al., 2005; Schröder et al., 2011; Schulte et al., 2006; Sharpley et al., 2013).

Disaggregating net nutrient balance

Life-cycle-NNB provides an aggregated average nutrient loss per unit area used, which can mask different loss profiles along the supply chain, as well as differences between non-point and point source losses. Given that most losses are taking place during the crop and pasture stage, we disaggregated the NNB of this stage to understand the differences between feed resources. The Figure 3.6 and Figure 3.7 illustrated average NNB results for the in-country crop, in-country grass and ex-country crop in comparison to the Life-cycle-NNB for N and P. In-country crop and in-country grass refer to feed crops and forage produced within a country, whereas ex-country crop refers to imported feed crops from different regions of the world.

Across countries, the variability of NNB values was large for N. The highest averages NNB_N were estimated at 256 kg N ha^{-1} for in-country grass and at 145 kg N ha^{-1} for the in-country crop in the Netherlands. For ex-country crop, NNB_N was high in the Republic of Serbia at 46 kg N ha^{-1}. The lowest NNB_N were estimated at 16 kg N ha^{-1} for the in-country grass in Latvia, at 27 kg N ha^{-1} for the in-country crop in Estonia, and at 11 kg N ha^{-1} for the ex-country crop in Ireland. There were no significant differences in average NNB_N between the in-country crop and in-country grass, whereas the difference was significant between the in-country and ex-country crops. The reason is that countries with high livestock density, such as Belgium and the Netherlands, might apply manure to cropland at rates that are above plant nutrient requirements in 2005. Despite methodological differences, these results are consistent with those of Velthof et al. (2009); Leip et al. (2011); Csathó and Radimszky (2012) for Belgium, Luxembourg and the Netherlands, but slightly lower from those of Lassaletta et al. (2014) who reported values of N losses higher than 50 kg N ha^{-1} for most of European countries. The differences may be related to the type of crop or grass considered as well as to the inputs and outputs data.

Similarly to N, there was large variability in P pressures across countries. The highest NNB_P was obtained at 23 kg P ha^{-1} for in-country grass and at 58 kg P ha^{-1} for the in-country crop in the Netherlands. For ex-country crop, NNB_P was high in Croatia at 10 kg P ha^{-1}. Latvia had the lowest NNB_P for the in-country grass at 1 kg P ha^{-1} and for the in-country crop at 7 kg P ha^{-1}. The lowest NNB_P for the ex-country crop was obtained in Estonia at 3 kg P ha^{-1}. NNB_P was generally higher for in-country crop, compared to the in-country grass and to ex-country crop.

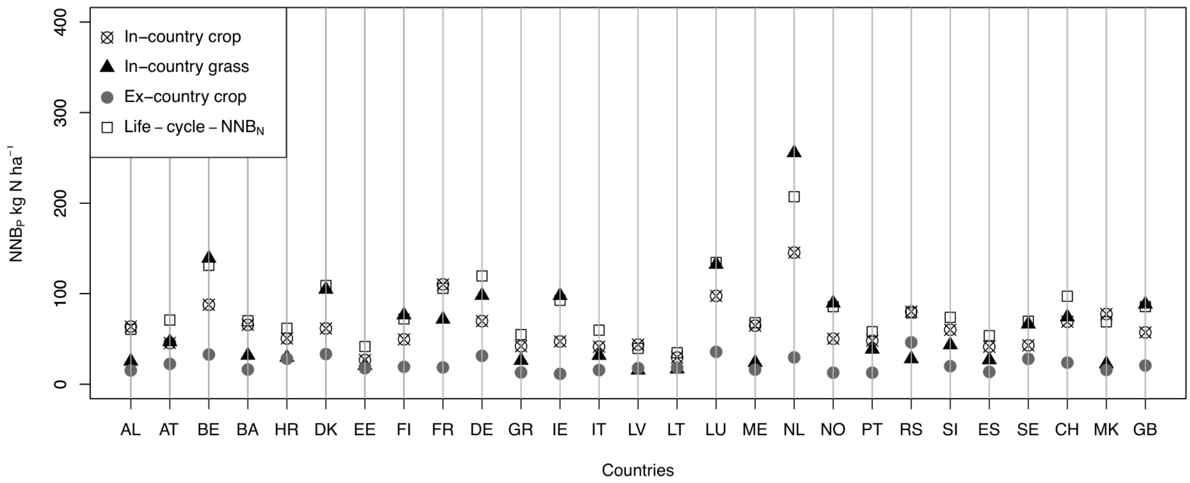


Figure 3.6: Disaggregated NNB$_N$ for different crops and grasses and Life-cycle-NNB$_N$ for different crop and pasture resources. Countries are indicated in the plot by their country-acronym: AL: Albania, AT: Austria, BE: Belgium, BA: Bosnia and Herzegovina, CH: Switzerland, DE: Germany, DK: Denmark, EE: Estonia, ES: Spain, FI: Finland, FR: France, GB: United Kingdom (of Great Britain and Northern Ireland), GR: Greece, HR: Croatia, IE: Ireland, IT: Italy, LT: Lithuania, LU: Luxembourg, LV: Latvia, MK: Macedonia, NO: Norway, NL: Netherlands, PT: Portugal, ME: Montenegro, MK: The former Yugoslav Republic of Macedonia, RS: Republic of Serbia, SE: Sweden, SI: Slovenia.

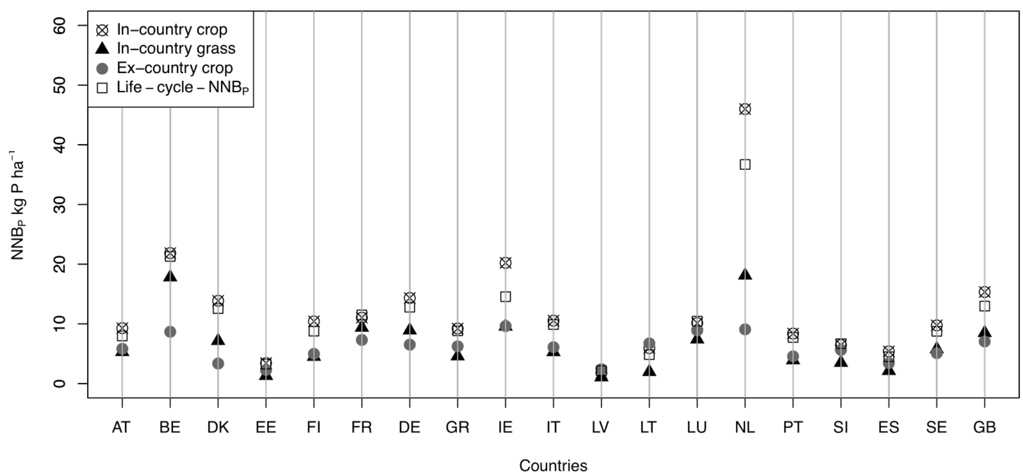


Figure 3.7: Disaggregated averages NNB$_P$ for different crops and grasses and Life-cycle-NNB$_P$. Countries are indicated in the plot by their country-acronym: AT: Austria, BE: Belgium, DE: Germany, DK: Denmark, EE: Estonia, ES: Spain, FI: Finland, FR: France, GB: United Kingdom (of Great Britain and Northern Ireland), GR: Greece, IE: Ireland, IT: Italy, LT: Lithuania, LU: Luxembourg, LV: Latvia, NL: Netherlands, PT: Portugal, SE: Sweden, SI: Slovenia.

The reason is that P rates are higher for the in-country crop, compared to the in-country grass and ex-country crop dominated by soybeans. For instance, 70% of soybeans produced in Brazil is exported to Europe (Cavalett and Ortega, 2009). In this study, NNB$_P$ values were comparable to those obtained for P surplus by the OECD (2008) (range -1-23 kg P ha^{-1}), Eurostat (2014) (range -8-20 kg P ha^{-1}), and Csathó and Radimszky (2012) (range 4-40 kg P ha^{-1}).

Moreover, these modelled NNB$_P$ were higher compared to results of direct measurements of P losses at different scale (Daverede et al., 2003; Hart et al., 2004; Jordan et al., 2005; McDowell and Sharpley, 2001; Schulte et al., 2010). The reason is that NNB$_P$ includes a share of unsustainable P build-up and measures potential, rather than actual environmental pressure from P sources. Thus, to convert P pressure into potential impact, a pressure-pathways-response-impact approach is required (Schulte et al., 2006; Yuan et al., 2014).

3.3.3 Uncertainties related to stock change estimations

Results for N presented above were computed using the method (M2) proposed by Velthof et al. (2009) . Here, we compare these results with those obtained from methods M1 and M3, focusing on the crop and pasture stage. These methods gave contrasting results for NUE$_N$ ($p<0.05$) showed in Figure 3.8. Method M1 based on field balance tends to neglect potential N losses to the environment for low inputs systems. For high input system, however, it resulted in low NUE$_N$ values associated with high risk of losses, which is consistent with previous studies (Lassaletta et al., 2014; Leip et al., 2011). This approach is widely used at the field, farm or regional scales because of its simplicity and was adopted by Eurostat/OECD (Eurostat, 2013) for national nutrient balances reporting. However, the fact that it does not inform on the potential contribution of soil stock change raises concerns with regard to the reliability of results (Gerber et al., 2014).

Method M2 is a process-based and estimates N losses following biochemical dynamics of nutrient in different unit processes. It relies on algorithms that are extrapolated from experimental studies, and on several assumptions based on expert knowledge. These algorithms are used, for example, to estimate N emission factors, N leaching fractions, N excretion in animal manure, whereas assumptions are used to estimate some N flows such as biological N fixation or the amount of manure applied to cropland (see Velthof et al. (2009)). This method delivers N soil stock change from the N balance. The importance to include the contribution of soil N stock change in NUE assessment was discussed in literature (Brock et al., 2012; Godinot et al., 2014; Özbek and Leip, 2015; Velthof et al., 2009). This method was used in different process-based models (Britz and Leip, 2009; Dumas et al., 2011; Velthof et al., 2009). Method M3 is new and is based on the assumptions on NUE values, and then estimates soil stock change (Özbek and Leip, 2015). To test this method, we used default minimum and maximum NUE values for all countries (33% and 85%) in an empirical equation that resulted in comparable NUE$_N$ for all countries. For all countries, NUE$_N$ values obtained using M3 were lower than values obtained using M1 and M2, except for the Netherlands and Republic of Serbia. The reason is that nutrient input to crop and pasture is relatively high in the Netherlands, because of relatively high livestock density: 2.5 dairy cows per ha, against an average of 0.98 among European countries (Eurostat, 2014). In the case of the Republic of Serbia, however, the explanation lies in the low nitrogen recovered in crops and pasture (98 kt N), compared to the nitrogen input (208 kt N). In such circumstances, the default maximum NUE value assumed in M3 is higher than the computed

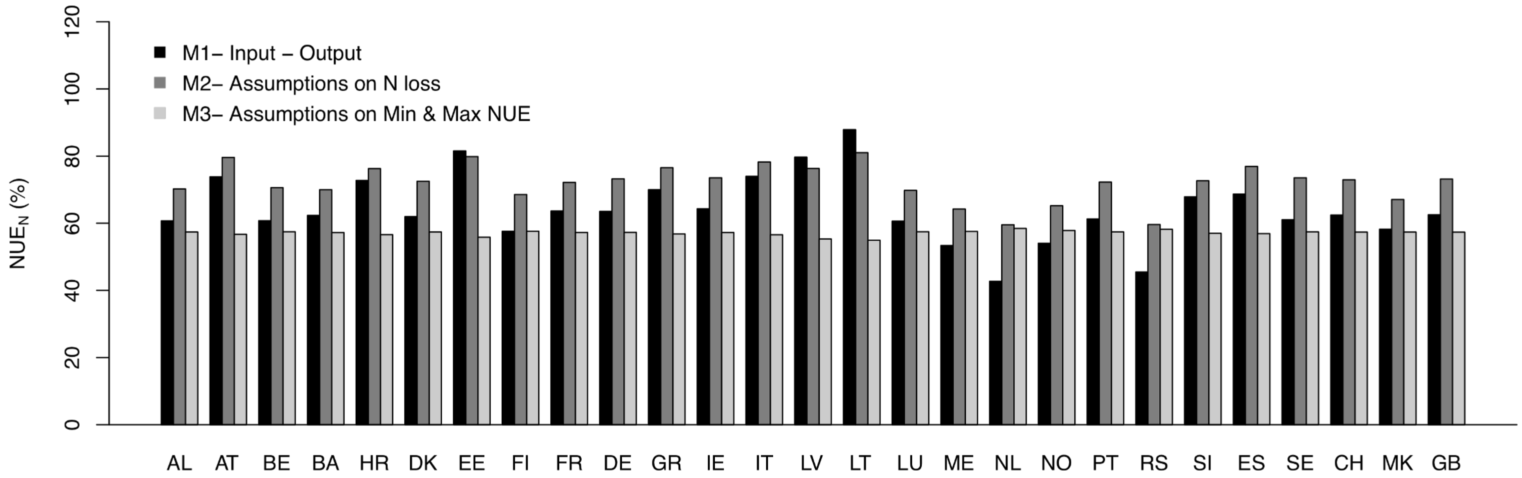


Figure 3.8: Comparison of three methods for NNB$_N$ in crop and pasture production system. Countries are indicated in the plot by their country-acronym: AL: Albania, AT: Austria, BE: Belgium, BA: Bosnia and Herzegovina, CH: Switzerland, DE: Germany, DK: Denmark, EE: Estonia, ES: Spain, FI: Finland, FR: France, GB: United Kingdom (of Great Britain and Northern Ireland), GR: Greece, HR: Croatia, IE: Ireland, IT: Italy, LT: Lithuania, LU: Luxembourg, LV: Latvia, MK: Macedonia, NO: Norway, NL: Netherlands, PT: Portugal, ME: Montenegro, MK: The former Yugoslav Republic of Macedonia, RS: Republic of Serbia, SE: Sweden, SI: Slovenia.

NUE. For low inputs systems, M1 and M2 resulted in similar NUE_N values, such as for Estonia or Lithuania.

None of the three methods is fully satisfactory for regional or global NUE assessment. In the absence of data and background knowledge of the system, it may be appropriated to use the method M1. In addition, method M1 is suitable for general scoping analysis aiming to get a broad insight into potential NUE. Thus, results should be interpreted with caution especially for low input systems because nutrient losses may be underestimated. Method M2 seems most adapted for representative regional and global NUE analysis; however, it requires more data and expert knowledge of the system to validate the assumptions made. This method requires continuous efforts to develop specific parameters by crop, country or animal species and to improve its limitations in accounting for the large geo-climatic variability that governs nutrient losses factors, manure excretion factors, and nutrient dynamics into soil. In addition, uncertainties around N loss estimates end up in the residual term, N stock change. Regarding M3, this approach may be adapted for nutrient assessment at field or region scale where it can rely on comprehensive empirical observations of NUE. This method, however, needs some valid assumptions to upscale results from experimental studies to national or regional scales. Due to lack of data at these larger regional scales, this method may not yet be suitable for life-cycle-NUE analysis; however, it is a useful tool to assess the contribution of soil stock change to the NUE.

Regarding P, the field balance results in similar uncertainties like for N. However, there are additional sources of uncertainties related to the method used to estimate the sustainable and unsustainable P build-up shares of the soil stock change. To estimate soil stock change, data on soil P contents were not statistically representative for some countries, for instance, Luxembourg, which may increase the uncertainties for this country. Moreover, P recovery fractions were delivered from the soil modelling that may be a major source of uncertainties (Batjes, 2011). Furthermore, other sources of uncertainties may be associated with data collection and default parameters such as emission factors used. However, these uncertainties are not discussed in this study.

3.4 Conclusion

We presented a framework for regional and global life-cycle nutrient use efficiency assessment based on life-cycle thinking. This framework accounts for more processes, pools, recycles and end-uses than previous studies. Moreover, we proposed three indicators, including Life-cycle-NUE, Life-cycle-NNB and NHI required to fully describe the nutrient dynamics of livestock supply chains. We concluded that these indicators are suitable and all describe different aspects of nutrient dynamics and are therefore all needed. Moreover, we demonstrated that the combination of these indicators gives relevant and complementary information to concisely benchmark and monitor nutrient management performance. The disaggregation of Life-cycle-NUE and Life-cycle-NNB showed a large variability and potential for interventions across production stages and regions of the supply chain. Furthermore, we also found that estimation of stock change improves the accounting of indicators despite the uncertainties of their estimations. Given these uncertainties, more than one method is available to conduct a nutrient use efficiency assessment at the chain level, depending on the goal and scope of the exercise, as well as data availability. Based on our results, further research should apply this framework at a regional and global level for contrasting livestock supply chains, and on further refining and harmonising methodologies. Moreover, a representative life-cycle NUE assessment requires a large amount of activity data and parameters, which may be hardly available in some parts of the world. Thus, for consistency sake, further research should identify minimum data requirements.

3.5 Acknowledgements

This project is funded by the Teagasc Walsh Fellowship Scheme (Ref: 2012230), the FAO Livestock Environmental Assessment Performance (LEAP) Partnership (GCP /GLO/369/MUL), and Animal Production and Health Division (AGA) of Food and Agriculture Organization of the United Nations.

Chapter 4

Selective improvement of global datasets for the computation of locally relevant environmental indicators: a method based on global sensitivity analysis

Aimable Uwizeye[1,2,3], Pierre J. Gerber[1,2], Evelyne A. Groen[1], Mark A. Dolman[1,4], Rogier P.O. Schulte[3,4], Imke J.M. de Boer[1]

[1]Animal Production Systems group, Wageningen University, PO Box 338, 6700 AH, Wageningen, the Netherlands
[2]Food and Agriculture Organization of the United Nations, Animal Production and Health Division, Viale delle Terme di Caracalla, 00153 Rome, Italy
[3]Teagasc – Crops, Environment and Land Use Programme, Johnstown Castle, Wexford, Ireland
[4]Farming Systems Ecology, Wageningen University, P.O. Box 430, 6700 AK, Wageningen, the Netherlands

Environmental Modelling & Software 96 (2017) 58e67

Abstract

Several global datasets are available for environmental modelling, but information provided is hardly used for decision-making at a country-level. Here we propose a method, which relies on global sensitivity analysis, to improve local relevance of environmental indicators from global datasets. This method is tested on nitrogen use framework for two contrasted case studies: mixed dairy supply chains in Rwanda and the Netherlands. To achieve this, we evaluate how indicators computed from a global datasets diverge from same indicators computed from survey data. Second, we identify important input parameters that explain the variance of indicators. Subsequently, we fix non-important ones to their average values and substitute important ones with field data. Finally, we evaluate the effect of this substitution. This method improved relevance of nitrogen use indicators, therefore, it can be applied to any environmental modelling using global datasets to improve their relevance by prioritizing important parameters for additional data collection.

Software and data availability

The modelling done in this paper is performed in the R project for statistical computing. R is an open source statistics software and can be downloaded from https://www.r-project.org. Data from a global dataset were obtained from the Global Livestock Environmental Assessment Model (GLEAM) version 2, which was developed by the Animal Production and Health Division (AGA) of Food and Agriculture Organization of United Nations (FAO). The description of GLEAM can be found at http://www.fao.org/gleam/en/. The code is programmed in R and is available at https://github.com/uaimable/GSA.

4.1 Introduction

The sustainability of agricultural systems depends on many social, economic and environmental factors related to technology, practices, and innovation, including a proper management of nutrient flows, such as nitrogen (N) (Sutton et al., 2013a). Insight into the efficiency of nutrient use along livestock supply chains can help to identify targeted nutrient management interventions (Uwizeye et al., 2016a). Recently, Uwizeye et al. (2016a) proposed a comprehensive framework to assess nutrient use performance along livestock supply chains. This framework includes three complementary indicators: life-cycle nutrient use efficiency (life-cycle-NUE), life-cycle net nutrient balance (life-cycle-NNB) and nutrient hotspot index (NHI). Quantification of these indicators, however, is quite data intensive and its application on a regional or global level can be laborious because livestock supply chains are relatively long, internationalized and diverse (Gerber et al., 2013; Henderson et al., 2017). Moreover, data on livestock production is scarce, often outdated and does not reflect the spatial and temporal variability of the livestock systems. Given the limited availability and quality of data, regional averages, default parameters and expert knowledge are often used in the global datasets, despite their high uncertainties and potential biases (Krueger et al., 2012; Uusitalo et al., 2015). These uncertainties in input parameters have an influence on the variance of the model output (Ferretti et al., 2016; Heijungs and Kleijn, 2001; Oenema et al., 2003; Saltelli et al., 2008; Sarrazin et al., 2016a; Heijungs, 1996). Also, averaged or reference values do not reflect the heterogeneity of production units within a given system (Henderson et al., 2017). Gourley et al. (2015), for example, found that milk production per hectare varied by a factor of 12 among grazing dairy farms in Australia. Such variability can also be found for other farm parameters, such as the accumulation of the nutrients in the soil or feed intake. Overlooking this heterogeneity may lead to wrong decision-making or mitigation interventions (Anastasiadis and Kerr, 2013; Henderson et al., 2017).

The effect of the uncertainties of input parameters can be evaluated through sensitivity analysis. Generally, there are two types of sensitivity analyses. First, the local sensitivity analysis (LSA) is based on changing of input parameters around a reference (nominal) value and ranking the magnitude of the effect for each parameter (Campolongo et al., 2007). Second, the global sensitivity analysis (GSA) is be based on the variation of input parameters according to their distribution function, and subsequently determine how much each parameter explains the model output variance (Saltelli et al., 2008; Pianosi et al., 2016). An example of an LSA, widely used in environmental studies (Ferretti et al., 2016), is to determine the effect of a change in one of the input parameters at-a-time on the model results (Groen et al., 2017; Kohn et al., 1997; Oenema et al., 2003; Suh and Yee, 2011). This approach presents some shortfalls, such as assuming that environmental models are approximately linear and additive (Ferretti et al., 2016; Saltelli et al., 2000) or assuming that the variation of one parameter is not associated with any change in the space of all another parameter (Ferretti et al., 2016; Saltelli and Annoni, 2010). The GSA obviates these shortfalls and explores how much each input parameter contributes to the variance of the model output (Ferretti et al., 2016; Pianosi et al., 2016; Saltelli et al., 2008; Sarrazin et al., 2016b; Sin et al., 2011; Uwizeye et al., 2016b; Wolf et al., 2017). Moreover, GSA allows to select the i,mportant input parameters and to simplify the model by fixing the non-important parameters to their average values (Saltelli et al., 2008). GSA approach, therefore, can provide more insights into the improvement of the relevance and reliability of environmental indicators.

An increasing number of global datasets are becoming available for environmental modelling, opening new opportunities for global sustainability analyses e.g. the Global Livestock Environmental Assessment model (GLEAM) (FAO, 2017) or the ecoinvent (Frischknecht et al., 2005). Most of these datasets, however, do provide information which may not always be directly usable at a country or sub-regional level (Elduque et al., 2015). Consequently, indicators computed from available global datasets can hardly be used for a country-level decision-making, which significantly limits their relevance given that most policy decisions are made at that level. The objective of this paper is thus to propose a method to improve the local relevance of environmental indicators computed from global datasets. We first evaluate how indicators computed from a global dataset diverge from the same indicators computed from local survey data. Second, we determine the important input parameters, which explain most of the variance of the indicators computed from a global dataset. Subsequently, we fix the non-important ones to their average values and substitute the important ones with field data. Finally, we evaluate the effect of this substitution. The method is tested in the case of N use performance indicators for two contrasted case studies: mixed dairy supply chains in Rwanda and the Netherlands. In Rwanda, there is less information available on national statistics, whereas, in the Netherlands, the statistics are available in the national and European database.

4.2 Materials and methods

4.2.1 Description of the proposed method

The different steps of the proposed method are summarized in Figure 4.1.

Indicators of N use performance at supply chain level

We used the nutrient use performance framework developed by (Uwizeye et al., 2016a) to assess N management in mixed dairy systems of the Netherlands and Rwanda. The indicators considered are: (i) Life-cycle-NUE$_N$, which defines the efficiency of which N inputs are recovered in the end-products; (ii) Life-cycle-NNB$_N$, which defines the amount of N that is available for loss to the environment; and (iii) NHI$_N$ which defines the relative distribution of N balances along the supply chain. The framework quantifies the N flows in crop/pasture production, animal production, manure management system (MMS), and processing of animal products including the internal processes and loops. The detailed description of this procedure and these indicators can be found in Uwizeye et al. (2016a).

Global sensitivity analysis

We described four steps to perform the GSA. Step 1, we selected the probability density function (PDF) for each input parameter. For input parameters described by an average or reference

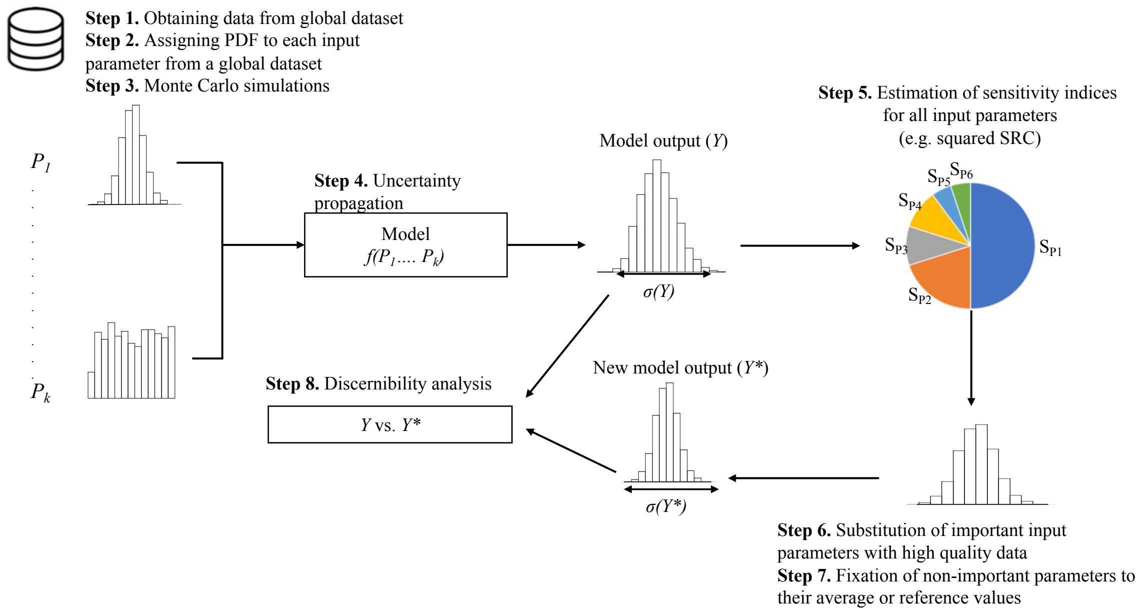


Figure 4.1: Flow diagram summarizing the methodological steps to improve environmental indicators computed from a global dataset based on Groen et al. (2014b) and Saltelli et al. (1999).

value without any information on their variability, a normal distribution is assumed, and a coefficient of variation (CV) of 20% was applied (IPCC, 2006). For input parameters described by fixed minimum and maximum and a specific likely value (e.g. EF), we assign a triangular distribution. Finally, we assign a uniform distribution of input parameters that are only defined by the minimum and maximum, as advised by van Gijlswijk et al. (2004). Step 2, we performed Monte Carlo simulation (MCS), which consists of generating N random values from a specified PDF of each input parameter (Groen et al., 2014b; Saltelli et al., 2008). The sample size was set to $N = 5{,}000$. We denoted the matrix of Monte Carlo simulations of each input parameter used in the model as P, the number of the input parameters as k, the size of the MC runs, as N, and the number of the model evaluations during the GSA as r. Step 3, the uncertainties of k input parameters were propagated through the model that results in the sample of the model outputs in this case for the Life-cycle-NUE$_N$, Life-cycle-NNB$_N$, and NHI$_N$. Step 4, we used the squared standardized regression coefficients (squared SRC) method for GSA (Groen et al., 2017; Saltelli et al., 2008) to estimate the sensitivity indices. Several examples of the application of SRC method are found in the literature (Basset-Mens et al., 2009; Cosenza et al., 2013; Sin et al., 2011; Groen et al., 2017; 2014a; Sattari et al., 2016; Sin et al., 2011; Uwizeye et al., 2016b). The squared SRC method is based on multilinear regression between the model output and the input parameters, and is noted as follows (Eq. 4.1):

$$\mathbf{Y}_i = \beta_0 + \sum_{j=1}^{k} \beta_j . \mathbf{P}_{ij} + \epsilon_i \tag{4.1}$$

where $\mathbf{Y}$ refers to the model output, the constant β_0 represents the intercept, β_j refers to the slope (regression coefficient), P refers to the sampling matrix of input parameters, k the number of the input parameters and ϵ_i is the error term, which is assumed to be normally distributed with a constant variance. The squared SRCs S_j^2 are estimated as follows:

$$S_i^2 = \frac{V(\mathbf{P}_j)}{V(\mathbf{Y})}\beta_j^2 \tag{4.2}$$

where $V(\mathbf{P}_j)$ refers to the variance of each input parameter, $V(\mathbf{Y})$ refers to the variance of the model output and β_j^2 refers to the squared linear regression coefficient of each input parameter.

The squared SRCs S_i^2 take values between 0 and 1, and their sum represents the coefficient of determination R^2 which is 1 for an approximately linear model (Saltelli et al., 2008). In this study, the model refers to the algorithms that were used to calculate the N use indicators: Life-cycle-NUE_N, Life-cycle-NNB_N, and NHI_N (Uwizeye et al., 2016a). Since the sum of the squared SRC are close to 1, we assumed that the model to calculate each of these indicators behaves linearly. The SRC method gives insight into the 1st order effects and does not give any information about the interaction among input parameters (Cosenza et al., 2013). We use sensitivity packages of R (Pujol et al., 2015) to estimate the squared SRC through a bootstrap technique (Canty and Ripley, 2014). The bootstrap technique allows to improve the accuracy of the squared SRC estimates. The number of model evaluations was set to r = 1000 based on Saltelli et al. (2008).

Identification of the important input parameters

The squared SRCs were calculated to explain the variance of Life-cycle-NUE_N, Life-cycle-NNB_N, and NHI_N. Based on squared SRCs, we ranked the input parameters in two categories based on the threshold value of 0.01 (Cosenza et al., 2013; Sin et al., 2011). The non-important input parameters were identified with a squared SRC lower than 0.01, whereas, the important input parameters had squared SRC greater than 0.01.

Discernibility analysis

The discernibility analysis based on Heijungs and Kleijn (2001) was performed to evaluate if the indicators computed with the global dataset are similar to those obtained from field survey datasets. The discernibility analysis is an approach to assess the similarities between pairwise Monte Carlo simulations or sample of model outputs (Yao and Zhao, 2007). It allows counting the number of occurrences where the difference computed as introduced in Eq. 4.3 is positive:

$$(n^i_{1>2})_z = ((\mathbf{Y}^i_1)_z - (\mathbf{Y}^i_2)_z) \tag{4.3}$$

$$n_{1>2} = \sum_{i=1}^{N} \frac{(n^i_{1>2})_j.100}{N} \tag{4.4}$$

where Y^i_1 is a vector of the model output for dataset 1; Y^i_2 is a vector of the model output for dataset 2, N refers to the size of the Monte Carlo samples, and z refers to an occurrence of positive difference. The percentages show how often the N use indicators of the global dataset are higher than farm survey. When $\alpha-$value of 0.05 is applied, values between 2.5% and 97.5% indicate that the N use indicators are not significantly different. For example, if Life-cycle-NUE$_N$ computed from a global dataset is lower than that from field survey in 4000 out of 5000 runs, the discernibility analysis equals to 20%, which means that the two Life-cycle-NUE$_N$ are not significantly different.

Local relevance of environmental indicators computed from a global dataset

Following the GSA, we improved the N use indicators results from a global dataset, and simplified the model by fixing the distribution of the non-important input parameters using their reference or average values within their ranges. This step does not significantly change the variance of the results (Pianosi et al., 2016; Saltelli et al., 2008; Sin et al., 2011). Then, we substituted the distribution of the important parameters in a global dataset with corresponding distributions from the farm survey datasets as suggested by Heijungs (1996) and Uwizeye et al. (2016b). The distributions of farm survey data were selected based on PDF that gives the best goodness-of-fit. We assumed that farm survey data accurately represent reality and that their variability reflected the heterogeneity of dairy systems. We compared new computed N use indicators to those estimated from field survey to determine the potential improvement.

4.2.2 Case studies

We applied the GSA to the framework describing N use performance in mixed dairy systems from cradle to the primary processing stage, in Rwanda and The Netherlands.

Global dataset: GLEAM

Data on mixed dairy systems for Rwanda noted as GLEAM$_{RW}$ and the Netherlands noted as GLEAM$_{NL}$, were extracted from the second version of the GLEAM with the reference year of 2010 (FAO, 2017). The dataset contains detailed information on synthetic fertilizer application, crop residues, manure applied, manure recycled, feed resources, feed rations, animal number, herd structure, MMS, emissions factors and slaughterhouse activities. These data are based on national statistics reports and literature. For Rwanda, regional data are used where country-specific data are missing e.g. leaching rate of N during MMS or yield of grain. Most of the GLEAM data are based on reference or average values, with no information about their coefficients of variation. A detailed description of the GLEAM data is provided in the supplemental information (SM01).

In total, for GLEAM$_{RW}$, 121 input parameters were included in the sensitivity analysis, whereas 98 input parameters were included for GLEAM$_{NL}$.

Table 4.1: List of all 26 important input parameters identified for N use indicators computed from a global model of GLEAM for mixed dairy systems in Rwanda and the Netherlands.

No	Symbol	Description
1	P_{DC}	Carcass dressing
2	P_{EFLMF}	Emission factor of indirect N emissions from liquid manure for cows in manure management system
3	P_{EFMC}	Emission factor of indirect N volatilization from applied manure
4	P_{EFSYF}	Emission factor of indirect N emissions from applied synthetic fertilizer
5	P_{EFSM}	Emission factor of indirect N emissions from solid manure in manure management system
6	P_{EFSMF}	Emission factor of indirect N emissions from solid manure for cows in manure management system
7	P_{GT}	Grazing time
8	P_{LLM}	Leaching rate from liquid manure
9	P_{MG}	Applied manure to pasture
10	P_{MMC}	Mineralization rate of organic fertilizer applied to crop
11	P_{MP}	Milk yield
12	P_{NCFG}	N content of the fresh grass
13	P_{NCGR}	N content grain
14	P_{NCH}	N content of hay
15	P_{NCMG}	N content of maize gluten
16	P_{PCM}	Protein content of milk
17	P_{SYFC}	Applied synthetic fertilizer to the cropland
19	P_{SYFG}	Applied synthetic fertilizer to the grassland
20	P_{SMG}	Rate of maize gluten in the total feed ration
21	P_{SGR}	Rate of grain in the total feed ration
22	P_{SH}	Rate of hay in the total feed ration
23	P_{SST}	Stall time
24	P_{YFG}	Yield of the fresh grass
25	P_{YRS}	Yield of rapeseed
26	P_{YSB}	Yield of soybean

Data from farm surveys

Mixed intensive dairy system in Rwanda

The Rwandan mixed dairy system that was the subject of this study is relatively intensive compared to average dairy production in the country and is characterized by the purchase of feed, high stock density, and high milk production per hectare. The system is composed of three main processes: feed production, animal production, and processing of milk and meat. Feed production includes on-farm grasses and legumes and off-farm production of concentrates, including soybean meal, maize bran and cotton meal. Animal production includes dairy cows, young stocks, and replacement animals. Manure effluents are collected and stored in lagoons or dried after being mixed with litter. Manure is then applied to the on-farm cropland or sold to other crop farmers. Activity data were collected through direct interviews and consultation of bookkeeping from 15 dairy farms located in the peri-urban area of Kigali city. Additional data related to EF and other coefficients were obtained from secondary resources: EFs (IPCC, 2006), N content of crops (Feedpedia, 2012), runoff and leaching coefficients (Gerber et al., 2013). A detailed description of the data for the mixed intensive dairy system in Rwanda is provided in the supplemental information (SM01). Furthermore, the N use indicators computed from substituted GLEAM dataset are noted as $GLEAM_{RW}*$.

Mixed intensive dairy system in the Netherlands

The Dutch mixed dairy system is characterized mainly by the reliance on highly managed pastures, maize silage, and imported feed; a high stock density, and high milk production per hectare. Data for 249 dairy farms were derived from the Dutch farm accountancy data network (FADN) for 2010, noted as $FADN_{NL}$ and were used in GSA. $FADN_{NL}$ provides detailed data on on-farm feed production including fertilizer application, manure application, nutrient, and energy content, and purchased concentrates (Dolman et al., 2014; Thomassen et al., 2009). Purchased feeds are aggregated into three main types of compounded feed (singular dry concentrates, wet-by products, and milk replacers). A standard protein ration composition was used to determine the feed composition of the compound feed (Middelaar et al., 2013). Data related to their cultivation in their country of origin are taken from Middelaar et al. (2013) and Vellinga et al. (2013). The processing stage was not covered by $FADN_{NL}$. Data related to milk processing and slaughterhouse processes, therefore, were obtained from Uwizeye et al. (2016b). A detailed description of the data for the mixed intensive dairy system in the Netherlands is provided in the supplemental information (SM01). Furthermore, the N use indicators computed from substituted GLEAM dataset are noted as $GLEAM_{NL}*$.

4.3 Results and discussion

4.3.1 Uncertainties of N use indicators

The results of the uncertainty analysis for N use indicators in the Netherlands are shown in Figure 4.2. The mean value of Life-cycle-NUE_N for $FADN_{NL}$ was similar to $GLEAM_{NL}$ (46%). For other indicators, however, there was a larger difference in mean values. The mean value of Life-cycle-NNB_N was lower for $FADN_{NL}$ (105 kg N ha^{-1}) as compared to $GLEAM_{NL}$ (132 kg N ha^{-1}). Regarding the NHI_N, the mean value was higher for $FADN_{NL}$ (158%) as compared $GLEAM_{NL}$ (147%).

The distribution of NHI_N was similarly leptokurtic with a bottom-skew for both $FADN_{NL}$ and $GLEAM_{NL}$. The large skewness of $FADN_{NL}$ suggested a large heterogeneity within the dairy system suggesting that less performing dairy farms are largely distant to the average farms. This heterogeneity was partially captured by the CV of 20% assumed for GLEAM dataset.

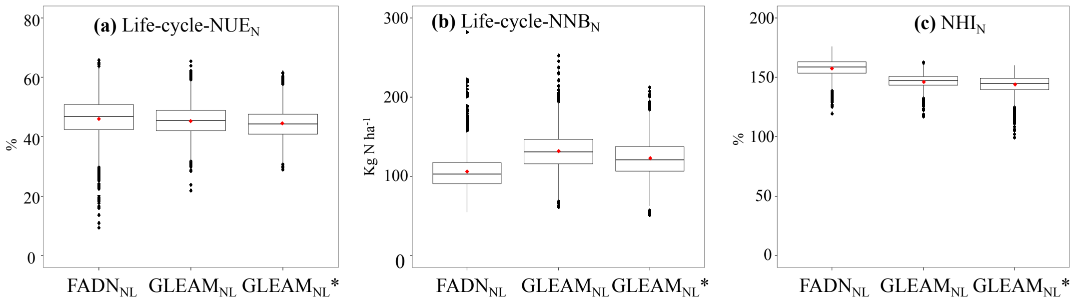


Figure 4.2: Box plots of the distributions of the nitrogen use indicators: (a) life-cycle nitrogen use efficiency, (b) life-cycle net nitrogen balance, and (c) nitrogen hotspot index, for the mixed dairy systems in the Netherlands computed from farm survey data (FADN$_{NL}$), global dataset (GLEAM$_{NL}$) and improved global dataset with high-quality data from survey (GLEAM$_{NL}$*). The red points indicate the mean values.

Oenema et al. (2015) found substantial heterogeneities of N flows in Dutch dairy farms related to differences in farm structure, farm characteristics, and production intensity. Mu et al. (2017) similarly determined large heterogeneities for N flows among European dairy farms including Dutch farms. Despite these differences, the discernibility analysis showed no significant difference between N use indicators computed for FADN$_{NL}$ and GLEAM$_{NL}$ as shown in Table 4.2. The reason is that the GLEAM dataset provided reliable and representative estimates of mean value and variability of N use indicators when the uncertainties of the input parameters are considered. The reason is that GLEAM data for the Netherlands were collected from abundant and reliable statistics and literature.

The results of N use indicators for mixed dairy systems in Rwanda are shown in Figure 4.3. The observed mean value of Life-cycle-NUE$_N$ for FIELD$_{RW}$ was slightly higher (33%) as compared to GLEAM$_{RW}$ (32%). For NHI$_N$, the mean value for FIELD$_{RW}$ was lower (134%) as compared to GLEAM$_{RW}$ (155%). The largest difference was found for the Life-cycle-NNB$_N$ where the mean value for FIELD$_{RW}$ differed greatly about twofold (65 kg N ha^{-1}) as compared to GLEAM$_{RW}$ (31 kg N ha^{-1}). This difference was found to be significant by the discernibility analysis (Table 4.2). The reason is that data in GLEAM$_{RW}$ strongly differed from field survey data for some input parameters. The milk yield given in the GLEAM dataset, for example, is 504 kg year^{-1}, whereas the average from FIELD$_{RW}$ was 4,879 kg year^{-1}; similarly, the synthetic fertilizer applied to cropland was assumed to 0 in GLEAM$_{RW}$, whereas the average value was 51 kg N ha^{-1} in FIELD$_{RW}$. These differences may be related to the fact that GLEAM data are collected from available sources, that might occasionally be outdated or only existing at a regional level.

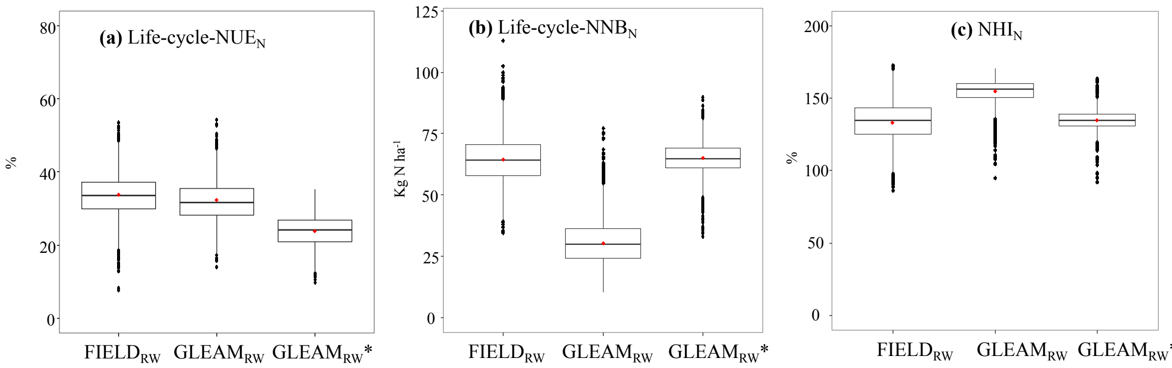


Figure 4.3: Box plots of the distributions of the nitrogen use indicators: (a) life-cycle nitrogen use efficiency, (b) life-cycle net nitrogen balance, and (c) nitrogen hotspot index, for the mixed dairy systems in the Rwanda computed from farm survey data (FIELD$_{RW}$), global dataset (GLEAM$_{RW}$) and improved global dataset with high-quality data from survey (GLEAM$_{RW}$*). The red points indicate the mean values.

4.3.2 Identification of the important input parameters

We classified the input parameters of both GLEAM$_{NL}$ and GLEAM$_{RW}$ subjected to GSA as important and non-important. The list of important input parameters identified is provided in Table 4.1. This is different from Huang et al. (2015) who classified them into more detailed categories, namely: very important, important, slightly important, and unimportant. For the Netherlands, Figure 4.4 summarizes the results of the squared SRCs of the important input parameters that contributed to at least one of the variances of the three N use indicators computed from a global dataset (GLEAM$_{NL}$).

Out of 98 input parameters, only 23 parameters were identified as important. The most important ones for Life-cycle-NUE$_N$ were EF of indirect N volatilization from applied manure to cropland, which contributed 24% to the variance of Life-cycle-NUE$_N$, 11% to the variance of Life-cycle-NNB$_N$, and 7% to the variance of NHI$_N$. The main reason is that this input parameter is highly uncertain and can vary up to a factor of 10, from 0.05 to 0.5. The applied manure to grassland was the important source of uncertainties and contributed 11% to the variance of Life-cycle-NUE$_N$ and 19% to the variance of Life-cycle-NNB$_N$, but with no effect on NHI$_N$. The applied synthetic fertilizer to the cropland was also most important and contributed 10% to the variance of Life-cycle-NUE$_N$, 35% to the variance of Life-cycle-NNB$_N$, and 24% to the variance of NHI$_N$.

For Rwanda, the results from the sensitivity analysis were shown in Figure 4.5 for the GLEAM dataset (GLEAM$_{RW}$). Out of 121 input parameters, only 9 parameters were identified as important. The applied manure to grassland was the most important source of uncertainties and contributed 30% to the variance of Life-cycle-NUE$_N$, 46% to the variance of Life-cycle-NNB$_N$ and 34% to the variance of NHI$_N$. N content of hay was also the important source of un-

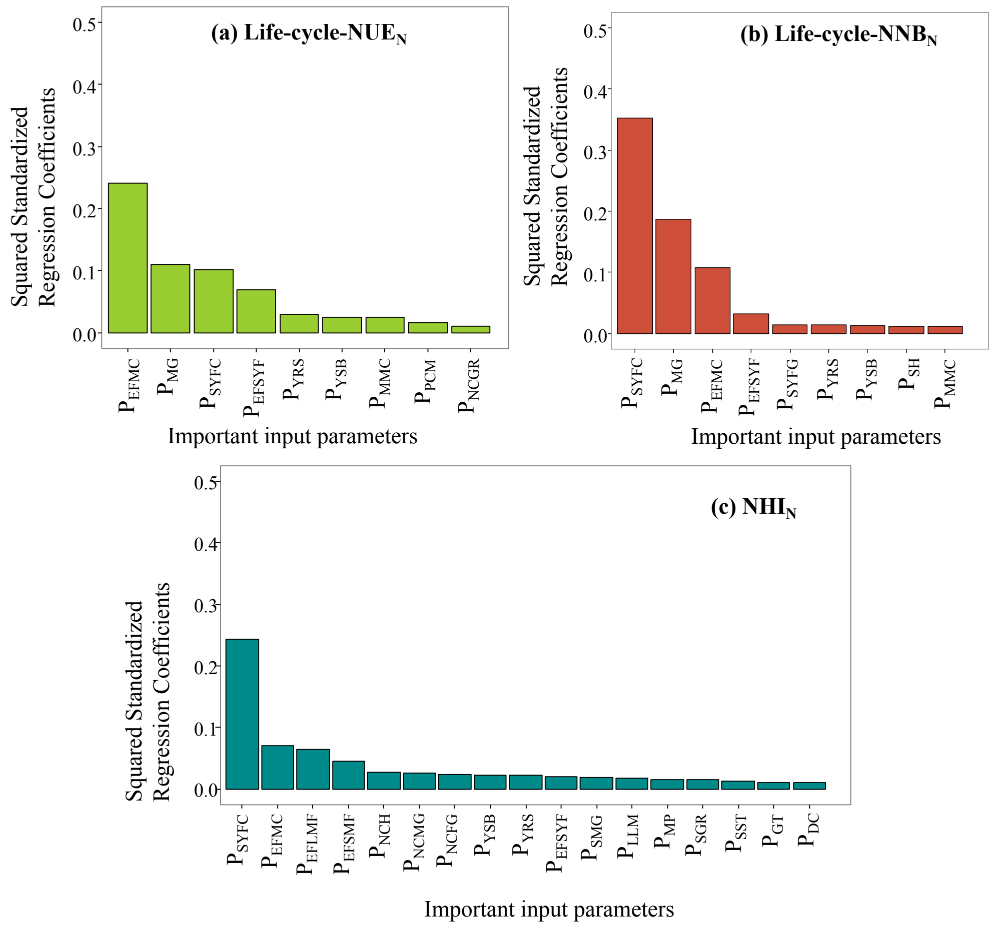


Figure 4.4: Important input parameters for mixed dairy systems in the Netherlands, ranked by squared standardized regression coefficients S_i^2, for (a) life-cycle nitrogen use efficiency, (b) life-cycle net nitrogen balance, and (c) nitrogen hotspot index. Computation based on GLEAM dataset (CV = 20%) (GLEAM$_{NL}$). The list of the acronym of the important parameters is provided in Table 4.1.

certainties and contributed 14% to the variance of Life-cycle-NUE$_N$, 11% to the variance of Life-cycle-NNB$_N$, and 14% to the variance of NHI$_N$. EF of indirect N volatilization from the applied manure contributed 14% to the variance of Life-cycle-NUE$_N$ but had less effect on other indicators.

Our results can be compared with studies in life cycle assessment (LCA) of dairy supply chains. Wolf et al. (2017) identified milk production, feed intake, N_2O and CH_4 emission factors as the most important parameters. Godinot et al. (2014) used a variance decomposition method for GSA for N use efficiency model at a farm level and identified high-protein crop output, change in soil N stock, synthetic fertilizer, milk production and quantity of applied manure as

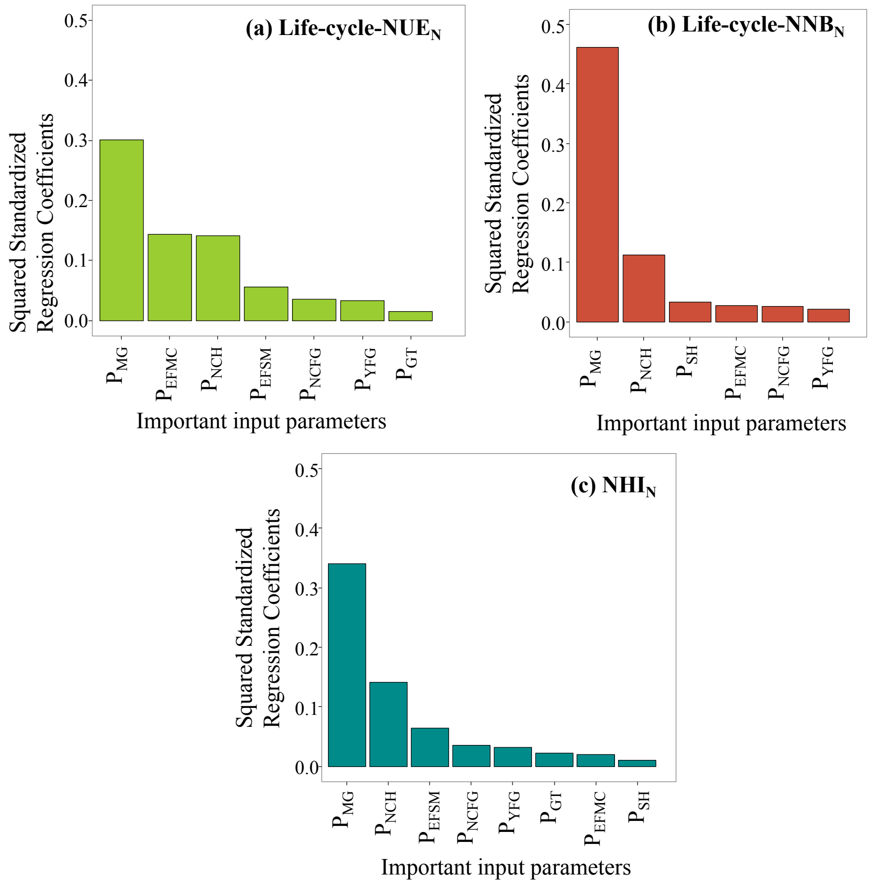


Figure 4.5: Important input parameters for mixed dairy systems in Rwanda, ranked by squared standardized regression coefficients S_i^2, for (a) life-cycle nitrogen use efficiency, (b) life-cycle net nitrogen balance, and (c) nitrogen hotspot index. Computation based on GLEAM dataset (CV = 20%) (GLEAM$_{RW}$). The list of the acronym of the important parameters is provided in Table 4.1.

important parameters. Despite differences between models and methods used in these studies and this study, we identified milk production, EF and applied synthetic fertilizer as important input parameters. It is surprising that some input parameters that were important for mixed dairy systems in The Netherlands were not important for Rwanda such as the milk yield. The reason may be related to the fact that other input parameters may have larger uncertainties, and therefore decreased the effect of this parameters during the sensitivity analysis.

4.3.3 Effectivity of the sensitivity analysis results

The sum of squared SRC (R^2) was greater than 0.77 for GLEAM$_{NL}$ (Life-cycle-NUE$_N$=0.78, Life-cycle-NNB$_N$=0.81, and NHI$_N$=0.85); whereas it was greater than 0.70 of GLEAM$_{RW}$ (Life-cycle-NUE$_N$=0.78, Life-cycle-NNB$_N$=0.76, and NHI$_N$=0.74). These values of R^2 are in

Table 4.2: Results of discernibility analysis for mixed dairy systems in Rwanda and the Netherlands based on pairwise comparing Monte Carlo simulations obtained from GLEAM dataset and farm survey data. The percentages show how often the N use performance indicators of the GLEAM dataset are higher than farm survey. When $\alpha-$value of 0.05 is applied, values between 2.5% and 97.5% indicate that the N use indicators are not different. The significant differences are indicated in the bold-printed percentages.

		Life-cycle-NUE$_N$	**Life-cycle-NNB$_N$**	**NHI$_N$**
		FADN$_{NL}$		
	Life-cycle-NUE$_N$	54%		
GLEAM$_{NL}$	**Life-cycle-NNB$_N$**		19%	
	NHI$_N$			88%
	Life-cycle-NUE$_N$	61%		
GLEAM$_{NL}$*	**Life-cycle-NNB$_N$**		28%	
	NHI$_N$			91%
		FIELD$_{RW}$		
	Life-cycle-NUE$_N$	58%		
GLEAM$_{RW}$	**Life-cycle-NNB$_N$**		**99%**	
	NHI$_N$			9%
	Life-cycle-NUE$_N$	91%		
GLEAM$_{RW}$*	**Life-cycle-NNB$_N$**		47%	
	NHI$_N$			49%

the application range of the SRC method (Saltelli et al., 2008). Our results, thus, are robust and confirm that the framework model to assess the N use indicators in the livestock supply chains is approximately linear.

Building on global datasets to assess nutrient use indicators at local level

After the substitution of the important parameters in GLEAM dataset by the field survey data, the results of N use indicators changed as follows. For the Netherlands, the mean value decreased for Life-cycle-NNB$_N$ (from +25% to +16% closer to FADN$_{NL}$ value) and was slightly decreased for both Life-cycle-NUE$_N$ (from 0% to -4% closer to FADN$_{NL}$ value) and NHI$_N$ (from -6% to -9% closer to FADN$_{NL}$ value), as shown in Figure 4.2. The discernibility test was not different as shown in Table 4.2. The substitution of the important parameters did reproduce the lower bound tail of Life-cycle-NUE$_N$ and the upper bound tail of both Life-cycle-NNB$_N$ and NHI$_N$.

For Rwanda, results are shown in Figure 4.3. The mean value of GLEAM$_{RW}$* was substantially improved for Life-cycle-NNB$_N$ (from -52% to a similar value of FIELD$_{RW}$) and NHI$_N$ (from +16% to +1% closer to FIELD$_{RW}$ value) but not for Life-cycle-NUE$_N$ which decreased after substitution (from -3% to -27% closer to FIELD$_{RW}$ value). The discernibility analysis confirmed the improvement of the results from GLEAM dataset, after the substitution of the important parameters, with no significant difference between GLEAM$_{RW}$* and FIELD$_{RW}$ as shown in Table 4.2. The ranges of the distributions of the three indicators were also improved and were within the ranges of FIELD$_{RW}$.

The two case studies showed that the proposed method allows improving the local relevance of the N use indicators computed from a global dataset. This method saves considerable data collection effort compared to the findings of Elduque et al. (2015). These authors recommended the direct measurement of all input parameters to reduce the discrepancies and uncertainties of environmental indicators related to moulding process for plastics initially computed from a global dataset of the ecoinvent (Frischknecht et al., 2005).

Reliance on this method allows to systematically improve the important input parameters, irrespectively of methodological choices and expert knowledge used in the model. The expert's knowledge, for example, has been explored by Krueger et al. (2012). The authors proposed an approach to reducing the bias which includes the expert calibration, where experts are asked to judge the same variables in two or different ways. Furthermore, the proposed method also allows to reflect the heterogeneity within the system and thus improve the decision-making relevance of the environmental indicators at a country level.

Applicability of this method

The method proposed in this study showed that by targeting a few input parameters during the data collection phase, the reliability of results of environmental studies could be effectively improved at a local level. Data collection efforts and cost, therefore, can be substantially reduced. For example, the important parameters represent about 23% of all input parameters considered in the Netherlands and 7% in the case of Rwanda. This method may be used in any other study on environmental modelling for livestock systems or other agricultural systems to improve the local relevance of the environmental indicators computed from a global dataset such as GLEAM. By carrying out this analysis before the data collection, using globally available data, any environmental study can cut on the cost of data collection by focusing on important input parameters that can be enhanced through good practices in data collection. Moreover, performing GSA allows acquiring knowledge on the variability of the system under study. It can facilitate the design of environmental policies and robust mitigation options through the establishment of targeted incentives to the farmers based on their environmental performance (Henderson et al., 2017), which is not informed when average values are only used.

Limitation of this method

While this study demonstrated that the global sensitivity analysis is an important method for input parameters prioritization during the data collection phase, the SRC method used has some shortfalls. This method may under/overestimate the sensitivity indices in case of correlated input parameters (Groen and Heijungs, 2017). In this study, we assumed that input parameters were independent and uncorrelated, and the model was linear. Before applying this method to any other study, however, it would be important to verify correlations among input parameters (Groen and Heijungs, 2017) and to use a more appropriate GSA method i.e. Sobol in the case of a non-linear model (Groen et al., 2014a; Sobol, 2001). Moreover, due to the assumption of a standard CV for all activity data in the global dataset, it is likely that not all important input parameters are identified. This assumption is a compromise because the distribution of most of the global data is unknown. Despite this limitation, the method proposed is robust to improve the local relevance of the environmental indicators computed from a global dataset. Furthermore, it is important to verify the quality of the global dataset before the application of this method, because if the quality of data is poor, it is likely that the identification of important input parameters would be erroneous leading to incorrect results. The data quality indicators are described in several guidelines such as UNEP SETAC life cycle initiative (Sonnemann and Vigon, 2011), International Reference Life Cycle Data System (ILCD) Handbook (European Commission, 2010) or Livestock Environmental Assessment Performance (LEAP) Partnership guidelines (FAO, 2016). Furthermore, the proposed method demonstrated that EFs are important input parameters. Thus, the improvement of their estimates would reduce the uncertainties and improve the reliability of N use indicators significantly. Several studies on LCA of livestock systems had also identified the importance of EFs in the quantification of greenhouse gas emissions (Groen et al., 2016; Wolf et al., 2017; Zhu et al., 2016). Attempts should thus be made to estimate locally-specific EFs, based measurements or mathematical models that take into consideration environmental conditions, such as climate, temperature, and management. Since IPCC guidelines (IPCC, 2006), efforts have been made to quantify locally-specific EFs e.g. Borhan et al. (2011) and Redding et al. (2015), but these data are not available for all global regions and production systems. Because in most of the assessment, EF default values are used, we recommend that the choice of these values should be documented and supported with a global sensitivity analysis for better interpretation of results.

4.4 Conclusions

This study proposed a method to improve the local relevance of the environmental performance indicators computed from a global dataset, by identifying important input parameters through a GSA that shall be prioritized and established with high-quality data. We demonstrated that N use performance indicators computed from GLEAM dataset were relatively close to those estimated from farm survey data in the Netherlands than in Rwanda. However, by substituting the important input parameters for activity data with high-quality data from farm survey, the average values of N use performance indicators were improved for both the Netherlands and Rwanda. The use of a standard coefficient of variation also allowed to represent farm heterogeneity but failed to capture the skewed nature of the distribution.

By applying this method, any environmental modelling assessment using globally available datasets can improve the local relevance of the assessment by focusing on important input parameters for additional detailed data collection. However, the quality of the global data should be checked for the reliability because this method depends on it. Further work on the assessment of nutrient use indicators in the global livestock supply chains will benefit from this method to generate analysis that is locally relevant and at lesser data collection cost.

4.5 Acknowledgements

This work is supported by the TeagascWalsh Fellowship Scheme (Ref: 2012230), the Livestock Environmental Assessment Performance (LEAP) Partnership (GCP/GLO/369/MUL) and the Livestock Information, Sector Analysis and Policy Branch (AGAL) of Food and Agriculture Organization of United Nations (FAO). We would like to thank Bas Angel for his statistical advice and the GLEAM development team for their support in data handling.

Chapter 5

Nitrogen flows in global pork supply chains and potential improvement from feeding swill to pigs

Aimable Uwizeye[1,2,3], Pierre J. Gerber[1,2,4], Carolyn I. Opio[1], Giuseppe Tempio[1], Anne Mottet[1], Harinder P.S. Makkar[1], Alessandra Falcucci[1], Henning Steinfeld[1], Imke J.M. de Boer[2]

[1]Food and Agriculture Organization of the United Nations, Animal Production and Health Division, Viale delle Terme di Caracalla, 00153 Rome, Italy
[2]Animal Production Systems group, Wageningen University & Research, PO Box 338, 6700 AH, Wageningen, the Netherlands
[3]Teagasc – Crops, Environment and Land Use Programme, Johnstown Castle, Wexford, Ireland
[4]The World Bank Group, 1818 H Street NW, Washington DC, 20433 USA

This chapter is published in **Resources, Conservation and Recycling, 146, 168-179** with slight modifications.

Abstract

The global pork sector contributes to food security and supports livelihoods for millions of households but also causes nitrogen (N) pollution. Here we assess N flows, losses, and N use indicators for global pork supply chains, from “cradle-to-primary-processing-gate” and for three production systems: the backyard, intermediate and industrial systems. Subsequently, we evaluate the effects of feeding swill to industrial pigs on N flows and land use. To produce 3.5 Tg N of pork globally, 14.7 Tg N are lost into the environment, of which 68% is lost to watercourses in the form of nitrates and organic N and the reminder emitted to the atmosphere as N-gas (e.g., NH_3, NO_x and N_2O). We found that the efficiency of N use, hotspot and magnitude of N losses per unit of area depend on the region, production system, origin of feed, and manure management systems. Swill feeding increases N use efficiency and reduces N losses at the feed production stage. It achieves a saving of 31 Mt of soybeans and 20 Mt of grains on dry matter basis, equivalent to 16 Mha of land required. Its adoption would require innovative policies to preserve food safety and public health. Future research may explore the feasibility and requirements to adopt swill feeding at a country level and may investigate potential impacts on other sustainability objectives.

5.1 Introduction

The global pork sector contributed around 38% of the global meat supply in 2014 (OECD/FAO, 2015), with a production volume that has been steadily growing (3% per year) over the past four decades (FAOSTAT, 2018). As it grows, the sector is evolving from local, and horizontally integrated production units to globalised, confined and vertically integrated supply chains, which rely on international trade for both inputs (e.g., high-quality feed) and outputs (e.g., pork products) (Gerber et al., 2010). This trend involves automation, specialisation, and reduction of transport costs. Such "industrialised" systems are responsible for about 56% of the global pork supply (MacLeod et al., 2013). The livelihoods of millions of producers and consumers benefited from such growth (FAO, 2011). But, negative environmental externalities have occurred on a large scale, including nitrogen (N) losses and their associated environmental impacts such as climate change, acidification, and eutrophication (Hamilton et al., 2018; Poore and Nemecek, 2018). In recent decades, N losses from anthropogenic activities have increased to a level that has surpassed the planetary boundaries (Steffen et al., 2015), and are expected to rise further (Bodirsky et al., 2014). Pig systems contribute to such N losses, via gaseous emissions and loads of nitrates (NO_3^-) and organic N to ground and surface water (Galloway et al., 2003).

Several studies have examined N use in global pig systems, e.g. Lassaletta et al. (2016) or livestock systems, e.g. Liu et al. (2017) and pointed at the significant interactions between pig production and global N cycle. The majority of these studies did not stipulate the details of N dynamics in the pork supply chain and ignored losses associated with feed production happening outside the pig production unit. Excluding these issues lead to a partial assessment of N use and related N losses, potentially misguiding decision-making (Uwizeye et al., 2016b). Leip et al. (2014) assessed N flows along pork supply chains for the European Union, but no such study exists at the global level.

Research on reducing N losses from pork supply chains has focussed on farm-scale practices such as manure management (Oenema and Tamminga, 2005), feed production, and housing (Philippe et al., 2011). Reducing protein content of pig rations, for instance, was shown to have the potential to reduce NH_3 emissions by 50% (Philippe et al., 2011). Further reduction can be achieved through manure injection into soils during the field spreading (Bittman et al., 2014). For internationalised pig supply chains, where feed and animal production stages are disconnected, it is essential to explore novel improvement pathways in the food systems, e.g., by examining new recycling options.

Feeding food losses and wastes (FLW), known as swill, to pigs is practised in backyard and intermediate systems (Thieme and Makkar, 2017) but, it is not frequent in industrial pig systems, despite the wealth of swill available. It is estimated that about one-third of the food produced (around 1300 Tg y^{-1}) gets lost in the food production chain, (FAO, 2011b). Experiments have shown that the practice does not alter animal performance, regarding carcass characteristics and organoleptic quality (Westendorf et al., 1998). From a natural resource management perspective, feeding swill has the potential to reduce land and water use (van Zanten et al., 2018; zu Ermgassen et al., 2016) and contributes to the reduction of greenhouse gas (GHG) emissions (Brancoli et al., 2017; Papargyropoulou et al., 2014). In case studies modelled by Lassaletta et al. (2016), feeding swill has been found to increase N use efficiency (NUE_N) at herd level, when combined with lower levels of protein in the diets. Swill feeding can contribute to a circular bio-economy by recycling wasted nutrient back into the food system (HLPE, 2014; Kirchherr et al., 2017). Despite these promising results, no study has yet explored the effect of feeding swill to industrial pigs as a broad strategy to reduce N losses for global pork supply chains.

The objectives of this study are to assess N flows and N use indicators for global pork supply chains and to evaluate the effects associated with feeding swill to industrial pigs. N flows and N use indicators are quantified for 2010, based on the framework developed in Uwizeye et al. (2016a), by region and production system. The potential effect of feeding swill to industrial pigs is explored as a novel improvement option. The choice of industrial system is based on its importance in the global pork supply chains and its potential contribution to the sustainability of food systems.

5.2 Materials and Methods

5.2.1 Global pork supply chains

Pig production systems differ in the size of operations, housing, feed, energy use, and technology. The classification used in this paper is based on the Global Livestock Environmental Assessment Model (GLEAM), and organises pig production into three simplified systems: backyard, intermediate and industrial (MacLeod et al., 2013). The distribution of the pig population per region and production system in 2010 is provided in Figure 5.1.

Backyard systems consist of small-scale production units, in which pigs are reared for the local market or home consumption. Pigs are partially confined. The primary sources of feed include swill from households and restaurants, scavenging, other on-farm feed materials, such as second-grade grains (deemed unfit for human consumption), and local agro-industrial by-products, such as maize brans. In East and Southeast Asia (ESEA), pigs can be integrated with other animal species, such as fish. Feed and animal production are coupled, and manure is collected and recycled on cropland, grassland or applied to fish ponds. But manure can be illegally disposed into the environment in regions where production units are concentrated in densely populated areas. Pigs in backyard systems are concentrated in ESEA, predominantly in China (Figure 5.1).

Intermediate systems tend to emerge near cities, due to the increasing demand for pork products (Gerber et al., 2013). They supply pork for urban markets with medium capital investment and show an improved level of animal performance as compared to the backyard system. Pigs are confined, with solid floor and roof. Feed is partially sourced on-farm, although most of it is purchased locally or from international markets. Swill from restaurants and canteens can contribute a considerable part of the diet (Gerber et al., 2013). Animal performance is limited by a lower level of management and biosecurity, compared to industrial systems. Manure is partially recycled on cropland or grassland, and partially dumped into the environment as the peri-urban location of production units limits nearby recycling options, e.g., in China (Bai et al., 2014)

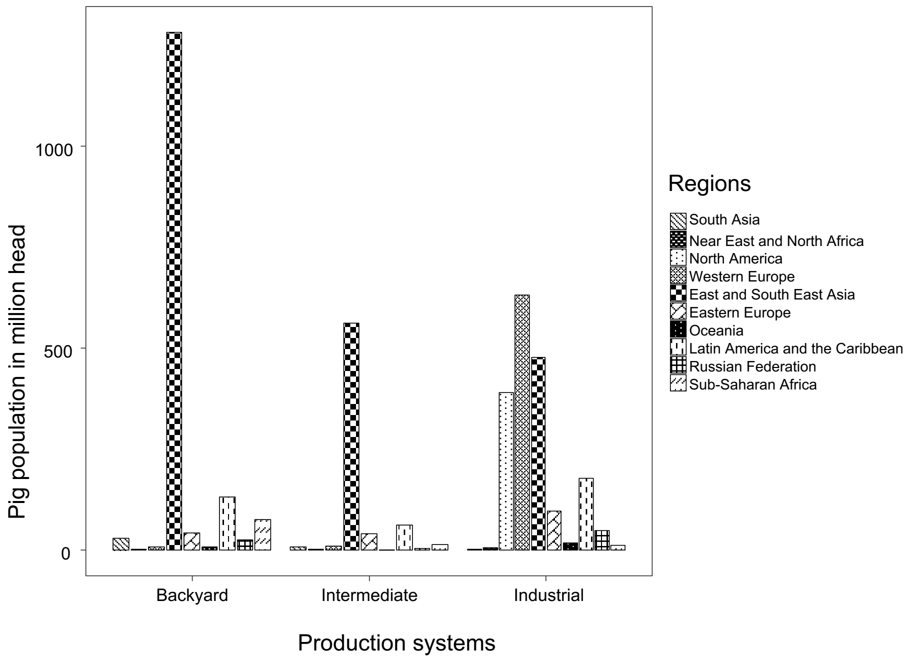


Figure 5.1: Global pig population by region and production system based on FAOSTAT (2018).

or Thailand (Schaffner et al., 2009). Intermediate systems are mostly found in ESEA (Figure 5.1).

In the industrial systems, production units supply pork to local and international markets and use high-quality feed, which can be produced on-farm, or processed from locally sourced or internationally-sourced feed materials. Pigs are confined in modern and automated barns with a high level of biosecurity, and feed conversion ratio is optimised. Manure management system is usually improved as companies invest in long-term sustainability and economies of scale apply to manure management. Industrial farms are often concentrated in areas, where operating costs are minimised, regardless of the increased transportation distances (Gerber et al., 2010). This geographical concentration, combined with the sheer size of production units and with the import of feed create an area of high nutrient losses (Costanza et al., 2008). High transport costs dissuade producers from moving manure to distant crop areas in need of fertilizer. As a result, over-application on land and dumping into watercourses become frequent where regulation on manure management is weakly enforced (Strokal et al., 2014). Industrial systems are mostly found in Western Europe (WE), ESEA, North America (NA) and Latin America and the Caribbean (LAC)(Figure 5.1).

5.2.2 Modelling framework

In this study, we consider the "*cradle-to-primary-processing-stage*" part of the life cycle of pork, around the year 2010. The framework developed by Uwizeye et al. (2016a) is used to assess N flows, losses, and N use indicators. This framework relies on a modular modelling approach, which follows the fate of N step-by-step through the different stages of pork supply chains: feed production, animal production, and processing of animal products. Figure 5.2 shows the overview of the N flows modelling used in this study. N emissions from energy production, use and transport are excluded from the assessment. Manure exported for application on non-feed crops or fishponds is considered to cross the system boundary. Any loss occurring from these activities is excluded from the analysis.

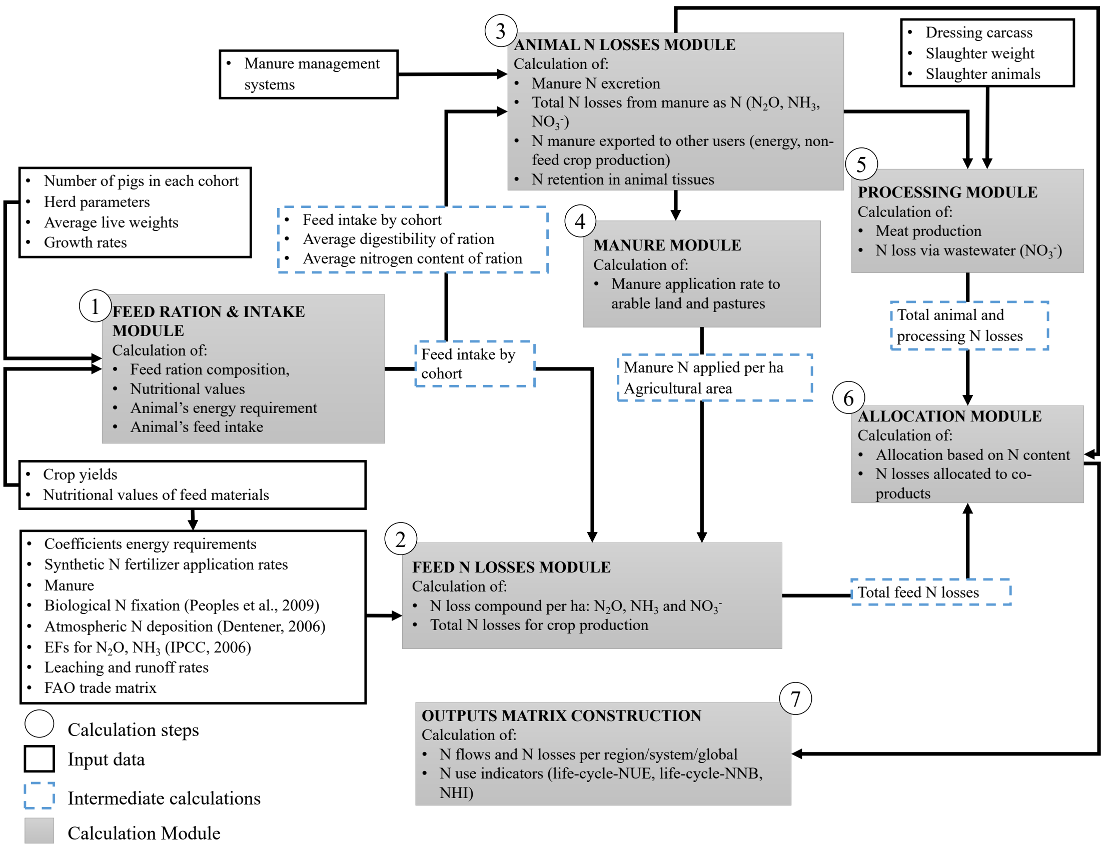


Figure 5.2: Overview of N flows modelling for global pork supply chains adapted from FAO (2017).

The framework draws on three indicators. Life-cycle nitrogen use efficiency (Life-cycle-NUE$_N$, %) refers to the efficiency with which N sources are mobilised from nature or other agricultural systems to generate the final animal products. It considers the linear and multidimensional processes of N flows such as feedback loops of recycling crop residues, manure as well as the change in soil nutrient stocks. Life-cycle net nitrogen balance (Life-cycle-NNB$_N$, kg N ha^{-1}) aggregates the total amount of N losses that are emitted into the environment at each supply chain stage, regardless of the actual areas where they occur in the chain. It is calculated by dividing total N losses with total land required to produce feed. Nitrogen hotspot index (NHI$_N$, %) characterises the evenness distribution of N balances along the supply chain and is calculated as a coeffecient of variation of N losses at each supply chain stage. These three indicators are calculated using the supply-and-use matrices, previously published in detail by Uwizeye et al. (2016a).

N flows and losses at feed production stage

Types of feed materials include feed crops, crop residues, agro-industrial by-products, swill and concentrates. They are divided into two categories: locally produced feed, and non-locally produced feed. Locally produced feed refers to crops, by-products, and swill produced on-farm or sourced from national production, whereas non-locally-produced feed refers to feed purchased from international markets. N flows associated with the production of each feed material are modelled, except for swill and synthetic amino acids. Swill is considered a residual with no allocation of upstream N losses. For synthetic amino acids, no sufficient data is available to estimate N losses associated with their production on a global scale. Total N loss is calculated as the sum of all N compounds released via volatilisation, runoff and leaching. Data for 245 countries and territories on synthetic fertilizer application rate, manure application rate, crop residues left on fields, crop yields, N content of feed materials, and emissions factors for N volatilisation, leaching/runoff rates, and composition of feed rations are obtained from GLEAM (FAO, 2018a). Country-specific data for biological N fixation are based on literature (Herridge et al., 2008; Peoples et al., 2009), and those on the atmospheric deposition are obtained from Dentener (2006). For non-locally-produced feed, average data are collected from the FAO trade matrix from 2008-2010 (FAOSTAT, 2018). The description of feed materials is available in SM, Table B.1, and the compositions of the rations and nutritional values are provided in supplementary information, Table B.2 for backyard, Table B.3 for intermediate and Table 5.1 for industrial systems.

N flows and losses at animal production stage

The pig herd is divided into five cohorts: sows, boars, sub-adult males and females for replacement and fattening pigs. Country-specific data on the number of animals and herd parameters, e.g. live weights, fertility, mortality, daily-weight gain for each system are obtained from GLEAM (FAO, 2018a). These herd parameters are used to calculate N intake, N retention, and N excretion. Excreted N is partitioned into different manure management systems and subsequently multiplied by specific emission factors obtained from IPCC (2006). Additional data are collected on manure dumped into the environment, for instance, in ESEA (Bai et al., 2014; Schaffner et al., 2009; Vu et al., 2007; Thien Thu et al., 2012). N losses are estimated as the sum of manure dumped N losses via leaching, and emission of N-gases (e.g., NH_3, NO_x, and N_2O) to the atmosphere.

N flows and losses at processing stage

Finished pigs are slaughtered for meat, edible offals, and non-edible co-products. N losses take place through wastage of organic solids, meat unsuited for human consumption, and wastewater. Data on post-harvest pork products, processes and flows are obtained from GLEAM (FAO, 2018a).

Table 5.1: Average feed ration composition for the industrial system between baseline and scenario expressed in percentage of each ingredient in the diet by region

Feed ingredients	South Asia		ESEA[1]		W. Europe[12]		E. Europe[11]		Russian Fed.[2]		N. America[10]		LAC[9]		Oceania		SSA[3]		NENA[4]	
	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)	Baseline (%)	Scenario (%)
Locally-produced feed ingredients (on-farm and by-products)																				
Swill	-	42	-	35	-	32	-	26	-	26	-	19	-	49	-	32	-	7	-	35
Others[5]	-	-	-		-		-		-	-	-	-	-	-	-	-	25	25	7	7
Non-local feed ingredients (off-farm and imported)																				
Pulses	-	-	-	-	3	3	-	-	-	-	1	1	-	-	-	-	-	-	-	-
Cassava	11	10	8	7	-	-	-	-	-	-	-	-	5	4	-	-	11	11	14	12
Wheat	-	-	1	1	21	16	29	23	34	27	10	9	23	14	22	17	-	-	-	-
Maize	20	13	30	22	20	14	26	21	20	16	44	38	22	13	-	-	19	18	22	15
Barley	-	-	2	1	16	12	9	7	10	8	28	24	-	-	16	13	-	-	-	-
Millet	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15	14	4	4
Rice	21	13	21	14	-	-	-	-	-	-	-	-	5	3	-	-	7	6	17	11
Sorghum	6	5	1	1	-	-	-	-	-	-	-	-	1	1	41	32	11	11	3	3
Soybean	3	1	2	0	0	0	-	-	-	-	-	-	10	2	-	-	-	-	3	1
Soybean meal	24	5	20	4	19	4	15	3	15	3	11	2	25	5	19	4	4	1	19	4
Oil seed meal	-	-	-	-	10	9	10	9	10	9	3	2	-	-	-	-	-	-	-	-
Cotton seed meal	3	2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0	0	-	-
Palm cake	-	-	3	2	0	0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Fishmeal	5	4	4	3	0	0	3	3	4	4	1	0	4	3	-	-	3	3	4	4
Molasses	-	-	2	2	4	3	-	-	-	-	-	-	0	0	-	-	4	4	1	1
DDG[6]	5	4	5	4	5	5	6	6	5	5	3	2	5	4	-	-	-	-	4	3
Supplement[7]	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	1	2	2
Nutritional values																				
GE[8] (MJ/kg DM)	18.8	18.4	18.7	18.4	18.8	18.6	18.9	18.7	18.9	18.7	18.8	18.7	19.4	18.6	18.9	18.6	18.3	18.3	18.7	18.4
N (g/kg DM)	37	32	32	29	33	30	33	31	34	32	24	23	39	33	26	24	20	20	32	29

[1]Eastern and Southeast Asia
[2]Russian Federation
[3]Sub-Saharan Africa
[4]Near East and North Africa
[5]Others: crop residues, grain, and by-products from grain industries: bran and middling
[6]Distillers dried grain
[7]Amino acids, vitamins
[8]Gross energy
[9]Latin America and the Caribbean
[10]North America
[11]Eastern Europe
[12]Western Europe

5.2.3 Scenario analysis

The effect of feeding swill to industrial pigs on N use indicators is explored by substituting swill for grains (e.g., barley) and soybean. These feed materials are selected for substitution because of the sustainability issues related to their production. They are edible by humans, and their use poses questions of land and water use for food production (van Zanten et al., 2018). Besides, soybean production is a driver of deforestation and land use change, thus associated with GHG emissions and biodiversity losses (Gibbs et al., 2015). The amount of swill available by country is estimated based on FAO (2011b), by multiplying regional FLW per capita with the human population in 2010 (FAOSTAT, 2018), assuming an average DM content of swill of 25% (zu Ermgassen et al., 2016). Food losses during harvesting are excluded because of their collection in fields is onerous. We assume that only 39% of swill available is used as animal feed, in line with the level of swill used in Japan (zu Ermgassen et al., 2016). In our scenario, swill is added in proportions that conserve the overall gross energy and N content of the feed, and to a maximum level of 50% of the rations (zu Ermgassen et al., 2016). For more details, see SM section S2.1. The feed compositions for the baseline and scenario are presented in Table 5.1.

5.2.4 Computation

The calculation model and programming code were performed using the R project (R Core Team, 2013). A descriptive statistical analysis and multivariate linear regression were carried out to compare pig systems. Detailed programming code and data are available upon request.

5.3 Results and discussion

5.3.1 N flows in the global pork supply chains

Overview

Globally, 51.9 Tg N are mobilised annually to sustain feed production for pigs, of which 4 Tg N are recycled from crop residues and manure (Figure 5.3). Pigs consume around 16.3 Tg N, of which 1.5 Tg N in the form of swill and 1.4 Tg N in the form of synthetic amino acids, to produce 3.5 Tg N of edible pork products (equivalent to 113.7 Tg in carcass-weight, in supplementary information Table B.5). Pork production generates co-products at different stages of the supply chains, including co-products from feed production, e.g., grains (14.5 Tg N), manure exported to non-feed croplands, grasslands or fishponds (6.9 Tg N) and inedible co-products from meat processing (0.04 Tg N).

The total N losses are estimated at 26.6 Tg N (sum of all N losses flows in Figure 5.2), , of which 14.7 Tg N are allocated to pork products, while the remainder is allocated to crop co-products and manure exported (Table 5.2). With a contribution of 76%, feed production is the primary contributor to total N losses. Losses from pig housing and manure management contribute 22%

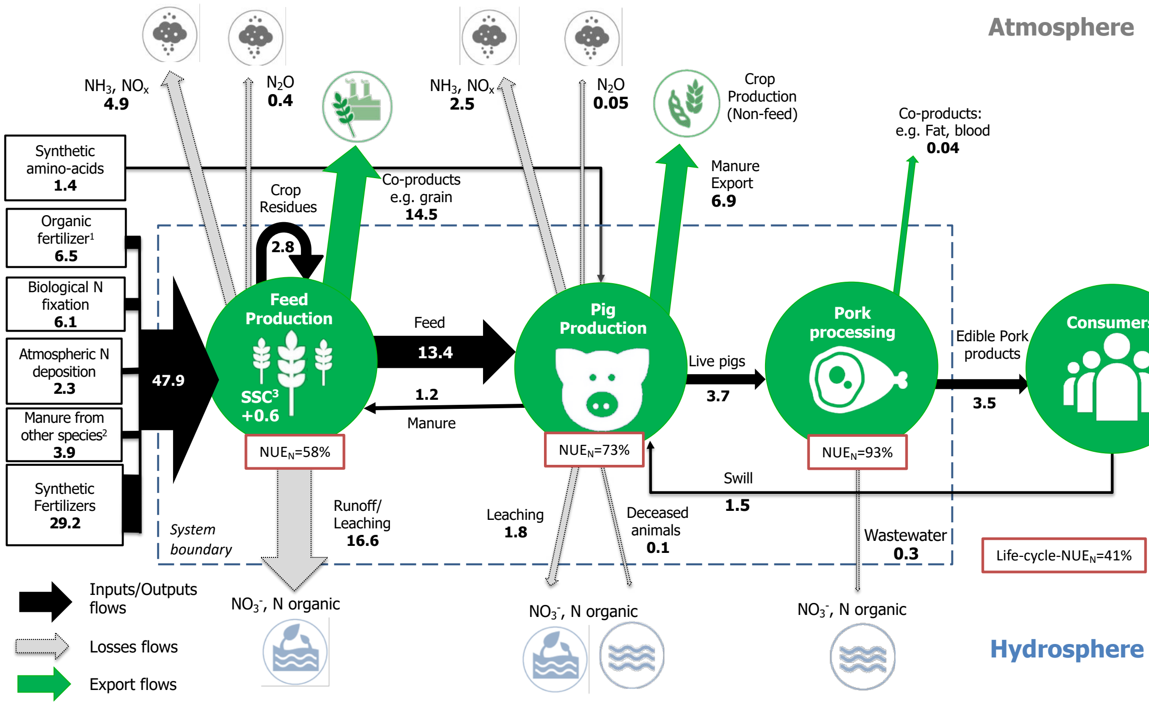


Figure 5.3: Aggregated global N flows associated with pork supply chains in Tg N y^{-1} for the current situation. N flows are the sum of fluxes for the three production systems: backyard, intermediate and industrial without allocation. [1] Organic fertilizer refers to the crop residues from the previous crop, in crop rotation or green manure. [2] Manure from other species, e.g., cattle or poultry. [3] SSC relates to soil N stock change.

to total N losses, whereas post-farm activities contribute only 2% (Table 5.2). About 58% of the total N losses take place in the backyard system (Table 5.2), but this system contributes only 27% to total pork production (in SM, Table B.5). The industrial system adds 23% to total N losses but generates about 56% to the global pork production. The intermediate system contributes equally to the total N losses and pork production, around 19%. About 88% of the total N losses take place in three regions ESEA, WE and Eastern Europe (EE), which account for 74% of total pork production (in SM, Table B.5). Considering the composition of N losses by compounds, about 68% of the total N losses are lost to watercourses in the form of NO_3^- and organic N, which cause freshwater and marine eutrophication. NH_3 and NO_x emissions explain 29% of total N losses and are sources of air pollution, human health damages, and acidification. N_2O emissions are much lower (2%, Table 5.3), but they significantly contribute to global warming. The backyard system provides 60% to total NO_3^- losses, whereas the industrial, like the intermediate system, contribute each around 20%. The backyard system is the primary sources of NH_3 and NO_x (51%) and N_2O (57%) emissions. The intermediate system contributes 17% to both NH_3, NO_x and N_2O emissions, whereas the industrial system contributes 32% to NH_3 and 26% to N_2O emissions. These compound losses take place primarily in feed production, ranging from 82-88% for NO_3^- losses, from 52-61% for NH_3 and NO_x, and from 83-90% for N_2O emissions. The backyard and intermediate systems have similar loss composition because of the similarity of their manure management systems, whereas manure is treated as a liquid or is stored in the lagoon for the industrial system, resulting in the high proportion of NH_3 emissions (38%).

These results differ from previous studies. For example, N intake estimated in this study is 30% higher than that reported by Lassaletta et al. (2016) for 2005: this is mostly related to an increase of pig population (15% between 2005 and 2010 (FAOSTAT, 2018) and differences in feed composition data. Global emissions of N_2O estimated in this study are two-fold higher than those modelled by Oenema et al. (2005) for 2000. This difference is partially explained by 25% of pig sector growth recorded between 2000 and 2010 (FAOSTAT, 2018), and more importantly by the differences in system boundaries. Oenema et al. (2005) focussed on animal production – including manure management – whereas this study included feed production and post-harvest activities.

Highly variable nitrogen losses at region and system level

Examining Table 5.2 reveals that N losses by the system and by region follow animal numbers (Figure 1). In most regions, backyard and intermediate systems contribute less to total N losses than they contribute to the entire pig population (ratio < 1, i.e., losses per head are lower than average in the region). This observation is not the case in for South Asia (SA) and ESEA. In contrast, industrial systems contribute more to N losses than they contribute to the pig population (ratio > 1) in most regions, except for NA, Oceania (OC), and LAC. These findings relate to differences in production practices, and especially in feed production, manure management, and animal productivity.

Examining Table 5.3 shows that regional N losses are dominated by NO_3^- emissions to ground and surface water, which are higher than gaseous N emission (e.g., NH_3, NO_x and N_2O), except in WE and Oceania for the backyard system. In these regions, NO_3^- losses are comparable to NH_3 and NO_x emissions. The reason is that NO_3^-leaching in feed production is low in these regions, because of the relatively high N uptake in crops. N losses from dead animals are minor for all regions. These results suggest that different regions experience contrasting issues for N management, implying a need for country-specific policy interventions. Such interventions would target mostly feed production stage focussing on best practices during the application of N inputs to cropland and collection and storage of manure.

The combined estimates of NH_3 and NO_x emissions for both South Asia (SA) and ESEA are different from those of Yamaji et al. (2004) for 2000 (2.9 Tg N versus 0.8 Tg N). The differences between these two studies are related to the growth of the pig sector and increased dependence on imported feed in 2010 (Gale et al., 2015). For example, SA and ESEA imports of soybean cake and maize have increased by 107% and 24%, respectively between 2000 and 2010 (FAOSTAT, 2018).

Table 5.2: Total N losses at each stage allocated to pork supply chains expressed in Gg y^{-1} by region for the baseline

Systems	Regions	N losses (Gg)				Ratio[1]	NHI_N[2] (%)
		Feed	Animal	Processing	Total		
BACKYARD	South Asia	134.9	16.1	1.5	152	1.01	144
	North America	-	-	-	-	-	
	Western Europe	9.7	3.5	0.5	14	0.32	103
	ESEA[3]	6,143.8	1,457.0	59.2	7,660	1.14	125
	Eastern Europe	138.4	21.4	2.3	162	0.73	136
	Oceania	4.4	3.4	0.5	8	0.16	74
	LAC[4]	138.3	87.9	6.8	233	0.34	85
	Russian Federation	58.6	12.5	1.5	73	0.53	125
	Sub-Saharan Africa	50.9	28.9	2.4	82	0.21	89
	NENA[5]	6.7	1.3	0.1	8	0.67	129
	Sub-Total	6,685.7	1,632.1	74.7	8,392.5		
	Share N losses	58%					
INTERMEDIATE	South Asia	50.9	4.8	0.6	56	1.66	149
	North America	-	-	-	-	-	
	Western Europe	13.3	5.3	0.6	19	0.49	100
	ESEA	1,824.0	549.1	35.5	2,409	1.10	115
	Eastern Europe	79.0	24.3	2.6	106	0.66	111
	Oceania	0.7	0.2	0.0	1	0.62	118
	LAC	58.9	43.2	4.0	106	0.43	80
	Russian Federation	5.8	2.6	0.2	9	0.60	98
	Sub-Saharan Africa	29.5	4.9	0.7	35	0.66	133
	NENA	3.8	1.0	0.1	5	0.75	119
	Sub-Total	2,065.9	635.3	44.3	2,745.5		
	Share N losses	19%					
INDUSTRIAL	South Asia	13.5	2.0	0.2	16	2.67	137
	North America	316.0	90.9	30.2	437	0.58	103
	Western Europe	1,107.1	341.2	48.4	1,497	1.22	110
	ESEA	542.3	363.9	36.4	943	1.02	82
	Eastern Europe	166.3	63.4	7.3	237	1.27	102
	Oceania	17.6	5.5	0.9	24	0.73	108
	LAC	127.9	99.6	13.7	241	0.70	74
	Russian Federation	87.3	34.7	3.7	126	1.33	101
	Sub-Saharan Africa	57.1	2.6	1.0	61	2.54	158
	NENA	12.5	4.5	0.5	18	1.28	105
	Sub-Total	2,447.6	1,008.2	142.3	3,598.1		
	Share N losses	23%					
All systems	World	11,199.2	3,275.6	261.3	14,736.0		
	Percentage	76%	22%	2%			

[1]Ratio refers to the relative contribution to N losses and is calculated by dividing the proportion of the regional N losses to total N losses and proportion of regional pig population to global pig population per system.
[2]NHI refers to Nitrogen hotspots index
[3]Eastern and Southeast Asia
[4]Latin America and the Caribbean
[5]Near East and North Africa

Table 5.3: Total N losses by N compounds allocated to global pork supply chains expressed in Gg N y^{-1} by region for the baseline

Regions	Backyard			Intermediate			Industrial			All systems		
	N_2O	NH_3[1]	NO_3^{-}[2]	N_2O	NH_3	NO_3^-	N_2O	NH_3	NO_3^-	N_2O	NH_3	NO_3^-
South Asia	2.5	37.5	112.1	0.9	12.0	43.4	0.3	3.6	11.8	3.6	53.2	167.3
North America	-	-	-	-	-	-	9.9	168.4	239.6	9.9	168.4	239.6
Western Europe	0.4	6.5	6.7	0.4	8.9	9.7	30.3	619.6	832.2	31.1	635.1	848.6
ESEA[3]	133	1,949	5,557	37.6	613	1,749	14	282	632	185	2,845	7,938
Eastern Europe	3.2	49.7	108.8	2.2	44.0	59.4	4.7	102.4	127.7	10.1	196.0	295.9
Oceania	0.2	4.1	4.0	0.0	0.3	0.5	0.4	7.7	15.3	0.6	12.1	19.8
LAC[4]	5.4	100.3	126.2	2.8	48.3	54.4	6.0	105.6	124.2	14.1	254.2	304.8
Russian Fed.[5]	1.6	25.8	45.0	0.2	4.0	4.4	2.1	53.4	69.1	4.0	83.2	118.6
SSA[6]	1.8	34.5	45.3	0.7	9.7	24.5	0.9	11.0	48.4	3.5	55.2	118.2
NENA[7]	0.2	2.8	5.1	0.1	1.7	3.0	0.4	5.9	11.1	0.7	10.4	19.2
World	149	2,210	6,010	45	742	1,949	69	1,360	2,111	262	4,313	10,070
Percentage	2%	26%	72%	2%	27%	71%	2%	38%	59%	2%	29%	68%

[1] NH_3 includes both NH_3 and NOx emissions
[2] NO_3^- includes Nitrates + organic N
[3] Eastern and Southeast Asia
[4] Latin America and the Caribbean
[5] Russian Federation
[6] Sub-Saharan Africa
[7] Near East and North Africa

Life cycle net nitrogen balance

Given the relevance of N losses for eutrophication and acidification, we compute N losses per unit of land area required in the supply chains to produce feed, expressed as Life-cycle-NNB$_N$ indicator. Global Life-cycle-NNB$_N$ is 64 kg N ha^{-1}. Figure 5.4 reveals the high heterogeneity among regions and within systems. For backyard systems, Life-cycle-NNB$_N$ ranges from 10 kg N ha^{-1} in Sub-Saharan Africa (SSA) to 104 kg N ha^{-1} in ESEA (global average value: 79 kg N ha^{-1}). For intermediate systems, Life-cycle-NNB$_N$ ranges from 18 kg N ha^{-1} in SSA and Near East and North Africa (NENA) to 79 kg N ha^{-1} in ESEA, (global average value: 66 kg N ha^{-1}), whereas, for industrial systems, Life-cycle-NNB$_N$ ranges from 30 kg N ha^{-1} in LAC to 64 kg N ha^{-1} in SA (global average value: 40 kg N ha^{-1}).

Except in ESEA, backyard and intermediate systems always have lower Life-cycle-NNB$_N$ than industrial systems, because of the connection between feed and animal production stages, giving more opportunities to recycle manure and crop residues. Moreover, both low fertilizer supply and low crop yields result in the use of more land to produce feed and in low N losses per unit of area. The higher Life-cycle-NNB$_N$ in industrial systems are related to high N application to feed-crops and limited availability of land to recycle manure due to the disconnection between supply chain stages. Production practices cause highest Life-cycle-NNB$_N$ in ESEA for backyard and intermediate systems. For example, the instauration of subsidies for synthetic fertilizer in China disincentivised the recycling and application of manure to cropland (Bai et al., 2018b; Chen et al., 2016). Consequently, farmers dumped more manure into water courses without adequate pre-treatment (Bai et al., 2018b). We observe that the spread of Life-cycle-NNB$_N$ values at the regional level is smaller for industrial systems than for other systems, because of the standardisation of production, including manure collection and handling, feeding strategies,

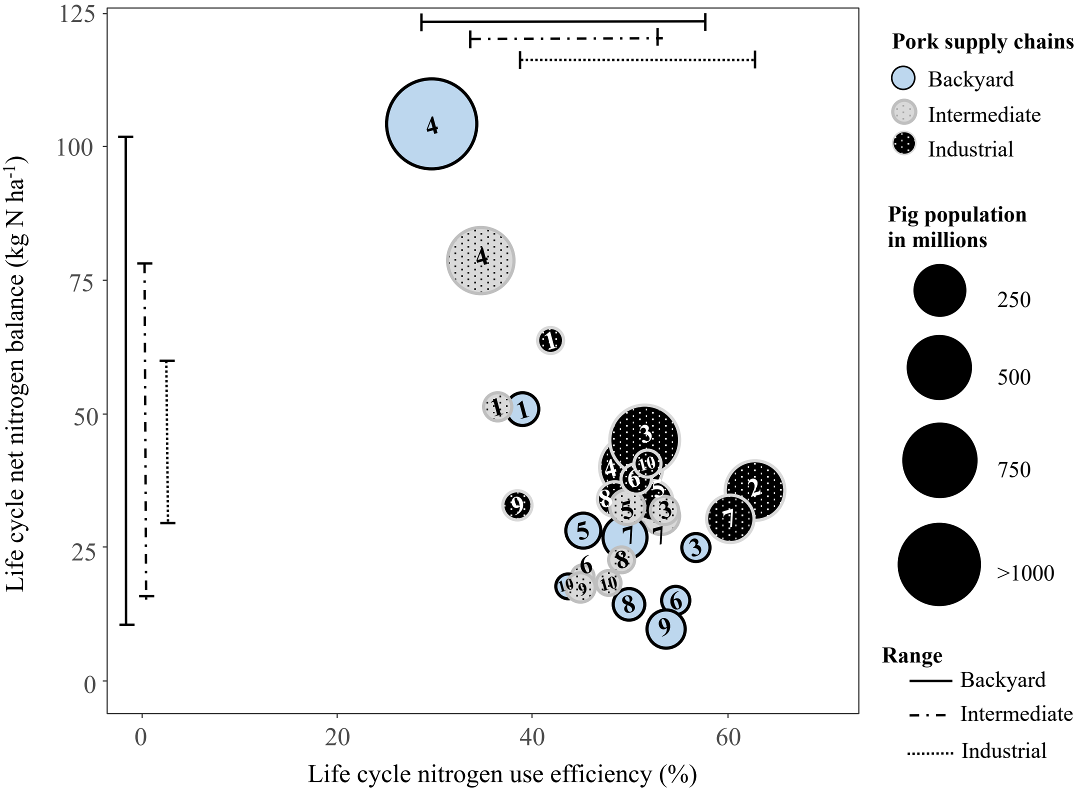


Figure 5.4: The relationship between life-cycle nitrogen use efficiency and life-cycle net nitrogen balance for global pork supply chains for the baseline, see Table B.7. The size of the bubble represents the pig population by region. Description of regions: 1: South Asia, 2: North America, 3: Western Europe, 4: East and Southeast Asia, 5: Eastern Europe, 6: Oceania, 7: Latin American and the Caribbean, 8: Russian Federation, 9: Sub-Saharan Africa, 10: Near East and North Africa. The interval lines show the ranges for the various production systems.

and pig genetics.

Life cycle nitrogen use efficiency

Global Life-cycle-NUE$_N$ of pork production is 41%, which is estimated by considering all N flows at each stage. Figure 5.4 reveals a considerable variability of Life-cycle-NUE$_N$ among regions and systems. For backyard systems, Life-cycle-NUE$_N$ ranges from 30% in ESEA to 57% in WE (global average value: 28%). For intermediate systems, Life-cycle-NUE$_N$ ranges from 35% in ESEA to 54% in WE (global average value: 37%), whereas, for industrial systems, Life-cycle-NUE$_N$ ranges from 39% in SSA to 63% in NA (global average value: 64%). The comparison by region shows that in some, backyard systems have higher Life-cycle-NUE$_N$ than industrial systems (e.g., SSA, WE), whereas in other regions it is the opposite (e.g., LAC). For industrial systems, a low feed conversion ratio, high animal productivity, and proper manure handling explain high efficiencies. For backyard systems, the use of crop co-products and swill as feed and reliance on soil N stock partly compensates for high N losses from manure management and benefits Life-cycle-NUE$_N$. In the situation where high efficiencies coincide with low N input levels (such as in SSA), the system is estimated to deplete N stock in the soil, which will reduce efficiency in the long-run (Uwizeye et al., 2016a).

These results are different to those reported in the literature because of methodological differences. Liu et al. (2017) found that global NUE$_N$ of mixed crop-animal systems ranged from 5% to 65%, Their study included all livestock species and excluded processing. Our findings of NUE at animal production only range from 62% to 91% (SM, Table B.6) and are higher than those reported in the literature, ranging from 10% to 44% (Gerber et al., 2014) or from 7% to 25% (Lassaletta et al., 2016). Discrepancies are caused by the use of different definitions of valuable outputs, i.e., the inclusion or exclusion of exported manure as valuable output. In this study, manure is considered part of useful outputs because of its role in soil fertility and quality (Uwizeye et al., 2016a), resulting in high efficiency. For feed production, NUE$_N$ values (range 40%-80%, SM, Table B.6) agree with those of Lassaletta et al. (2016).

Hotspots of nitrogen losses

NHI$_N$ values are above 100% in most regions, for all systems, indicating a concentration of N losses in one stage of the supply chain: feed production as previously shown in 5.3.1. For instance, about 90% of N losses take place during feed production in SSA for industrial systems, explaining an unusually high NHI$_N$ (158%). Mitigation options, therefore, should focus on this stage, by timing and dosing the application of synthetic fertilizer and manure based on crop requirements and adopting low-emission application techniques such as dilution of the slurry or injection of manure into soil. For NHI$_N$ <100%, N losses are spread evenly across different stages of the supply chain. For instance, for the backyard system in OC, 53% of N losses take place during feed production and 41% in animal production (Table 5.2). Mitigation options would need to focus on all stages of the supply chain to reduce the N losses, for example by lowering the synthetic fertilizer application rate and covering manure during storage to limit the volatilisation.

Combination of nitrogen use indicators

Figure 5.3 shows that, as expected, low Life-cycle-NNB$_N$ is associated with high Life-cycle-NUE$_N$ and vice-versa. There are some exceptions. For instance, backyard systems in LAC and Russia Federation (RF) have equal Life-cycle-NUE$_N$, but Life-cycle-NNB$_N$ in LAC is two-fold higher than the one of RF. Similarly, NA in the industrial system has high Life-cycle-NUE$_N$ combined with a high Life-cycle-NNB$_N$. The reason is that pig production is geographically concentrated in these regions, resulting in high N losses per unit of land use. For these regions, mitigation options would target the reduction of N losses per unit of land while preserving the level of NUE by adopting low-emission manure treatment techniques such as biological nitrogen removal or bio-thermal drying. It is essential, however, to evaluate their environmental performance to avoid shifting impacts from one environmental category (e.g., freshwater eutrophication) to

Table 5.4: Comparison Life-cycle-NUE_N between baseline and scenario and change in N losses at feed production stage after substitution of swill for grain and soybean in industrial pig system.

Regions	Life-cycle-NUE_N			Change in N losses in
	Baseline (%)	Scenario (%)	Change	feed production Stage
South Asia	42	54	30%	-53%
North America	63	67	6%	-28%
Western Europe	51	58	13%	-33%
ESEA[1]	50	58	17%	-56%
Eastern Europe	52	59	12%	-35%
Oceania	51	57	13%	-31%
LAC[2]	60	66	9%	-50%
Russian Federation	48	57	18%	-53%
Sub-Saharan Africa	39	41	7%	-11%
NENA[3]	52	60	17%	-49%

[1]Eastern and Southeast Asia
[2]Latin America and the Caribbean
[3]Near East and North Africa

another (e.g., climate change), because these techniques are energy intensive (Corbala-Robles et al., 2018).

5.3.2 Scenario analysis

Effect of swill feeding on N use indicators and N losses

The substitution of swill for grains and soybeans in pig rations improves N use indicators (Table 5.4 and Table B.8). Life-cycle-NUE_N increases in all regions from 6% in NA to 30% in SA. The difference between regions is explained by the difference in the amount of swill added to the ration, itself depending on the baseline proportion of grains and soybeans and the feed ration composition. The low level of Life-cycle-NUE_N improvement modelled in SSA, LAC and NA are related to lower protein levels in the baseline feed rations (Table 5.1). The increase of efficiency is achieved through a reduction of N losses at the feed production stage, ranging from -56% in ESEA to -11% in SSA. As expected, no significant decrease in N losses is observed at animal production stage because of the conservation of the level of N intake in the substitution scenario. This decrease of N losses translates to the reduction of N losses per unit of area and hotspots along the supply chains.

Globally, this scenario would achieve a saving of 31 Mt of soybeans and 20 Mt of grains in DM, equivalent to 16 Mha of land used. On the global basis, this area saved is equal to 11% of arable land used to produce oilseeds and 1% of arable land used to produce cereals for livestock feed (Mottet et al., 2017). These findings agree with previous studies that analysed the effect of feeding food loss and leftovers on land use for pig production in Europe (zu Ermgassen et al., 2016; van Zanten et al., 2016). Swill feeding has not been considered a potential large-scale strategy to improve environmental sustainability in global pig systems, except by Lassaletta et al. (2016). The reason is that swill feeding is illegal in many countries due to the risks of infectious diseases and public health, when swill is not adequately treated (Horst et al., 1997). For instance, illegal feeding of untreated swill triggered the foot and mouth disease outbreak in the United Kingdom in 2001, causing high economic losses (around £3 billion). Since then, policy-makers have adopted the precautionary principle to reduce potential risks of swill (Haydon et al., 2004).

Feasibility of the use of swill in pig feeding

Swill feeding for industrial pigs is practised in a few countries such as Japan and South Korea. These countries have put in place innovative policies and regulations for the collection, preparation and heat-treatment of swill to improve safety and traceability (Takata et al., 2012). For instance, Governments have developed education programs for urban households for the separation of food wastes and incentivised private sector investments in manufacturing units that collect, treat and process food wastes into animal feed is crucial (Liu et al., 2016). Initiatives such as the "No Food Loss Project" by the Japanese Government promote the reduction of FLW and the use of swill as animal feed (Liu et al., 2016). A specific "Ecofeed" market premium is developed in Japan to provide market incentives to pig farmers.

In other countries, raising awareness with pig production stakeholders on the positive effects of swill feeding on the environment would pave the way towards swill legalisation. A recent study found a strong support among farmers and stakeholders for swill legalisation in the United Kingdom, despite concerns about disease control and consumer acceptance (zu Ermgassen et al., 2018). The availability of swill is a limiting factor to the production of pork from this feed resource. Efforts by food policy-makers and private stakeholders to reduce FLW, furthermore, may eventually reduce the amount of swill available for feed (Garrone et al., 2016; HLPE, 2014). Despite these efforts, a minimum amount of FLW seems to be unavoidable, e.g. non-edible parts of food such as fruit skins (Papargyropoulou et al., 2014) and will be available as animal feed (Beausang et al., 2017).

5.3.3 Limitations

The framework used in this study builds on modular modelling and requires detailed country-specific data, e.g. on N inputs into soils, crop and grass yields, herd parameters per production system, emission factors, and manure management. Most of these data were derived from the GLEAM database. The development of GLEAM involved an extensive effort to compile national statistics and inventories, literature and expert knowledge at global level (FAO, 2018a; Gerber et al., 2013; MacLeod et al., 2013). The uncertainties of the data have a large influence on the results. For instance, a previous study using this framework identified synthetic fertiliser, manure application and deposition, crop yields and emission factors as the most important data to explain the variance of Life-cycle-NUE_N, Life-cycle-NNB_N, and NHI_N (Uwizeye et al., 2017). The data used in this study, however, do not capture the variability of N use practices within a system in the country, which can be observed if spatial differentiated data are available.

For some parameters, country-specific data are lacking, and we use global or regional default values, such as for emissions factors, and gross energy and N contents of feed materials, which increases the uncertainty of our results. Other uncertainties are related to the mass balance approach used, which may under/overestimate the amount of manure exported out of the supply chain or the magnitude of soil N stock change. These parameters can profoundly influence the results. Other limitations are related the data on swill used in this study. These data may have large uncertainties because they have been estimated based on limited literature available (FAO, 2011b), but are a good starting point to explore the potential of swill feeding. The scenario tested, in which we replace a large amount of soybean and grains for swill in the rations, therefore, is not meant to be realistic in all countries, but to assess the technical potential.

5.4 Conclusion

This study provides an overview of N flows, losses, and three N use indicators in global pig supply chains and explores the land use and N management implications of partially replacing grains and soybeans with swill for industrial pigs. It provides sound information on the N use efficiency, the magnitude of N losses and the stage at which they take place along the supply chain. This information is of direct relevance to policy or decision makers interested to improve the sustainability of N management along the supply chain, by considering non-local N losses associated with the import of feed. This study revealed regional and system heterogeneities for N use indicators that emphasise the need for region-specific mitigation strategies. The substitution of swill for soybean and grains in industrial pig diets reduced N losses at the feed production stage and thus improved Life-cycle-NUE$_N$. Swill feeding has the potential to reduce land use and waste streams, especially in countries where swill is widely available. Swill feeding will require innovative policies to frame the collection, treatment, and usage of swill, and ensure safety and traceability. Further research should investigate the N flows and N use indicators in the global livestock sector and evaluate the potential impacts of swill feeding on other environmental issues such as climate change.

5.5 Acknowledgements

This work is supported by the Teagasc Walsh Fellowship Scheme (Ref: 2012230), the Livestock Environmental Assessment Performance (LEAP) Partnership (GCP/GLO/369/MUL) and the Livestock Information, Sector Analysis and Policy Branch (AGAL) of Food and Agriculture Organization of United Nations (FAO). We would like to thank Rogier Schulte for his advice on the design of this research.

Chapter 6

A disaggregated assessment of nitrogen use and emissions in global livestock supply chains

Aimable Uwizeye[1,2,*], Imke J.M. de Boer[2], Carolyn I. Opio[1], Rogier P.O. Schulte[3] , Alessandra Falcucci[1], Giuseppe Tempio[1], Felix Teillard[1], Flavia Casu[1], and Pierre J. Gerber[1,2,3]

[1]Food and Agriculture Organization of the United Nations, Animal Production and Health Division, Viale delle Terme di Caracalla, 00153 Rome, Italy.
[2]Animal Production Systems group, Wageningen University & Research, PO Box 338, 6700 AH, Wageningen, the Netherlands.
[3]Farming Systems Ecology, Wageningen University & Research, P.O. Box 430, Wageningen 6700 AK, The Netherlands
[4]The World Bank, 1818 H Street, NW Washington, DC 20433 USA.

This chapter is under preparation for submission.

Abstract

Although the global livestock supply chains have effectively supplied animal-source food to a growing population, they have also altered the nitrogen (N) flows, threatening the environment and human health. A quantitative assessment of their impacts on N flows is however lacking. Here, we fill this knowledge gap by providing a disaggregated assessment of N use and emissions in global livestock supply chains for 2010. We find that livestock supply chains are responsible for 65 Tg N y^{-1} , which is roughly one-third of the total human-induced N emissions, of which 63% take place in 2 regions (i.e. South Asia and East and Southeast Asia), and 61% at the feed production stage. These emissions are in the form of $NO_3{}^-$ (28 Tg N y^{-1}), NH_3 (26 Tg N y^{-1}), NO_x (8 Tg N y^{-1}) and N_2O (2 Tg N y^{-1}). The magnitude and concentration of N losses imply that there is both urgent need to reduce these emissions and the opportunity to design targeted mitigation interventions. The wide range of values calculated for N use indicators further indicates that good practices are available and already implemented in parts of the value chains. The design and implementation of interventions should consider potential trade-offs and synergies with other sustainability dimensions, such as climate change, resource scarcity and food security. Our study suggests that N challenges are global and cannot be tackled without considering the contribution of global livestock supply chains, thus requiring a global convention with a strong representation of stakeholders in the livestock sector.

6.1 Introduction

The global livestock sector is rapidly transforming. Over the past few decades, many livestock systems over the world have evolved from local, small-scale mixed crop-livestock systems to global and demand-driven supply chains, in which feed and animal production stages are often disconnected (De Haan et al., 2010; Freeman et al., 2006). These changes, driven by economic opportunities, have altered the way livestock production impacts global nitrogen (N) flows, which, overall, have transgressed the safe operational space, defined by planetary boundary thresholds (Springmann et al., 2018; Steffen et al., 2015). Globally, N flows are increasingly imbalanced, with a few countries consuming around 95% of synthetic N fertilizer, while the rest of the world have little access to N fertilizer thus limiting their crop and animal production (Sutton et al., 2013b).

The livestock sector contributes to shaping global N flows through the application of synthetic N fertilizer and manure to produce feed, the accumulation and management of manure, and the transportation of N-rich products, such as feed, food, and manure (Bai et al., 2018b;a; Costanza et al., 2008; Sutton et al., 2011). These developments have changed the pattern of atmospheric N emissions such as nitrous oxide (N_2O), a potent greenhouse gas, ammonia (NH_3) and nitrogen oxides (NO_x), which contribute to air pollution, damage of human health and terrestrial acidification (Galloway et al., 2008; Sutton et al., 2013b). They have also increased emissions of nitrates ($NO_3{}^-$) and organic N, which are abundant sources of water pollution and groundwater and freshwater eutrophication (Hamilton et al., 2018; Galloway et al., 2008). $NO_3{}^-$ emissions can infiltrate into lower soil strata and can be stored in vadose zones, turning into delayed sources of groundwater pollution (Ascott et al., 2017). This delayed pollution may cause long-term damages to human health, aquatic and terrestrial biodiversity, and exacerbate climate change (Ascott et al., 2017; Sutton et al., 2013a).

The urgency with which these environmental threats must be understood and mitigated, while maintaining the livelihoods of hundreds of millions of people, has resulted in a large body of research on the management of N in livestock systems and indeed the global livestock supply chains. Most studies have focussed on the animal production stage of the livestock supply chain — including manure management and on-farm feed production, but excluded other stages, in which most of N emissions can take place (Bouwman et al., 2013; Liu et al., 2017).

Some studies have considered the entire livestock supply chain, as a component of broader food systems (Bodirsky et al., 2014; Conijn et al., 2018; Springmann et al., 2018; Sutton et al., 2013a) or of global economy (Oita et al., 2016), but their level of aggregation and simplification does not allow to draw specific conclusions for the livestock sector. A recent study provided a more detailed analysis of the global acidification and eutrophication induced by the production of animal-source food (Poore and Nemecek, 2018), but it relies on a database dominated by observations for commercial farms in industrialised countries. A comprehensive, yet detailed analysis of the contribution of the livestock supply chains to N emissions was carried out by Leip et al. (2015), for the European Union. It suggests that these supply chains represent 82% of the total NH_3 emissions and 73% for the entire N emissions to water bodies from agriculture. In summary, none of this work has performed a global, yet disaggregated assessment (i.e., spatially differentiated and distinguishing between different species, commodities and production systems) of N use in livestock supply chains and its contribution to N emissions.

Here we fill this knowledge gap, with the aim to elucidate the magnitude and diversity of N use in global livestock supply chains. Our study focuses on 275 countries and territories grouped in 10 regions (http://www.fao.org/geonetwork/srv/en/metadata.show?id=12691), using an updated version of the Global Livestock Environmental Assessment Model (GLEAM) for 2010. We use the most detailed geo-referenced information, highlighting the diversity of livestock supply chains. We, furthermore, identify hotspots of N emissions and design targeted improvement pathways.

6.2 Methods

6.2.1 Overview of GLEAM model

GLEAM is a spatially differentiated biophysical model developed at the Food and Agriculture Organization of the United Nations (FAO) to assess the contribution of global livestock supply chains to environmental issues (FAO, 2018a). Its structure is based on a life-cycle approach and covers the main stages of livestock supply chains, including feed crop and grass production, animal production, processing of animal products and transport. GLEAM has been used to estimate the contribution of the livestock systems to the global human-induced emissions of greenhouse gas (Gerber et al., 2013) and is here further developed to estimate nitrogen flows and associated emissions. GLEAM accounts for N flows and emissions at a resolution of 10 x 10 km at the equator, for a combination species, commodity, production system and agro-ecological zones. We apply the indicator framework developed by Uwizeye et al. (2016a) to estimate the life-cycle nitrogen use efficiency (Life-cycle-NUE$_N$), life-cycle nitrogen net balance (Life-cycle-NNB$_N$).

6.2.2 Development of the model

Specific new developments of GLEAM were carried out to perform this analysis.

N modelling in soils

We upgraded the feed module in GLEAM to account for all sources of N entering the soil; including biological N fixation, synthetic fertilizer, manure, crop residues and atmospheric N deposition and soil N stock change. We integrated a stepwise approach to reflect the N mass balance for each feed item and to account for N emissions from each source of N inputs (FAO, 2018a). We used the IPCC method to estimate NH_3 volatilisation and N_2O emissions from soils. Then, we combined the information on global land cover, soil classes, slope classes, and precipitation to calculate NO_3^- loads via runoff (Velthof et al., 2009). The fraction of N emissions via leaching was estimated using a mass balance approach. The release of N_2 was considered as a recycle flow to the atmosphere and, although calculated, it was excluded in the further analysis of N emissions. We also estimated NO_x emissions from field operations, harvesting and crop processing using a proxy of CO2 to NO_x ratio (Carslaw and Rhys-Tyler, 2013).

N modelling in international transport

Initial trade matrices were obtained from FAOSTAT for the following individual products: soybeans, soya cake, whole maize, maize bran, whole wheat, wheat bran, barley, whole cassava, dried cassava, palm cakes. To homogenise year-to-year variations in trade flows, a three-year average (2009-2011) was calculated. We then computed trade matrices for aggregated items (soy, maize, wheat, barley, palm and cassava) by summing individual items and applying FAOSTAT conversion factors between crop primary and secondary products (FAO, 2003; Kastner et al., 2014).

For each exporting country, if export was higher than the production, the difference was considered as re-export. Thus, the proportion of re-exports to total exports was calculated and assumed similar for each aggregated item and all its related individual items. For every single item, trade flows between re-exporters and final importers were reallocated to flows between primary exporters and final importers. This reallocation was done proportionally – re-exports were reassigned to primary exporters, according to their relative contribution to the imports of re-exporters. Then, they were allocated to final importers according to their relative contribution to the exports of re-exporters. For each item, the result was a corrected trade matrix with the same total volumes of trade and modified trade flows to link primary exporters to final importers directly.

We also corrected the trade matrix for feed/food use. Among the different individual items considered, it was assumed that 100% of soybean cake, maize bran and wheat bran were used as feed and hence allocated to livestock production. For the other items (wheat, barley, maize grains, soybeans, palm, and cassava), the total feed intake of all livestock species was retrieved from GLEAM, and we assumed that the total feed intake came from imports and national production proportionally to their relative value in each country. We estimated sea transport distances associated with the international trade of feed commodities: wheat, barley, maize, cassava, soybean and palm, based on the database developed by CERDI (French Centre for Studies and Research on International Development) (Bertoli et al., 2016). For major exporting or importing countries with a large area and several important ports (e.g. Argentina, Australia, Brazil, Canada, China, and the United States), sea distances were calculated by considering their two main ports (weighted average). A sea distance matrix for each feed commodity was thus created.

We estimated the fuel consumption, assuming an average fuel consumption of 1.3 g t^{-1} km^{-1} based on Notteboom and Cariou (personal communication). We assumed that 86% of fuel consumed was in the form of heavy fuel oil and the remainder in marine diesel oil across all countries (IMO, 2014). We then used the EEA approach (EEA and CLRTAP, 2016) to calculate the total NO_x emissions by multiplying the distance and volume of a commodity in the corrected FAO trade matrix, as well as the associated fuel consumption and emission factors. Finally, we assigned NO_x emissions to each production system proportionally to the volume of feed commodity used.

N losses during manure management

We borrowed the method developed by EEA (EEA and CLRTAP, 2016). This method estimates N emissions from houses or yards and storage from the fraction of the total ammonia nitrogen (TAN), which represents the total amount of N in the forms of NH_3 and ${NH_4}^+$. We estimated TAN based on mineralised N in urine from N in faeces according to Vonk et al. (2018). N emissions were estimated by multiplying TAN by the share of each manure management category and the corresponding emission factor. For NH_3, we considered emissions from house or yard and the manure storage, whereas for N_2O emissions, direct and indirect emissions from manure storage, and manure leaching were estimated. For NO_x emissions, we distinguished emissions from manure used as biofuel or manure incinerated to recover energy from those from manure management. ${NO_3}^-$ emissions were estimated based on manure leaching and unregulated disposal into surface and groundwater based on IPCC (IPCC, 2006) and literature data (Bai et al., 2014; Huang et al., 2016; Schaffner et al., 2009; Thien Thu et al., 2012; Vu et al., 2007).

N losses after farm gate

A mass-balance approach was used, and N emissions were estimated as the difference between N in primary products and live-animals and N in final products. It was assumed that most of N was lost in the form of wastewater and untreated organic wastes from slaughterhouses and milk processing plants.

6.2.3 Data collection

The GLEAM database was completed and updated for a number of topics. For feed production, we used the new version of the Global Agro-Ecological Zones (GAEZ) yield maps (Fischer et al., 2012) for feed crops (resolution: 10 x 10 km at the equator). We added new data on biological N fixation for legumes, estimated based on the Livestock Environmental Assessment and Performance guidelines (FAO, 2018b). For other non-legume crops, we considered default values from literature (Herridge et al., 2008; Peoples et al., 2009). Crop-specific data on synthetic fertilizer per ha were obtained by dividing the total fertilizer consumption by crop from the International Fertilizer Association (Heffer et al., 2017) and the harvest area from FAO statistics (FAOSTAT, 2018) for the main fertilizer consuming countries. Other data on synthetic fertilizer were obtained from Common Agricultural Policy Regionalised Impact model (CAPRI) for Europe (Leip et al., 2011), from Swaney et al. (2018) for the United States at a subnational level. For Australia, data were obtained from Navarro et al. (2016), while we used FAO statistics data for the rest of the world. Data on atmospheric N deposition were obtained from the literature (Dentener, 2006). Manure deposited on grassland and applied to cropland was calculated iteratively from the model, prioritising the application of manure to available arable lands in the cell where it was produced before applying it to other surfaces (grassland or grazed marginal land). Crop residues data were calculated from GAEZ yield maps based on IPCC equations (IPCC, 2006). Data on global land cover (Latham et al., 2014), slope classes (Reuter et al., 2007), and precipitation (Harris et al., 2014) were used to calculate spatially explicit runoff rate. For imported feed items, we estimated N inputs and yield data as the national average weighted by the trade volumes reported in the FAO trade matrix (FAOSTAT, 2018). For countries with missing data, we filled the gaps with regional or continental average data.

For animal production, we collected additional data on manure management for the main livestock producing countries. Data were based on national greenhouse gas inventories for Brazil, Australia, Japan, Switzerland and New Zealand reported to United Nations Framework Convention on Climate Change UNFCCC, NH_3 inventory for United States (EPA, 2004), and national statistics for Canada (Canada, 2003). Data for the European Union were detailed at the NUTS2 level (Bioteau et al., 2009), whereas data for China, India, Mexico and Vietnam were derived from literature (Dan et al., 2003; Gao et al., 2014; Gupta et al., 2007; Mink et al., 2015; Thien Thu et al., 2012).

6.3 Results

6.3.1 A substantial contribution to human-induced nitrogen emissions

Our analysis indicates that the livestock supply chains contribute around 65 Tg N y^{-1} to the human-induced N emissions for 2010. Most of these N emissions are in the form of NO_3^- (28 Tg N y^{-1}), NH_3 (26 Tg N y^{-1}), NO_x (8 Tg N y^{-1}) and N_2O (2 Tg N y^{-1}).

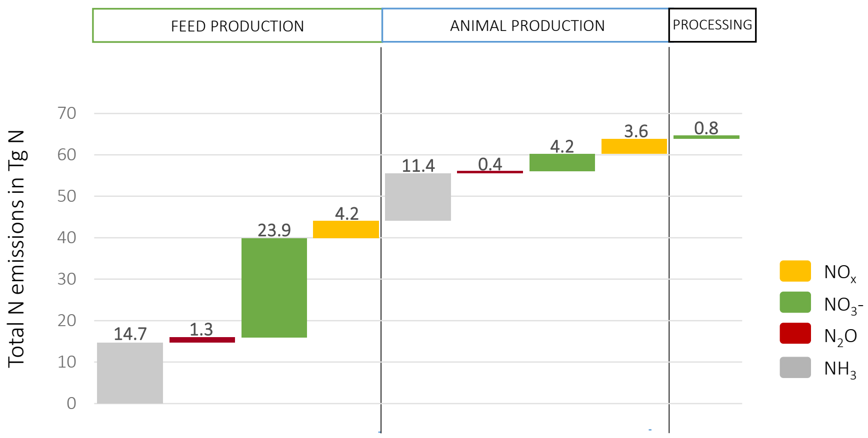


Figure 6.1: Contributions of the sources of N compounds emissions to the global N emissions for livestock supply chains

The bulk of N emissions takes place during feed production and manure management as illustrated in Figure 6.1. Feed production releases 44 Tg N y^{-1}, in particular through manure deposited on grasslands, manure spreading and synthetic fertilizer application to cropland, and crop residues recycled in the field. Manure management is the second source of N emissions with 20 Tg N y^{-1} lost through volatilisation, N leaching and manure use as energy sources, whereas the N emissions from animal product processing are minor in comparison.

6.3.2 Regional distribution of N emissions

Most of N emissions take place in South Asia (23 Tg N y^{-1}), East and South-Eastern Asia (18 Tg N y^{-1}), and Latin America and the Caribbean (7 Tg N y^{-1}), as shown in Figure 6.2. In South Asia, both buffalo and cattle production are responsible for 86% of N emissions. In Latin America and the Caribbean, beef and dairy cattle production systems account for 70% of the N emissions estimated in this for the region. Cattle production also contributes highly to N emissions in Sub-Saharan Africa and North Africa and Near East, while pig and cattle systems are the main contributors in Western and Eastern Europe.

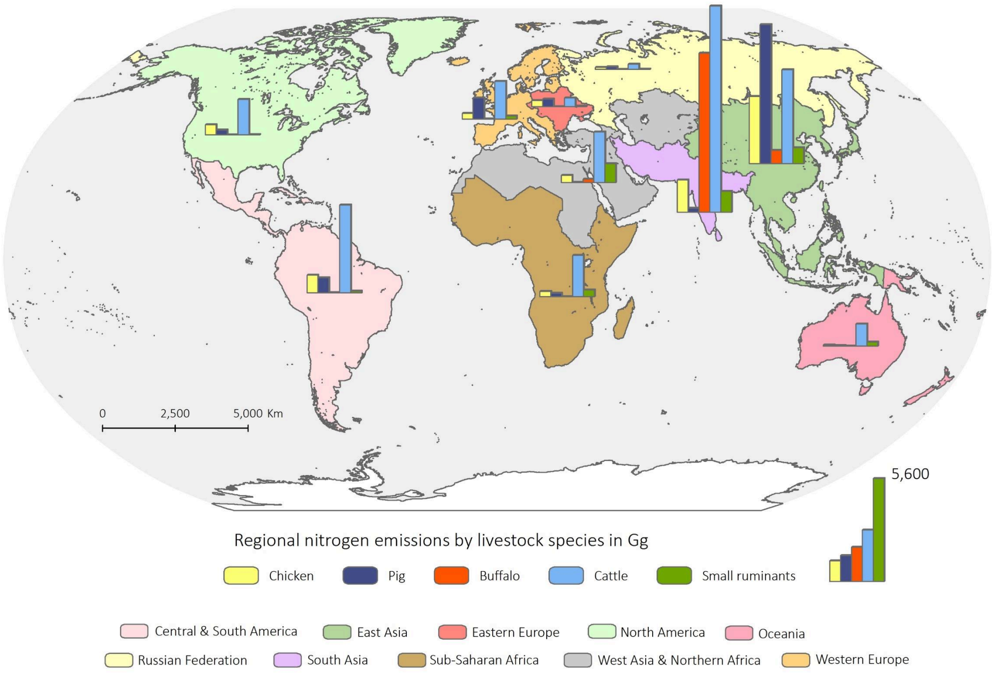


Figure 6.2: Distribution of N emissions by the livestock species for ten regions expressed in Gg N

6.3.3 A few types of supply chains dominate emissions

We find that globally, cattle, buffalo, goat and sheep supply chains release about 45 Tg N y^{-1}, representing 70% of the total N emissions from livestock. More specifically, 10 supply chains contribute nearly 85% of the total N emissions from livestock supply chains (Figure 6.3). Mixed cattle and buffalo systems alone – most of which are in South Asia – are responsible for 40% of the total emissions. Emissions from grazing cattle (dairy and beef) and pig systems (backyard, intermediate and industrial) are similar at 16% of the total N emissions.

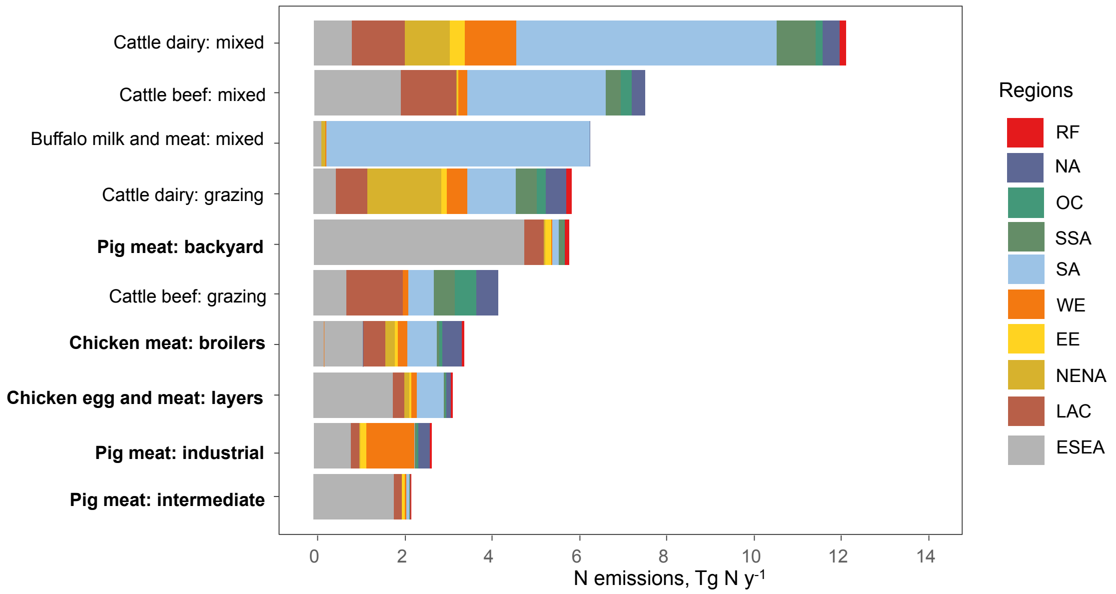


Figure 6.3: The regional contribution of different livestock systems to the total N emissions. Regions in the legend are indicated by their acronym: RF: Russia Federation, NA: North America, OC: Oceania, SSA: Sub-Saharan Africa, SA: South Asia, WE: Western Europe, EE: Eastern Europe, NENA: Near East and Northern Africa, LAC: Latin America and Caribbean, ESEA: East and South-Eastern Africa. The colour indicates a livestock species. Monogastric systems are presented in the bold case, whereas ruminant systems are in the normal case.

6.3.4 Areas of concentration

We find considerable spatial variability in aggregated N emissions along livestock supply chains (Figure 6.4). High NH_3 and NO_3^- emissions per unit of land are modelled for the Indo-Gangetic plain, East and South-Eastern Asia, Western Europe, Oceania, the Nile Delta and Latin America. In the Indo-Gangetic plain, these emissions are related to a high density of cattle and buffalo associated with poor manure management and high synthetic fertilizer application (Beig et al., 2017). In most of the East Asian countries, the geographical concentration of animal population such as large-scale pigs, chicken, and mixed dairy farms, unregulated manure disposal and high synthetic fertilizer application explain high N emissions (NH_3, N_2O, and NO_3^-) modelled. Most of NH_3 and N_2O emissions related to pig, chicken and cattle production, take place in Western Europe, North America and Latin America N emissions from manure management and daily spreading are, therefore, concentrated on relatively smaller agricultural areas, resulting in high emissions per ha. The same concentration of cattle and pig systems in North America and Western Europe also translates into hotspots of NH_3.

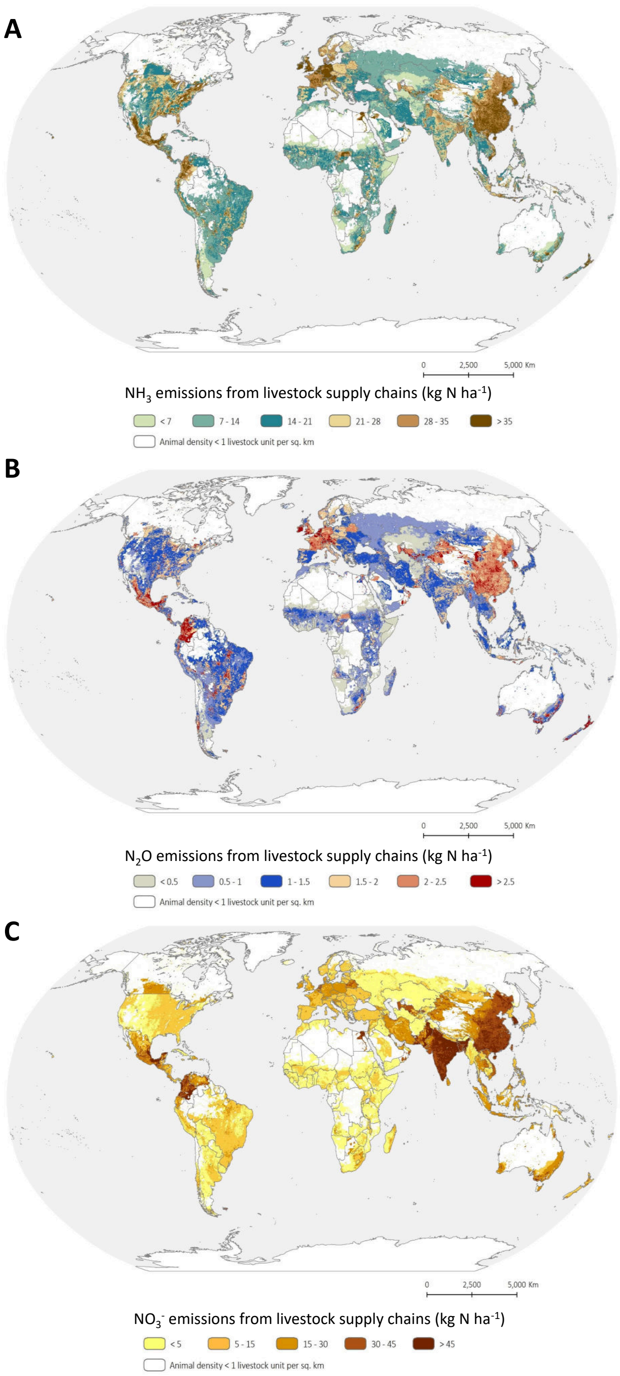


Figure 6.4: Spatially distribution of N emissions from livestock supply chains. The emissions are aggregated for all livestock species and consist of N emissions taking place in feed production, manure management and processing of animal products per unit of land required to produce feed. A. NH_3 emissions B., N_2O emissions to the atmosphere and C. NO_3^- emissions to surface and groundwater.

6.3.5 Highly variable nitrogen use indicators across regions and production systems

We analyse Life-cycle-NUE$_N$, referring to the efficiency of recovering N mobilised at each supply chain stage into the end-animal products, across production systems (Figure 6.5). Our results show a high variability across the country and within the different livestock systems, indicating the significant differences in livestock management practices, feed resources and animal performance around the world. Such variability is more significant in ruminant as compared to monogastric systems, except for the beef cattle feedlot.

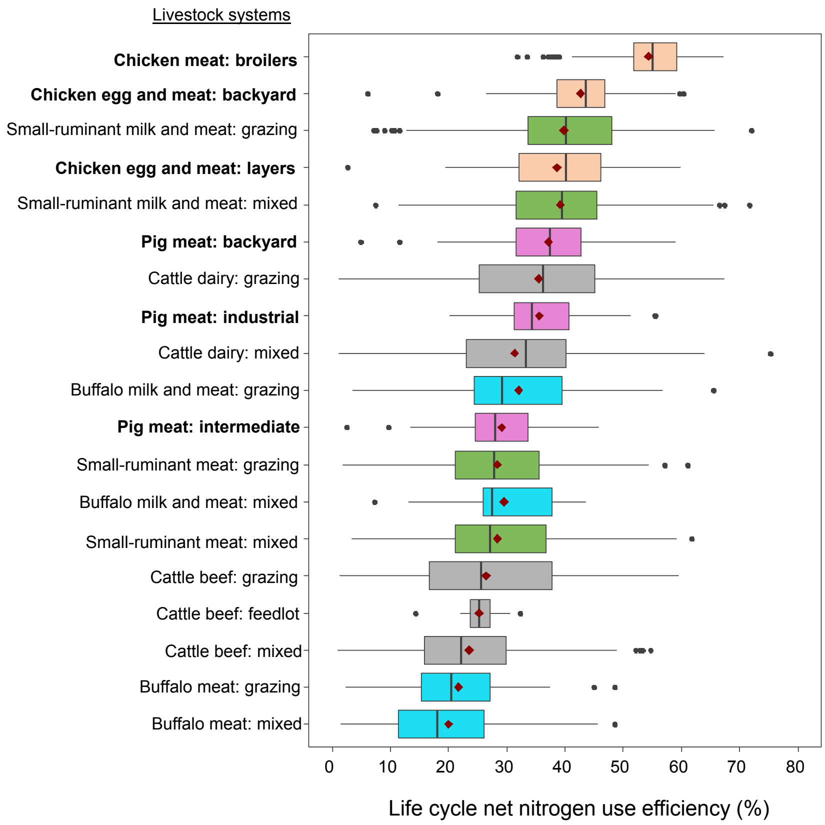


Figure 6.5: The distribution of life cycle nitrogen use efficiency by species, commodity and systems. The systems are ranked in decreasing order of the median values. The box shows the 25th to 75th percentile, and the dots represent outliers. The colour indicates a livestock species. Monogastric systems are presented in the bold case, whereas ruminant systems are in the normal case.

We find relatively high-efficient systems for both monogastric and ruminant species (Figure 6.5). The broiler chicken systems are found to be the most efficient system, with Life-cycle-NUE$_N$ values ranging from 32% to 67%. Highest median Life-cycle-NUE$_N$ values (>36%) are found for broiler, backyard and layers chicken, grazing and mixed small ruminant milk, backyard pig and grazing dairy cattle, despite the significant differences in animal performance and herd management practices across these production systems. The median Life-cycle-NUE$_N$ values of industrial and intermediate pigs, grazing and mixed dairy cattle, and grazing buffalo milk are medium, ranging from 28 to 36%. Low values of the median are found for grazing and mixed buffalo meat, mixed beef and feedlot cattle, implying that producing meat from ruminant is less efficient than monogastric due to the differences in animal physiology and feed resources between them.

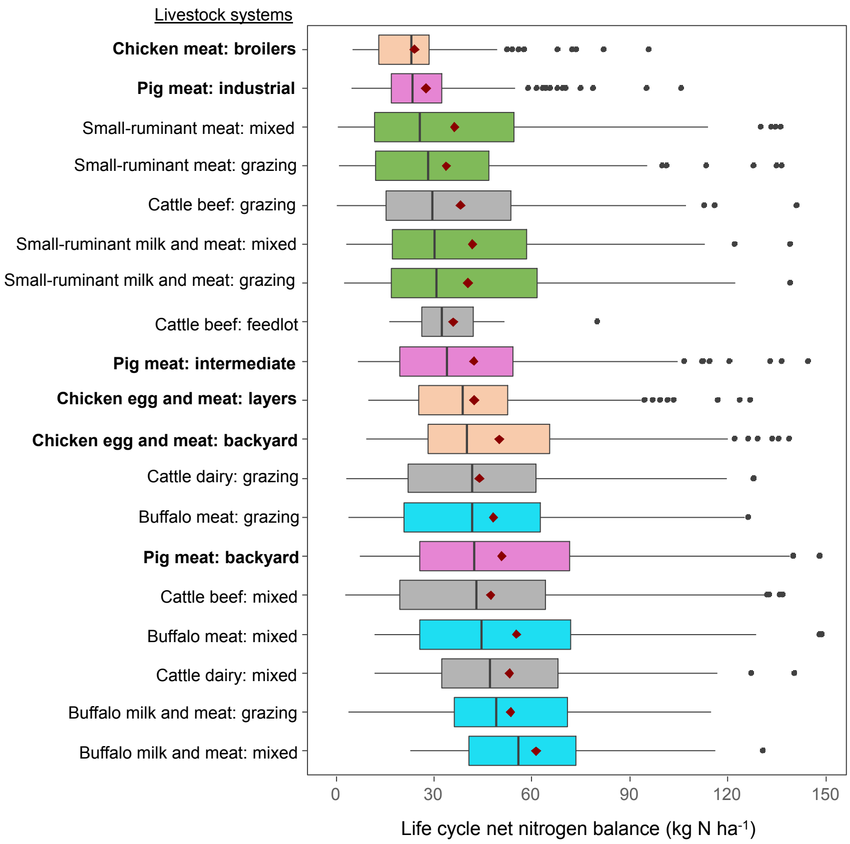


Figure 6.6: The distribution of life cycle net nitrogen balance for global livestock supply chains by species, commodity and systems. The systems are ranked in increasing order of the median values. The box shows the 25th to 75th percentile, and the dots represent outliers. The colour indicates a livestock species. Monogastric systems are presented in the bold case, whereas ruminant systems are in the normal case.

Life-cycle-NNB$_N$ is computed as the total N emissions by system divided by the entire area required to produce feed (Figure 6.6). The lowest values are modelled for broiler chicken, industrial pigs, small ruminant systems, mixed and grazing beef cattle, beef cattle feedlot and grazing beef cattle. These low values are related either to relatively low losses at an animal (e.g. broiler chicken), or feed production (e.g. industrial pigs, beef cattle feedlot) levels or to the large areas required for production (e.g. small ruminant systems), effectively 'diluting' the Life-cycle-NNB$_N$. The highest median values for Life-cycle-NNB$_N$ are estimated for grazing and mixed buffalo milk and meat, mixed buffalo meat, mixed dairy cattle, cattle beef feedlot, implying that these systems have high N emissions and/or are concentrated on relatively limited land areas.

6.4 Discussion

6.4.1 5.1 Contribution to total N emissions and planetary boundaries

N emissions from livestock supply chains computed in this analysis represent roughly 35% of the global agricultural emissions estimated by Bodirsky et al. (2014), or 37% of those same emissions estimated by Conijn et al. (2018), see Table 1. Compared to global N emissions from the entire global economy, they represent 29% of the total N emissions estimated by Hamilton et al. (2018), or 34% of those same emission estimated by Oita et al. (2016). Acknowledging these different sources, we can suggest that the sector and related supply chains contribute about one-third of global human-induced N emissions. Benchmarking against results from Oita et al. (2016), livestock supply chains are responsible for approximately 60% of total NH_3 emissions, 32% of N_2O emissions, 23% of NO_x emissions, and 39% of ${NO_3}^-$ released to surface and groundwater in the world.

We also compare our results with the two approaches of planetary boundaries for N flows available in the literature. The first planetary boundary approach defines the thresholds of annual N surplus to keep the planet in a safe operating space for humanity, which ranges from 67 to 146 Tg N y^{-1} (Springmann et al., 2018). Our estimate for total N emissions from global livestock value chains (65 Tg N y^{-1}) is slightly lower than the lower bound of this boundary. The second approach is based on the total N fixation, either industrially or biologically, required to ensure food security and limit eutrophication of aquatic ecosystems, and ranges from 62–82 Tg N y^{-1}(Steffen et al., 2015). Our estimate of the total N mobilised as synthetic fertilizer and biological N fixation to produce feed (76 Tg N y^{-1}, Table 1) is within the range of this planetary boundary. The livestock sector, therefore, leaves virtually no "N emission allowance" for other sectors.

6.4.2 Validation

As a way of discussing the validity of our results, Table 6.1 compares our results with previous studies, at various scales and for different parts of the value chain. Our estimates are similar to those of Oenema et al. (2014), and comparable to the livestock element of studies that focussed on agricultural systems as a whole (Billen et al., 2014; Bodirsky et al., 2014; Conijn et al., 2018). Overall N intake and N excreted are comparable with the two studies (Billen et al., 2014; Conijn et al., 2018), but differ with Bodirsky et al. (2014), which aggregated these flows at the regional level. Manure applied to cropland computed in this study is lower than the estimates published by Conijn et al. (2018), Billen et al. (2014), and Bodirsky et al. (2014), because this study considers only manure applied to feed crops, whereas these three studies consider manure applied to all crops. NH_3 emissions in this study are lower than those of Conijn et al. (2018), mostly because we allocated emissions form manure used to fertilize food crops to the crop sector, and not to livestock supply chains.

At the regional level, our estimates for NH_3 and NO_x for Western and Eastern Europe are in line with Leip et al. (2015). ${NO_3}^-$ emissions, nevertheless, differ significantly due to differences

Table 6.1: Comparison of our results estimated by GLEAM with previous studies

A. GLOBAL LEVEL	Unit	Animal level		Agriculture systems		
		Our study	Oenema et al. 2014[1]	Bodirsky et al. 2014[1]	Conijn et al. 2018[1]	Billen et al. 2014[2]
Feed production						
Synthetic fertilizer	Tg N y^{-1}	55		116	94	98
Manure applied to crop	Tg N y^{-1}	23		31	30	32
Biological N fixation	Tg N y^{-1}	21		36	43	29
Atmospheric deposition	Tg N y^{-1}	10		21	14	10
Crop residues	Tg N y^{-1}	21		42	51	
Outputs (crop + grass)	Tg N y^{-1}	127		149	152	174
N emissions	Tg N y^{-1}	40		105	102	
Animal production						
N intake	Tg N y^{-1}	120		167	138	132
N excretion	Tg N y^{-1}	98	107		127	121
NH_3 emissions	Tg N y^{-1}	11			28	
N_2O emissions	Tg N y^{-1}	0.4	0.22			
N losses at chain or system	Tg N y^{-1}	65		186	172	

B. REGIONAL LEVEL		Europe		Cattle beef in the US		China	
	Unit	Our study	Leip et al. 2015[3]	Our study	Rotz et al. 2019[4]	our study	Bai et al. 2018[1]
NH_3 emissions	Tg N y^{-1}	2.7	2.3			5.8	7.8
NO_x emissions	Tg N y^{-1}	0.5	0.23				
$NO_3{}^-$ emissions	Tg N y^{-1}	1.4	4.4			7.1	12
N losses	Tg N y^{-1}			1.1	1.7		

Reference year: [1]2010, [2]2009, [3]2004, [4]2013-2017

in animal numbers between 2004 and 2010 and in feed composition. For the United States, our estimates for total N emissions are comparable to those reported by Rotz et al. (2019), despite differences in definition of production systems and reference year. For China, our findings for NH_3 emissions are slightly lower than those of Bai et al. (2018b), because of the allocation of N emissions to manure exports considered in our analysis. A significant difference is found for $NO_3{}^-$ emissions to groundwater and freshwater, because Bai et al. (2014) assumed a higher rate of unregulated pig manure disposal into the environment, while in our study, an average rate was calculated from Huang et al. (2016) and Bai et al. (2014).

6.4.3 Modelling challenges

The framework used in this study, coupled with GLEAM model, is relatively data intensive and requires spatial-explicit information at a pixel level. The level of resolution of the data varied significantly, and so did the level of uncertainties in the data. Previously, synthetic fertiliser, manure application and deposition, crop yields and emission factors were found to strongly influence the variance of N use indicators (Uwizeye et al., 2017). In this study, therefore, we focussed on these variables and collected the most recent and reliable data for the main livestock producing countries (see Methods). Modelling uncertainties can also influence the outcomes of the analysis, in particular, the estimate of N emissions in feed production and manure management. For instance, N emissions depend on agro-climatic conditions such as wind, precipitation and temperature; these factors were only considered for N runoff, while NH_3 and N_2O were calculated based on the IPCC method (IPCC, 2006). Estimates of N emissions from manure were improved by deriving N emissions from local-specific TAN based on EEA recommendations (EEA and CLRTAP, 2016). The manure application rate was improved by considering different land-uses within the pixels where manure is produced. These improvements increased the confidence in our estimates of N flows for global livestock supply chains.

6.4.4 Drivers of N emissions and mitigation options

Our study, providing a disaggregated assessment of global N use and flows, and using a comparable level of granularity and accuracy for all supply chains, allows identifying some significant drivers of emissions worldwide. They are presented here, with a brief discussion of mitigation options.

Feed crop fertilisation. The level of fertilisation of the feed crop and grass production profoundly influences the magnitude of emissions. Our calculations indicate that losses from N inputs account for 95% of N emissions in feed production for pig and chicken systems, 60% in grazing ruminants, and 70% in mixed buffalo and cattle systems. These emissions are related to the use of industrial by-products, and crop residues originated from crops that receive important doses of synthetic fertilizer and manure. A significant exception is a soybean and derived products, for which around 50% of emissions are related to field operations, processing and international transport, due to the low fertilisation of this crop. This driver explains partially the high N emissions modelled in South Asia, East and Southeast Asia, Western Europe and North America.

Accelerations of the adoption of good management practices for the manure and fertilizer application in cropland and grassland systems can significantly reduce N emissions (Gao et al., 2017). Interventions would focus on the revision of fertilizer subsidies and other fertilizer pricing policies (Heffer et al., 2017). They may also incentivise the adoption of low-emission spreading techniques for manure (Sanz-Cobena et al., 2014), use of nitrification inhibitor (Ganeshamurthy et al., 2017; Guardia et al., 2017), closing yield gaps, and improving pasture productivity (Bittman et al., 2014; Reetz Jr et al., 2015).

Manure management. The magnitude of N emissions depends largely on the type of manure, and the technology used for its collection, storage and use or disposal. For all systems, more emissions of NH_3 (51 to 71%) are related to the management of solid manure in animal houses, yards and manure storage, daily spreading of manure to the land, and uncollected manure. High emissions of NO_x in buffalo systems are related to manure used as biofuel (Beig et al., 2017). Manure leaching from storage systems and unregulated disposal are the main drivers of ${NO_3}^-$ emissions from manure management. The reason is that the increase in demand for animal-source food has led to the development of large-scale farms (pig, chicken and dairy cattle), relying on feed imports, in areas with limited cropping activities (MacDonald and McBride, 2009). These farms have less opportunity to recycle manure, resulting in disposal of manure into watercourses (Bai et al., 2018b; Strokal et al., 2016; Schaffner et al., 2009) or over-application of manure to limited land (Costanza et al., 2008). This driver explains the hotspots of ${NO_3}^-$ and NH_3 emissions found in South Asia, East and Southeast Asia, Western Europe, Latin America and North America (Figure 6.4).

Mitigation interventions can focus on incentives to collect, transport and recycle manure to available croplands, but this intervention is often limited by high transport cost (Hendriks et al., 2016; Lauer et al., 2018). This calls for policies that improve the land/livestock balance by favouring the location of animal production units in area offering recycling options (type of crops, soils and water system). Other interventions can focus on the scaling up of innovative solutions such as liquid and solid separation, recovering NH_3 fertilizer from manure (Dube et al., 2016) or the recoupling of crop and animal production.

Animal husbandry and feeding strategy. The heterogeneity of animal husbandry also explains the variability of N use indicators. The production of chicken broilers relies on standardised management practices, high-quality feed and veterinary care, which explains high Life-cycle-NUE_N found. Backyard chicken and pig systems are location-specific and embedded in local food systems, where they rely on food leftovers and scavenging with no associated upstream emissions. Similarly, small ruminant browse tree leaves and other natural vegetation, for which we assume no N emissions or land use. This improves their score on our efficiency and loss indicators (Figure 6.6).

In some countries, innovative feeding strategies, such as the use of food wastes as feed for pigs (Uwizeye et al., 2019), can improve the environmental performance of pork production, while safeguarding health risks. In other countries, feeding high-quality feed, use of enzymes to improve feed digestibility or low-protein feed ration, can increase N retention in animal tissues and reduce N excreted (Bittman et al., 2014; Bodirsky et al., 2014; Oenema, 2006). These strategies, however, may increase the feed-food competition, mainly when the improved ration contains human edible products (van Zanten et al., 2018).

6.5 Conclusions

Our study assesses N use and flows in global livestock supply chains. It suggests that the sector is responsible for about one-third of the total human-induced N emissions, of which 63% take place in 2 regions, and 61% at the feed production stage. This finding implies that there is both an urgent need to reduce these emissions and opportunities to design targeted mitigation interventions. Focussing resources on a few regions, supply chains and steps along the chain can increase the effectiveness of interventions. The wide range of values calculated for N use indicators further indicates that good practices are available and already implemented in different parts of the value chains. Designing and implementing effective improvement interventions will need to recognise that: (i) there may be trade-offs with other sustainability dimensions, and (ii) that in the absence of effective mitigation, or in addition to those, a reduction in the size of the sector may be necessary in some parts of the world to reduce impacts to acceptable levels and keep global emissions within planetary boundaries. Both aspects, however, fall outside the scope of this paper. Our study shows that N challenges are global and cannot be tackled without considering the contribution of global livestock supply chains. This finding calls for a global initiative with a strong representation of livestock sector scientists and stakeholders to tackle the N pollution. Our findings will support dialogue among stakeholders to identify pathways to shape the sustainable development of livestock supply chains. Further research is required to assess the global mitigation interventions proposed and their co-benefits for climate change, and resource scarcity while ensuring food security.

6.6 Acknowledgements

This work is supported by the Teagasc Walsh Fellowship Scheme (Ref: 2012230), the Livestock Environmental Assessment Performance (LEAP) Partnership (GCP/GLO/369/MUL) and the Livestock Information, Sector Analysis and Policy Branch (AGAL) of Food and Agriculture Organization of United Nations (FAO). We would like to thank my colleagues: Monica Rulli for preparing the data and designing the maps, Giuseppina Cinardi for supporting the modelling of ruminants, and Juliana Lopez for her comments on an earlier version of this manuscript.

Chapter 7

General Discussion

7.1 Introduction

The livestock sector has sustained the livelihoods of billions of people since the domestication of animals (Larson and Fuller, 2014). In the last 50 years, its rapid growth has satisfied the demand for animal-sourced food from a growing and urbanised human population. Negative repercussions related to nitrogen (N) and phosphorus (P) use; however, have taken place along several stages of the livestock supply chains, resulting in large-scale environmental issues. Increases in manure and synthetic fertilizer application as well as poor manure management have resulted in increased N and P losses into the environment (Sutton et al., 2013a). The livestock supply chains are responsible for 14.5% of greenhouse gas emissions, including nitrous oxide (N_2O), thus contributing to climate change (Gerber et al., 2013). Emissions of inorganic and organic P and N to ground and surface waters from manure management, grazed grassland or soils cultivated for feed, can result in freshwater and marine eutrophication (Azevedo et al., 2013; Cosme et al., 2017; Hamilton et al., 2018). The implications of these impacts are enormous and can result in damage to human health and ecosystems as well as loss of biodiversity (Sutton et al., 2013a). Together with a large usage of natural resources, these impacts have raised global attention on the role of the livestock in sustainable food systems (Springmann et al., 2018; Sutton et al., 2013a; van Zanten et al., 2018; Röös et al., 2016). Addressing these nutrient challenges is a pivotal aspect for improving the sustainability of livestock supply chains.

The objectives of the thesis were to develop a framework of indicators to assess nutrient flows and emissions along global livestock supply chains, while identifying data, which can be improved to enhance the accuracy of the results; and to assess the impacts of the global livestock supply chains on the nitrogen flows, while exploring the improvement options. For the first objective, we defined the nutrient use efficiency at the chain level as an indicator to benchmark the environmental sustainability of the global livestock supply chains (Chapter 2). Subsequently, we developed a comprehensive framework to assess the nutrient use at a chain level (Chapter 3). Then, we proposed a method to identify the input parameters from global datasets on which improvements should focus to enhance the relevance of the environmental indicators (Chapter 4). In line with the second objective, we applied the framework to the global pig supply chains (Chapter 5) and subsequently to global livestock supply chains (Chapter 6).

While this framework is relevant for the assessment of both N and P flows and emissions, its application focuses primarily on N assessment in livestock supply chains, due to its implications for multiple environmental impact pathways. P is covered in Chapter 2 and 3.

This Chapter discusses the development of the framework of indicators at the chain level (section 7.2) and modelling challenges (section 7.3). It provides an overview of the results on the impacts of livestock supply chains on nutrient cycles and potential improvement pathways (section 7.4). Finally, it discusses the nutrient management in the context of the overall sustainability (section 7.5) and proposes a general conclusion and recommendations (section 7.6).

7.2 The framework of indicators at the chain level

Chapter 2 demonstrated the need for harmonised methods and associated indicators to address the nutrient challenges in livestock supply chains based on the existing literature. The existing methods identified were nutrient balance (NB), nutrient use efficiency (NUE), material flow analysis (MFA) and life cycle assessment (LCA). While NB and NUE focus on the animal, farm or geographical unit (e.g. region, country) as one entity, MFA and LCA focus on nutrient flows and emissions along the chain by including chain processes, resulting in information on the environmental impacts along the chain (see Chapter 2, Table 2.1). When applying these methods, several studies differ in their definition of system boundary, scale, and inputs and outputs flows included, complicating the comparison of computed indicators. To determine a nutrient balance at farm or food system level, for example, we need to consider all processes and activities, including non-livestock related activities (e.g. vegetable production) that might affect the overall efficiency of the farm or food system (Gourley et al., 2012b; Nevens et al., 2006; Powell et al., 2010). To determine the nutrient balance associated with livestock production, however, we need to disaggregate N losses of the farm or food system and consider only those activities and processes associated with livestock production, including losses associated with, for example, the production of imported feed (Godinot et al., 2014).

Recently, several researchers have developed approaches to analyse nutrient flows and NUE of livestock supply chains (Godinot et al., 2014; Mu et al., 2016) or entire food systems of a country (Erisman et al., 2008; Suh and Yee, 2011). These novel approaches differ in their concepts and objectives. The first focus on cradle to farm-gate analysis of livestock supply chains to compute NB and NUE , by including upstream N flows and emissions but excluding post-farm gate processing. The latter focus on a geographical unit, consider import, and export flows without fully accounting for potential nutrient emissions related to their production.

We built on the existing methods to develop a comprehensive framework of indicators to assess nutrient use along the livestock supply chain. The framework developed in Chapter 3 combines several features of NUE , NB and LCA, and is based on the life cycle approach to encompass all processes and stages along the chain. This framework accounts for detailed nutrient flows and emissions in three stages of livestock supply chains, namely feed production, animal production and post-farm gate processing. The system boundary was defined to span from “cradle-to-primary-processing-stage”, for which data is available. Table 2.1 in Chapter 2 shows the differences between the framework developed and the existing methods. The benefit of this framework is its capability to calculate in detail three combined indicators, providing a comprehensive understanding of agricultural efficiency and environmental pressures associated with nutrient flows in order to support tailored improvement interventions along the chain. The first indicator is the life cycle nutrient use efficiency (life-cycle-NUE, based on Suh and Yee (2011), which defines the efficiency upon which nutrient inputs mobilised from nature or other systems are recovered in final animal products while considering linear and multidimensional processes of nutrient flows in the supply chain. This indicator is computed using supply-and-use matrices (Suh and Yee, 2011) which considers feedback loops such as recycling of crop residues and manure. It is different from “full chain NUE” proposed by Sutton et al. (2013a), which consider the food chain as one entity and is calculated as a ratio between by final outputs divided by the aggregated “new” nutrient mobilised from nature (synthetic fertilizer and biological N fixation). This indicator is slightly different from the nutrient footprint (Leip et al., 2014), which is expressed per unit of product, or from the consumption-focused nutrient footprint, expressed per capita (Leach et al., 2012). A nutrient footprint approach, however, does not make a difference between new “nutrient” and recycled nutrient, which is the case in our framework.

Our second indicator is the life cycle net nutrient balance (life-cycle-NNB), which relates to the quantity of nutrients emissions per unit of land used, which is a good proxy for the environmental pressure. It aggregates all nutrient emissions regardless of the actual geographical location where they take place in the chain. This aggregation, however, can introduce compensation of high nutrient emissions of one process or supply chain stage by another (Mu et al., 2016). To solve this issue, we introduced a third indicator to identify hotspots in the supply chains, the nutrient hotspot index. The nutrient hotspot index refers to the relative distribution of nutrient emissions in the chain and pinpoints the existence of hotspots, where improvement interventions can target.

We tested this the framework for the case study of European mixed dairy systems (Chapter 3). The proposed indicators were found to be complementary and relevant for the benchmarking of the environmental sustainability of nutrient use in the livestock supply chains. This complementarity was tested using correlation analysis. For N, no correlation was found across the combination of three indicators, whereas, for P, there was a relative correlation between Life-cycle-NUE_P and Life-cycle-NNB_P (R^2=0.68) and Life-cycle-NUE_P and NHI_P (R^2=0.49).

By disaggregating NUE and N emissions along the chain, our framework identified the sources of inefficiencies and differences between the emission profiles of feed items used. This disaggregation is of particular relevance to the design of targeted improvement interventions. For instance, by identifying the stage or process in which high N emissions take place, livestock producers can adopt best practices to reduce the emissions such as sourcing efficient feed items or adopting new technology for manure management.

We remarked that the framework developed is data intensive like an LCA or MFA. This finding is different from our original hypothesis that NUE at chain level would require less information (see Chapter 2). Such a detailed analysis of nutrient flows at all stages of the supply chain, nevertheless, is the minimum information needed to fully analyse NUE and get better insights into the sources of inefficiency, identify hotspots of emissions and design targeted intervention options.

To summarise, the developed framework is found to be robust and the three indicators proposed are relevant and needed to describe different aspects of nutrient dynamics along the chain and to support the nutrient management intervention. The development of this framework has revealed some modelling challenges, which are discussed in the next section.

7.3 Modelling challenges

7.3.1 Methodological Developments

The framework developed in Chapter 2 relies on sequential methods to assess nutrient flows in each supply chain stage. They vary from specific Tier-2 methods used for nutrient modelling for feed and animal production to the simple input-output method used for the post-farm gate processing or transport. During the development, the question was: which method can be suitable for accurate nutrient assessment at each stage of the chain? Methodological choices have been found to influence the results of nutrient assessment significantly due to the simplification of nutrient dynamics in a given system or limitations to the knowledge in the flows that are difficult to estimate (Oenema et al., 2003).

For feed production, the goal was to identify a method that could account for nutrient emissions and soil nutrient stock changes at a global scale. Considering nutrient stock changes is essential because long-term positive or negative balances can result in accumulation or depletion of nutrient in the soils, respectively (Gourley et al., 2012a; Özbek and Leip, 2015). The accumulated nutrients can be beneficial for the subsequent crop, in particular through crop rotation or P build-up. In low fertilised soils, changes in nutrient stocks can highly influence nutrient uptake and emissions (Bahr et al., 2015).

For N, the choice of method to model changes in soil stocks and emissions has been identified as a critical step for the representativeness of the results. Out literature review elucidated three methods. The first method (M1) is based on input-output balance and widely used in previous studies and policy and assumes soil N stock change to be 0 ((Eurostat, 2013; OECD, 2001; Schröder and Neeteson, 2008). It results in a nutrient surplus that aggregates nutrient emissions and soil stocks. The second method (M2) relies on detailed information to model N emissions from surface runoff, volatilisation, denitrification/nitrification and leaching and deducts soil N stock change from a mass balance (Britz and Leip, 2009; Velthof et al., 2009). The third method (M3) assumes a range of NUE , which is used in an empirical equation to estimate potential stock change (Özbek and Leip, 2015). A closer look at our results (Chapter 3) revealed that these methods resulted in contrasting NUE N. We found that each of these methods could be used depending on the goal and scope of the study and the data availability. M1 can be used in scoping and screening analysis to get insight into potential nutrient pressures. M2 is detailed and allows to identify the sources of N emissions in soils. M3 can be used for feed crops with reliable information on the range of NUE s, but such required information is currently not available for each feed crop or grass around the world. We recommended using M2 for regional and global nutrient analysis because of its representativeness of N dynamics into the soil, despite the inclusion of uncertainties in N emissions in the estimates for soils nutrient stock changes.

For P, existing methods do not account for sustainable P build-up, in particular for P-deficient or optimum soils. We recommended considering this agronomic aspect by differentiating the unsustainable P accumulation in soils (Wall et al., 2013), which is not needed for soil P build-up; and the sustainable build-up P, which is needed to optimise P uptake from deficient soils. The consideration of this aspect, however, requires detailed data on soil P concentrations, which can be difficult to obtain in many regions. For Europe, we used the comprehensive dataset of soil P profiles for cropland and grassland (Tóth et al., 2013a) to illustrate this concept (Chapter 3). Adding this step, however, can result in additional uncertainties in the soil P stock change estimates.

To estimate N-gas emissions from feed production, we used default IPCC emissions factors (IPCC, 2006). N emissions from synthetic fertilizer and manure application deposited, however, can be significant, in particular when the climate is warm and windy, but these climate factors are not considered in this analysis (Guardia et al., 2017; Hafner et al., 2018; Misselbrook et al., 2005).

For the animal production stage, the method used to estimate the nutrient excreted in the manure has a significant impact on the accuracy of the estimates of NH_3, NO_x, and N_2O and ${NO_3}^-$ emissions from manure management. Most studies have used fixed values for nutrient excretion factors for different animal species and cohorts (Bouwman et al., 2013; Ma et al., 2010; Oenema et al., 2014). Velthof et al. (2015), however, found large discrepancies of N excretion factors for livestock species in the European Union and suggested a harmonisation of these factors to better inform policies on nutrient management. To address this issue and consistently estimate nutrient emissions, we derived nutrient excretion values from a mass balance between nutrient intake and nutrient retained in animal tissues and products using a Tier 2 approach (IPCC, 2006), see Chapters 3, 4, and 5. Then, N emissions were calculated from N excreted using emission factors from IPCC. This approach was found to be uncertain for the estimate of N-gas emissions from manure management because a significant fraction of manure is not mineralised directly. To improve the accuracy, in Chapter 6, we derived N-gas emissions taking place from housing, yards and manure management facilities from the total ammoniacal nitrogen (TAN) flow based on Vonk et al. (2018) and European Environmental Agency guidelines (EEA and CLRTAP, 2016).

Through the development of the framework, we remarked that the selected methods rely on a large number of data and assumptions; therefore, it was essential to analyse the uncertainties in the data by identifying the important input parameters that influence the variance of nutrient flows and nutrient use indicators.

7.3.2 Data quality

The framework developed requires detailed data such as nutrient inputs into soils, herd parameters, climate, emission factors, and manure management, to estimate nutrient flows and three nutrient use indicators. These data are highly variable at the global scale, resulting in large uncertainties due to the differences in geographical representation, time boundaries, technology and completeness (Bretz, 1999; ISO 14044, 2006). Several studies have evidenced the influence of the uncertainties in the data on the variance of the environmental indicators (Groen et al., 2014a; Heijungs, 1996; Oenema et al., 2013). Such uncertainties can limit the direct utilisation of these indicators at country-level, to shape the policy and decision-making (Elduque et al., 2015; Gerber et al., 2013; Groen et al., 2016; Mu et al., 2017).

In Chapter 4, we addressed this challenge by differentiating the important inputs parameters that can be established with high-quality data to improve the accuracy of the results from non-important parameters. We combined existing methods based on global sensitivity analysis in one approach and tested it using the cases studies of mixed cattle dairy systems in the Netherlands and Rwanda. The uncertainty analysis was conducted by comparing the Life-cycle-NUE_N, Life-cycle-NNB_N and NHI_N indicators computed using data from GLEAM and a farm survey. For the Netherlands, results showed no difference between N use indicators computed from GLEAM dataset and the Dutch farm accountancy data network (FADN), whereas, for Rwanda, results of N use indicators were different between GLEAM dataset and field survey data. The reason was that, for the Netherlands, the GLEAM dataset was derived from abundant and reliable national statistics, whereas for Rwanda, the GLEAM dataset was collected from available resources that might not represent current technologies or animal performance. For instance, the average milk production was assumed to be 504 kg cow^{-1} y^{-1} in GLEAM, whereas field survey data gave an average of 4879 kg cow^{-1} y^{-1}.

The findings of the global sensitivity analysis showed that uncertainties of a few important input parameters, such as manure deposited on grasslands, applied synthetic fertilizer, milk production and emission factors could explain most of the variance of the N use indicators. In this Chapter, important parameters in GLEAM dataset were substituted for field survey data, which substantially improved the results of N use indicators for Rwanda. Based on this analysis, efforts were made to substantially improve the data on synthetic fertilizer usage, manure application and deposition, and manure management, based on recent data (Chapter 6).

To sum up, this approach was found to be innovative and robust to identify sensitive input parameters that could be substituted with high-quality data to enhance the accuracy of N use indicators. The power of this approach was that it detects the important data to be improved irrespectively of the bias and errors in the space of non-important data. This approach can be used for any environmental assessment such as LCA indicators computed from global datasets such as the ecoinvent (Frischknecht et al., 2005).

7.4 Impacts of livestock supply chains on N cycle

The framework of indicators was applied to assess the magnitude of N flows and emissions, and related indicators in the global livestock supply chains (Chapter 6) and, in more detail, for pork supply chains (Chapter 5). The results showed that, globally, livestock supply chains were responsible for around 65 Tg N y^{-1} of the human-induced N emissions. Most of these emissions take place in feed production and manure management, primarily in forms of NO_3^-, NH_3, NOx and N_2O. The production of milk, meat and their related co-products such as hides and skins from ruminants contributed 70% to total N emissions from livestock systems, while the production of eggs and meat from chicken and pork contribute the remaining 30%. However, we must be cognisant of the role of ruminants in using marginal land and converting nutrients in grass and crop residues into animal-source foods, thus contributing to food security at the expense of this lower efficiency. This analysis revealed that, in absolute terms, most of the ruminant emissions were related to cattle and buffalo production located in South Asia, whereas those for monogastrics were concentrated in East and Southeast Asia.

The spatial analysis evidenced the presence of hotspots of N emissions per unit of land required to produce feed. These hotspots were firmly related to the concentration of the animals and the sources of animal feed, which in turn are closely related to the density of human population. For instance, large geographical concentrations of NH_3 and NO_3^- found in Indo-Gangetic plain were related to high densities of cattle and buffalo; and associated with high feed crop fertilisation, poor manure management and manure usage as fuel. High NH_3 and N_2O emissions in Western Europe and North America were related to industrial pig, chicken and cattle (beef and dairy) production. For Latin America and the Caribbean, and Sub-Saharan Africa, hotspots were related to backyard pig and chicken systems that relied mostly on swill and scavenging with low land requirements. Thus, expressing N emissions from the daily spreading of manure per unit of land revealed hotspots. Overall, we found that mixed systems (dairy cattle, beef cattle and buffalo milk) were responsible for 40% of livestock-induced emissions. Two regions were identified as the most significant contributors to N emissions: South Asia, East and Southeast Asia because of high population density managed by millions of smallholders' farmers.

We found a significant variability of nutrient use indicators within production systems, which suggests that there may be ample space for improvement. The difference in feed fertilisation, livestock management, genetics, feed resources, and digestibility explained most of these differences. We found that broiler chicken systems had the highest Life-cycle-NUE$_N$ and relatively lowest Life-cycle-NNB$_N$. This performance was related to the fact that their production practices are standardised with a low diversity of genetic material, animal husbandry and feeding of high-quality feed, such as cereals that are edible for humans. This finding implied a trade-off between NUE and feed-food competition (van Zanten et al., 2018), but the latter is not yet considered in this framework.

The drivers of the variability of N use indicators differed from a system to another. For instance, the low variability of industrial pig and broiler chicken is related to small differences in management practices between producers due to the standardisation of the production practices, feeding strategies, genetics and veterinary cares. For ruminants, this variability is related to low production units practised by millions of smallholder farmers around the world, in which ruminants provide additional production functions such as local food security, capital insurance, assets, draught power and social services (Weiler et al., 2014) at the expense of low N use efficiency. For Life-cycle-NNB$_N$, we evidenced that the variability is due to the land required to produce feed as well as the magnitude of the emissions. This indicator tended to be lower for small ruminant systems, for instance, because these commonly utilise vast areas of marginal land for grazing.

A closer investigation of pork supply chains in Chapter 5 showed that N emissions into the environment amount to around 14.7 Tg N y^{-1}. More than half of these emissions took place in the backyard, but this system contributed only 27% to total pork production. Industrial systems emitted 23% of total N emissions but contributed significant to pork production (56%). Intermediate systems contributed around 19% to both pork production and N emissions. We found that most of N emissions were in the form of NO_3^- and organic N to surface and groundwater with large implications for aquatic eutrophication. This finding was related to the significant quantity of manure lost, through unregulated disposal, into surface water, e.g. in East and Southeast Asia or to runoff and leaching of manure applied to the land. While the emissions of N_2O s were low (see Chapter 5 and 6), their implications on climate change are substantial, given the high value of global warming potential of this gas. Backyard and intermediate systems with a high connectivity between animal production and cropland had lower Life-cycle-NNB_N than industrial systems, except in East and Southeast Asia. Regions with high Life-cycle-NUE_N were found across all production systems, implying that in each system, there was a potential to improve N use indicators through the adoption of best practices in feeding, manure management, fertilizer use and animal management.

We emphasised that global livestock supply chains are playing a role in the transfer of soil fertility from grassland to arable land. We estimated that around 3 Tg N are transferred each year from grasslands to arable lands by grazing ruminants, and this transfer is around 1.5 Tg N from croplands to grassland through manure application. While this transfer is beneficial for crop production, it can result in the degradation of the grasslands, as demonstrated for P by Sattari et al. (2016). International trade of feed contributes to the shift of N embodied in feed between countries. For instance, soybeans and soybean cakes shifted around 10 Tg N, mainly from Brazil, Argentina and USA to several importing countries including China, Pakistan and Mexico. In China, for instance, a share of imported N in feed ends up as excreted manure, which is seldom recycled, resulting in unregulated disposals of manure in watercourses (Bai et al., 2014; Strokal et al., 2014). We found that total N emissions from livestock are slightly closer to the planetary boundary for N surplus (Springmann et al., 2018), indicating that the sector, therefore, does not leave virtually no “N allowance” for other agricultural sectors. For the livestock sector, it is relevant to adopt improvement pathways to reduce its impacts on human and ecosystem health.

7.5 Improvement pathways

The magnitude of N emissions and the significant variability of the N use indicators call for improvement pathways to reduce the environmental pressure of livestock supply chains. Interventions can aim to reduce the yield gaps of feed crops, improve grassland productivity through the incorporation of N-fixing legumes, or increase animal productivity, in particular for small-scale systems, which represent the largest share of global livestock population. Increasing productivity, however, would be possible only by combining best practices with other socio-economic solutions to increase household incomes such as employment, education and entrepreneurship.

Interventions can focus on the adoption of best management practices for the manure and fertilizer application in cropland and grassland systems (Gao et al., 2017) and incentives to adopt of low-emission spreading techniques for manure (Sanz-Cobena et al., 2014). Interventions can also target the revision of fertilizer subsidies and other fertilizer pricing policies (Heffer et al., 2017), and collection, transport and recycling of manure to available croplands. Innovative feeding strategies, such as the use of food wastes as feed for pigs (Chapter 5), feeding high-quality feed, use of enzymes to improve feed digestibility or low-protein feed ration, and N retention in animal tissues (Bittman et al., 2014). Feeding strategies can be designed to reduce the dependence on feed that is edible for humans, particularly in regions with food insecurity, and use alternative feed resources such as insects or food wastes and losses (Parodi et al., 2018; Röös et al., 2016; van Zanten et al., 2018). The implementation of these targeted interventions would need more investment, research, and technology transfer, in particular, for the systems managed by millions of smallholder farmers.

Manure management strategies, such as the direct separation of the liquid and solid phases, or recovering NH_3 fertilizer from manure (Dube et al., 2016), can reduce N emissions substantially during storage and spreading on the land. Manure transport to areas that need fertilisation can also reduce the unregulated manure disposal into the environment or the non-collection of manure from the farmyard. We demonstrated that the regulation of manure recycling and management could increase Life-cycle-NUE_N in East and Southeast Asia and in North America (Chapter 6).

While analysing the improvement options, it is relevant to identify potential trade-offs and synergies related to other mitigations strategies for climate change. For instance, supplementing dairy cattle with nitrates to reduce methane (Van Middelaar et al., 2014) can result in high N emissions from manure management. Strategies to reduce NH_3 emissions from manure management can result in the increase of CH4 emissions (Hou et al., 2015). Closing feed crop yield gaps would require the increase of synthetic fertilizer application per unit of land and water use for irrigation, thus resulting in high N emissions (Mueller et al., 2012). Improvement of grassland productivity through synthetic fertilizer or manure application can favour the proliferation of N-depended alien plants, thus damaging biodiversity (Nyfeler et al., 2009). Overall, mitigation strategies addressing a hotspot can result into the shift of N emissions into other stages of the chain, thus this issue must be taken into account while designing mitigation strategies (Hou et al., 2015; Oenema et al., 2014).

In Chapter 5, feeding swill to industrial pigs was evaluated as a global strategy to reduce N emissions from the pig sector and integrate better the livestock in circular bio-economy. The results evidenced that, the substitution of swill for grains and soybeans could improve N use indicators and abate N emissions substantially from 11 to 56%and simultaneously increase Life-cycle-NUE_N from 6% to 30%. It had co-benefits of reducing the livestock pressures on land use by saving 16 M ha of arable lands. This analysis indicated that swill feeding has potential to provide vast mitigation by reducing N emissions, land use and waste streams. Implementing swill feeding, however, would require innovative policies to guide the collection, treatment, and usage of swill, and ensure safety and traceability. Feeding swill, however, is banned in many countries due to the high risks of infectious diseases, but the example from Japan and South Korea showed that swill feeding could be regulated and controlled to limit these risks (zu Ermgassen et al., 2016). For instance, educating households on the separation of food wastes from other wastes, investing in manufacturing units that can collect, treat and process food wastes into animal feed is crucial (Liu et al., 2016).

The sole improvement of NUE, however, may not always translate into a reduction of total N emissions, because enhancing efficiency can cause a rebound effect: as efficiency increases, production costs may drop, resulting in a consumption surge. The overall result, despite reductions in emission per unit of product, may thus be an increase of absolute N emissions. This issue calls for approaches combining both efficiency gains and control of the sector's expansion, e.g. through dietary changes (not addressed in this thesis).

7.6 Nitrogen and sustainability

Addressing nutrient flows and emissions from livestock supply chains is an essential part of improving the sustainability of the livestock sector, and its contribution to the Sustainable Development Goals (FAO, 2018c). Nutrient pollution, nevertheless, is far from being the only sustainability challenge and sustainable development opportunity facing livestock supply chains.

The production of animal-sourced food relies on the use of a large number of natural resources, often with low efficiency. The livestock sector and requires 40% of global arable lands to produce feed crop (Mottet et al., 2017). The sector is also a significant user of freshwater that exacerbates the competition with other agricultural or human activities around the world (Poore and Nemecek, 2018). Concurrently, the sector contributes about 14.5% of global human-induced greenhouse emissions (Gerber et al., 2013). Considering livestock as a component of the food systems, Poore and Nemecek (2018) found that global food systems are responsible for about 32% of global terrestrial acidification and 78% of eutrophication. These environmental issues have fed the debate on the mitigation options as well as on the role of livestock in sustainable food systems.

Similarly, livestock supply chains are embedded in the economy and culture of societies. They thus contribute to rural development, human diets, trade balances, risk management and other relevant development outcomes. According to FAO (2018c), Livestock can help to build resilience and offer several options for the adaptation to climate change, while supporting the increase of children's cognitive development through balanced diets with high-quality animal-source products. Livestock can also negatively affect these outcomes, e.g. contributing to public health issues (diets, zoonoses, Anti-microbial resistance), and offering poor conditions to livestock producers and animals themselves. In some countries where livestock are integrated into cities, livestock can cause environmental and health threat due to poor manure management. Livestock can also affect gender in rural areas, where women and girls have limited access to resources, thus keeping them into poverty.

Addressing these issues requires research drawing on multi-criteria analysis to identify improvement pathways that can offer benefits on the environmental point of view while considering socio-economic dimensions and local realities. These improvement pathways would avoid potential adverse effects on livelihoods and maintain the resilience of the livestock farmers.

Several studies have analysed scenarios to improve the environmental sustainability of the livestock sector. Closing the productivity gaps and adopting best practices can reduce sectoral greenhouse gas emissions by 30% (Gerber et al., 2013). Producing animal-source food from food wastes, crop residues and grass resources would reduce the arable lands used for feed production by 25% (van Zanten et al., 2018), GHG emissions by 18% and N surpluses by 46% (Schader et al., 2015), but at the expense of the size of the livestock sector (Ripple et al., 2013). Other scenarios explored the abatement of environmental impacts of the livestock through dietary changes (Poore and Nemecek, 2018; Springmann et al., 2018; Willett et al., 2019), especially in countries with high consumption of the animal-source foods. Other studies have looked at options to improve other dimensions of sustainability such as the maintenance of the biodiversity and valorisation of ecosystem services provided by livestock, restauration of degraded land, and providing equal access to international markets to small-scale farmers (Brockhaus and Djoudi, 2008; Nori and Gemini, 2011; FAO, 2018c; World Bank, 2009).

Given the multiple dimensions of sustainability and the necessary combination of both objective metrics and value judgements, increasing the sustainability of the livestock sector requires international collaborations involving multiple livestock stakeholders, e.g. from the private sector, farmers organisations, civil society and policy-makers. One example of such a multi-stakeholder initiative is the Livestock Environmental Assessment and Performance (LEAP) Partnership, which aims to build scientific consensus on the methods and indicators to assess the environmental performance of the livestock supply chains. This thesis has contributed directly to the work of LEAP Partnership, which recommended the Life-cycle-NUE_N indicator for measuring the efficiency along the livestock supply chains (FAO, 2018b).

7.7 Conclusions

This thesis develops a comprehensive framework of nutrient use indicators and evaluates the impacts of the livestock supply chains on nitrogen flows. The framework developed incorporates the life cycle approach in nutrient use efficiency and allows for the identification of sources of inefficiencies along supply chains. It proposes three indicators: Life-cycle-NUE_N, Life-cycle-NNB_N and NHI_N that are required to comprehensively describe nutrient flows and emissions in livestock supply chains. The developed indicators are suitable, and their combination gives complementary information to concisely benchmark and monitor nutrient management performance of the entire supply chain.

Given that this framework is data hungry, we proposed a method to select important input parameters that need to be established with high-quality data to improve the accuracy of the results. This method reveals that a few input parameters such as manure deposited and applied, synthetic fertilizer usage, milk production and emission factors explains most of the variance of N use indicators. Establishing these input parameters with field survey data improves the accuracy of N use indicators. This method can be used for any environmental modelling assessment using globally available datasets to improve the accuracy of the estimates.

Our analysis of N emissions and three N use indicators for the global livestock supply chains reveals that the sector is responsible for 65 Tg N y^{-1} of total human-induced N emissions, of which 63% takes place in two regions, and 61% in the feed production stage. The significant variability of N use indicators between and within systems indicates that good practices are available to improve the performance. These findings imply that there is opportunity to design targeted mitigation interventions.

The global analysis shows that improved genetics, animal husbandry, and veterinary care in broiler chicken systems have paid off and resulted in high Life-cycle-NUE$_N$ and low Life-cycle-NNB$_N$ as compared to other production systems, but that such levels of efficiency are achieved at the expense of genetic variability and increased feed-food competition. However, the transferability of the broiler chicken model, to ruminant or small-scale monogastric systems may be limited. Ruminants convert mostly forage, crop residues and industrial by-products into valuable animal-sourced food at the expenses of low N use efficiency.

A detailed evaluation of the impact of global pig supply chains allowed the exploration of the effects of partially replacing grains and soybeans with swill for industrial pigs on N emissions and N use indicators. This substitution significantly reduces N losses at the feed production stage and improves the Life-cycle-NUE$_N$, but constrains the total amount of pork produced due to the limited availability of swill. This finding is of direct relevance to policy or decision makers interested in improving the sustainability of N management in pig supply chains. Policies focusing the swill collection, transport and treatment by manufacturers, retailers and farmers would be necessary to reduce potential health risks and feed safety related to swill feeding.

We propose improvement pathways focusing on the main sources of N emissions, including feed production, and manure management through the adoption of innovative technology and best practices. These improvement pathways can be effective because N emissions are concentrated in few regions, supply chains and steps along the chain and the wide variability of N use indicators offers opportunity to design mitigation interventions. The adoption of good practices would likely require additional investments, knowledge transfer and additional solutions to improve simultaneously the socio-economic conditions of farmers worldwide. Addressing N challenges will require the consideration of potential trade-offs and synergies with other sustainability dimensions such as climate change or socio-economic aspects.

Appendices

Appendix A

Supplementary material: Chapter 3

Table A.1: Supply-and-use matrix construction

		Product			Process					
		Crop/ Pasture	Animal products	Animal End-products*	crop production	Animal production	Processing	Final consumption	Export	Total
product	Crop/pasture				Crop residues	feed intake	0	0	Food crop	A
	Animal products				manure recycled	0	live animals and raw products	0	Exported animal or manure	B
	End-products				0	0	0	animal end-products	0	C
					INP[1]					
process	crop production	crop and pasture harvested, crop residues								
	Animal production		Manure recycle, live animals/ products							
	Processing			Processed animal products						
		PROD[2]								
import	Crop/pasture				0	0	0			
	Animal products				0	0	0			
	End-products				0	0	0			
					IMP[3]					
	Resource mobilisation				Resources mobilised	0	0			
					RES[4]					
	Change in stock				Stock Change	Stock Change	Stock Change			
					-SC[5]					
	Waste generation				Nutrient Losses	Nutrient Losses	Nutrient Losses			
					NNB[6]					
	Total	A	B	C	A	B	C			

[1] INP: Matrix of aggregated inputs to each stage, [2] PROD: Matrix of products of each stage, [3] IMP: Matrix of imported products, applied as inputs to each stage, [4] RES: Matrix of resources mobilised from the nature or other agricultural activities such as biological N fixation (BNF), synthetic fertiliser, atmospheric deposition, manure from other animal species, other organic fertiliser. [5] SC: Matrix of stock change at stage
[6]NNB: Matrix of nutrient losses at each stage. *Animal end-products: edible and non-edible products. The letters A, B, C represent the total nutrient at each stage (A: crop production, B: animal production, C: processing)

Appendix B

Supplementary material: Chapter 5

B.1 Country grouping

In this chapter, countries and territories are grouped in 10 regions based on the FAO Global Administrative Unit Layers (GAUL). GAUL can be found at: http://www.fao.org/geonetwork/srv/en/metadata.show?id=12691.

LATIN AMERICA AND THE CARIBBEAN (LAC): Anguilla, Antigua and Barbuda, Argentina, Aruba, Bahamas, Barbados, Belize, Bolivia, Brazil, British Virgin Islands, Cayman Islands, Chile, Colombia, Costa Rica, Cuba, Dominica, Dominican Republic, Ecuador, El Salvador, Falkland Islands (Malvinas), French Guiana, Grenada, Guadeloupe, Guatemala, Guyana, Haiti, Honduras, Jamaica, Martinique, Mexico, Montserrat, Netherlands Antilles, Nicaragua, Panama, Paraguay, Peru, Puerto Rico, Saint Kitts and Nevis, Saint Lucia, Saint Vincent and the Grenadines, Suriname, Trinidad and Tobago, Turks and Caicos Islands, United States Virgin Islands, Uruguay, Venezuela.

SUB-SAHARAN AFRICA (SSA): Angola Benin Botswana Burkina Faso Burundi Côte d'Ivoire Cameroon Cape Verde, Central African Republic, Chad, Comoros, Congo, Democratic Republic of the Congo, Djibouti, Equatorial Guinea, Eritrea, Ethiopia, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Kenya, Lesotho, Liberia, Madagascar, Malawi, Mali, Mauritania, Mauritius, Mayotte, Mozambique, Namibia, Niger, Nigeria, Rwanda, Reunion, Saint Helena, Sao Tome and Principe, Senegal, Seychelles, Sierra Leone, Somalia, South Africa, Swaziland, Togo, Uganda, United Republic of Tanzania, Zambia, Zimbabwe.

NEAR EAST AND NORTH AFRICA (NENA): Algeria, Armenia, Azerbaijan, Bahrain, Cyprus, Egypt, Gaza Strip, Georgia, Iraq, Israel, Jordan, Kazakhstan, Kuwait, Kyrgyzstan, Lebanon, Morocco, Oman, Qatar, Republic of Sudan, Saudi Arabia, South Sudan, State of Libya, Syrian Arab Republic, Tajikistan, Tunisia, Turkey, Turkmenistan, United Arab Emirates, Uzbekistan, West Bank, Western Sahara, Yemen, SOUTH ASIA: Afghanistan, Bangladesh, Bhutan, British Indian Ocean Territory, India, Iran (Islamic Republic of), Maldives, Nepal, Pakistan, Sri Lanka.

EASTERN EUROPE:, Belarus, Bulgaria, Czech Republic, Hungary, Moldova, Republic of Poland, Romania, Slovakia, Ukraine

RUSSIAN FEDERATION: Russian Federation

EAST AND SOUTHEAST ASIA Brunei Darussalam, Cambodia, China, Christmas Island, Democratic People's Republic of Korea, Hong Kong, Indonesia, Japan, Lao People's Democratic Republic, Macau, Malaysia, Mongolia, Myanmar, Philippines, Republic of Korea, Singapore, Thailand, Timor-Leste, Viet Nam.

OCEANIA: American Samoa, Australia, Cook Islands, Fiji, French Polynesia, Guam, Kiribati, Marshall Islands, Micronesia (Federated States of), Nauru, New Caledonia, New Zealand, Niue, Norfolk Island, Northern Mariana Islands, Palau, Papua New Guinea, Pitcairn, Saint Pierre et Miquelon, Samoa, Solomon Islands Tokelau, Tonga, Tuvalu, Vanuatu, Wake Island, Wallis and Futuna.

WESTERN EUROPE Albania, Andorra, Austria, Belgium, Bosnia and Herzegovina, Croatia, Denmark, Estonia, Faroe Islands, Finland, France, Germany, Greece, Guernsey, Iceland, Ireland, Isle of Man, Italy, Jersey, Latvia, Liechtenstein, Lithuania, Luxembourg, Madeira Islands, Malta, Monaco, Montenegro, Netherlands, Norway, Portugal, Republic of Serbia, San Marino, Slovenia, Spain, Svalbard and Jan Mayen Islands, Sweden, Switzerland, The former Yugoslav Republic of Macedonia, United Kingdom of Great Britain and Northern Ireland.

NORTH AMERICA: Bermuda, Canada, Greenland, United States of America.

B.2 Methods

B.2.1 Scenario analysis: calculation of swill substitution

To estimate the amount of swill that can substitute grains and soybean, an approach that combines the conservation of gross energy and N supply during the substitution is used as follows:

$\mu E_s = \rho E_g + \omega E_{so}$ Eq. (1)

$\mu N_s = \rho N_g + \omega N_{so}$ Eq. (2)

where μ,ρ and ω refer to amounts of swill, grains and soybean (kg DM). E_s and N_s refer to average gross energy and N content for swill, E_g and N_g refer to average gross energy and N contents for grains; E_{so} and N_{so} refer to the average gross energy and N contents for soybean products. For 1 kg DM of swill added, the amount of grains and soybeans substituted ρ and ω are calculated based on the combination of Eq. (1) and Eq. (2) as follows:

Table B.1: Description of feed materials considered for pig rations

Number	Materials	Description
Swill and scavenging		
1	Swill	Household food waste and other organic material used as feed.
Locally-produced feed materials (on-farm and by-products) a		
2	Fresh Grass	Any natural or cultivated fresh grass fed to the animals.
3	Pulses	Leguminous beans.
4	Straw	Fibrous residual plant material such as straw, from leguminous plants cultivation.
5	Cassava	Pellets from cassava (*Manihot esculenta*) roots.
6	Wheat	Grains from wheat (*Triticum aestivum*).
7	Maize	Grains from maize (*Zea mays*).
8	Barley	Grains from barley (*Hordeum vulgare*).
9	Millet	Grains from millet (*P. glaucum, E. coracana, P. miliaceum...*).
10	Rice	Grains from rice (*Oryza sp.*).
11	Sorghum	Grains from sorghum (*Sorghum sp.*).
12	Soy	Beans from soy (*Glycine max*).
13	Tops	Fibrous residual from sugarcane (*Saccharum spp.*)
14	Leaves	Leaves from natural vegetation found in trees.
15	Banana fruit	Fruit from banana trees (*Musa sp.*)
16	Banana Stems	Residual plant material such as stems from banana (*Musa sp.*)
17	Soybean meal	By-product from soy (*Glycine max*) referred to as 'soy cakes' or 'soybean meal'.
18	Cotton seed meal	By-product from cottonseeds (*Gossypium sp*) referred to as 'cottonseeds cakes'.
19	Other oil seed mean	By-product from oil production other than soy, cottonseed or palm oil.
20	Dry by-product grain industries	'Dry' by-products of grain industries such as brans, middlings, etc.
Non-local produced feed materials (imported) b		
21	Pulses	Beans.
22	Cassava	Pellets from cassava (*Manihot esculenta*)
23	Wheat	Grains from wheat (*Triticum aestivum*).
24	Maize	Grains from maize (*Zea mays*).
25	Barley	Grains from barley (*Hordeum vulgare*).
26	Millet	Grains from millet (*P. glaucum, E. coracana, P. miliaceum...*).
27	Rice	Grains from rice (*Oryza sp.*).
28	Sorghum	Grains from sorghum (*Sorghum sp.*).
29	Soybean	Beans from soy (*Glycine max*).
30	Rapeseed	Seeds from rape (*B. napus*).
31	Soy oil	Oil extracted from soybeans (*Glicyne max*).
32	Soybean meal	By-product from soy (*Glycine max*) referred to as 'soy cakes' or 'soybean meal'.
33	Cotton seed meal	By-product from cottonseeds (*Gossypium sp*) referred to as cottonseeds cakes.
34	Rapeseed meal	By-products from rape oil production, referred to as 'canola cakes'.
35	Palm cake	By-products from the production of kernel palm oil (*Elaeis guineensis*),
36	Other oil seed meal	By-product from oil production other than soy, cottonseed, rapeseed or palm oil.
37	Fishmeal	By-products from the fish industries.
38	Molasses	By-product from the sugarcane sugar extraction.
39	Distilleries Dried Grain	'Dry' by-products of grain industries from distilleries.
40	Distilleries wet Grain	'Wet' by-products of grain industries such as biofuels, distilleries, breweries.
41	Supplement: Amino acids, minerals, vitamins	Protein-rich supplement such as amino-acids or minerals.
a Feeds that are produced locally and used extensively in intermediate and backyard systems.		
b Feed materials that are purchased from international markets.		

Table B.2: Average feed ration composition and nutritional values for backyard system in percentage of the ration by region

Feed materials	South Asia	ESEA[1]	Western Europe	Eastern Europe	Oceania	LAC[2]	Russian Federation	Sub-Saharan Africa	NENA[3]
Locally-produced feed materials (on-farm and by-products)									
Fresh Grass	-	-	-	-	2	0	-	-	-
Swill	20	18	11	16	17	20	20	20	20
Pulses	1	1	1	0	2	1	1	2	1
Straw	9	6	6	3	13	7	5	22	9
Cassava	1	2	-	-	3	2	-	10	-
Wheat	3	5	18	17	5	1	21	1	24
Maize	3	7	19	13	13	6	2	8	4
Barley	0	0	7	7	3	0	6	0	9
Millet	0	0	0	0	-	-	0	2	0
Rice	30	24	1	0	1	5	0	6	1
Sorghum	0	0	0	-	0	1	0	3	0
Soy	0	0	1	0	0	1	1	0	0
Tops	8	10	0	-	23	37	-	10	2
Leaves	-	-	-	-	3	1	-	-	-
Banana fruit	0	0	0	-	0	0	-	0	0
Banana Stems	0	0	0	-	0	0	-	0	0
Soybean meal	5	7	7	5	0	9	10	4	1
Cotton seed meal	0	0	4	18	0	0	18	1	2
Other oil seed mean	1	0	2	0	0	0	0	2	6
DDG[4]	18	18	22	18	11	7	15	10	19
Non-local feed materials (off-farm and imported)									
Other oil seed meal	0	0	1	1	1	1	1	0	0
Cotton seed meal	0	0	1	1	1	1	1	0	0
Nutritional values									
GE[5] (MJ/kg DM)	18.4	18.5	18.8	18.8	18.5	18.6	18.9	18.4	18.6
N (g/kg DM)	23	24	27	32	19	24	35	22	25

[1]Eastern and Southeast Asia
[2]Latin America and the Caribbean
[3]Near East and North Africa
[4]Distillers dried grain
[5]Gross energy

Table B.3: Average feed ration composition and nutritional values for intermediate system in percentage of the ration by region

Feed materials	South Asia	ESEA[1]	Western Europe	Eastern Europe	Oceania	LAC[2]	Russian Federation	Sub-Saharan Africa	NENA[3]
Locally-produced feed materials (on-farm and by-products)									
Fresh Grass	-	-	-	-	1	0	-	-	-
Swill	-	0	2	1	-	0	-	0	-
Pulses	0	0	0	0	1	0	0	1	1
Straw	5	3	3	1	9	5	2	13	6
Cassava	0	-	1	-	2	1	-	7	-
Wheat	2	10	3	6	1	0	8	0	15
Maize	2	8	4	5	6	4	1	6	3
Barley	0	4	0	3	1	0	2	0	6
Millet	0	0	0	0	-	-	0	1	0
Rice	20	0	15	0	1	4	0	4	1
Sorghum	0	0	0	-	0	0	-	2	0
Soy	0	0	0	0	0	1	0	0	0
Tops	5	0	6	-	16	22	-	6	1
Leaves	-	-	-	-	1	0	-	-	-
Banana fruit	0	0	0	-	0	0	-	0	0
Banana Stems	0	0	0	-	0	0	-	0	0
Soybean meal	2	3	4	2	1	7	3	2	0
Cotton seed meal	0	1	0	7	-	0	7	1	1
Other oil seed mean	0	0	0	0	0	0	0	1	4
DDG[4]	12	11	11	7	5	4	6	6	12

Table B.4: Average feed ration composition and nutritional values for intermediate system in percentage of the ration by region (continue)

Feed materials	South Asia	ESEA[1]	Western Europe	Eastern Europe	Oceania	LAC[2]	Russian Federation	Sub-Saharan Africa	NENA[3]
Non-local feed materials (off-farm and imported)									
Pulses	-	2	-	-	-	-	-	-	-
Cassava	6	-	4	-	-	2	-	7	8
Wheat	-	10	1	20	12	11	24	-	-
Maize	10	13	15	19	-	11	14	12	12
Barley	-	8	1	6	9	-	7	-	-
Millet	-	-	-	-	-	-	-	10	3
Rice	10	-	10	-	-	2	-	5	9
Sorghum	3	-	1	-	22	1	-	8	2
Soybean	1	-	1	-	-	5	-	-	1
Soybean meal	12	11	10	10	10	12	11	3	10
Oil seed meal	-	6	-	7	-	-	7	-	-
Cotton seed meal	1	-	-	-	-	-	-	-	-
Palm cake	-	-	1	-	-	-	-	-	-
Fishmeal	2	0	2	2	-	2	3	2	2
Molasses	-	2	1	-	-	0	-	2	1
DDG[4]	2	3	2	4	-	2	4	-	2
Supplement[5]	1	1	1	1	1	1	1	1	1
Nutritional values									
GE[6] (MJ/kg DM)	18.6	18.6	18.8	18.9	18.7	19.0	18.9	18.3	18.7
N (g/kg DM)	28	27	30	33	21	31	34	20	27

[1]Eastern and Southeast Asia
[2]Latin America and the Caribbean
[3]Near East and North Africa
[4]Distillers dried grain
[5]Amino acids, vitamins
[6]Gross energy

Table B.5: Pork production by system and region in kt y^{-1} for 2010

Regions	Backyard	Intermediate	Industrial	Total
South Asia	604	252	101	956
North America	-	-	13,268	13,268
Western Europe	186	288	21,637	22,111
ESEA[1]	24,235	15,901	16,077	56,213
Eastern Europe	941	1,185	3,309	5,434
Oceania	191	12	392	596
LAC[2]	2,767	1,827	6,093	10,688
Russian Federation	589	111	1,679	2,379
Sub-Saharan Africa	1,010	332	427	1,769
NENA[3]	48	48	227	323
World	30,570	19,956	63,211	113,737

[1]Eastern and Southeast Asia
[2]Latin America and the Caribbean
[3]Near East and North Africa

Table B.6: Nitrogen use efficiency at each stage of the pork supply chain by region and system expressed in percentage.

Regions	**Feed production**			**Animal production**			**Processing**		
	BACK[4]	**INTER[5]**	**INDU[4]**	**BACK**	**INTER**	**INDU**	**BACK**	**INTER**	**INDU**
South Asia	53	49	54	74	78	77	93	93	93
North America			73			91			93
Western Europe	76	74	68	80	79	85	93	93	93
ESEA[1]	50	55	71	62	67	75	93	93	93
Eastern Europe	62	68	69	79	77	80	93	93	93
Oceania	68	60	65	83	83	83	93	93	93
LAC[2]	74	78	80	71	74	80	93	93	93
Russian Federation	69	71	65	78	74	77	93	93	93
Sub-Saharan Africa	77	57	40	74	82	92	93	93	93
NENA[3]	61	66	68	73	77	78	93	93	93
Global	**52**	**57**	**70**	**64**	**69**	**82**	**93**	**93**	**93**

[1]Eastern and Southeast Asia
[2]Latin America and the Caribbean
[3]Near East and North Africa
[4]Backyard system
[5]Intermediate system
[6]Industrial system

Table B.7: Nitrogen use indicators for pork supply chains by region

	Life-cycle-NUE$_N$ (%)			**Life-cycle-NNB$_N$ (kg N ha^{-1})**			**NHI$_N$ (%)**		
Regions	**BACK**[4]	**INTER**[5]	**INDU**[6]	**BACK**	**INTER**	**INDU**	**BACK**	**INTER**	**INDU**
South Asia	39	37	42	51	51	64	144	149	137
North America			63			36			103
Western Europe	57	54	51	25	32	45	103	100	110
ESEA[1]	30	35	50	104	79	40	125	115	82
Eastern Europe	45	50	52	28	32	34	136	111	102
Oceania	55	45	51	15	20	38	74	118	108
LAC[2]	50	53	60	27	31	30	85	80	74
Russian Federation	50	49	48	14	22	34	125	98	101
Sub-Saharan Africa	54	45	39	10	18	33	89	133	158
NENA[3]	44	48	52	18	18	41	129	119	105

[1]Eastern and Southeast Asia
[2]Latin America and the Caribbean
[3]Near East and North Africa
[4]Backyard system
[5]Intermediate system
[6]Industrial system

Table B.8: Change in nitrogen use indicators and N losses at animal production stage between baseline and scenario for global industrial pig supply chains

	Life-cycle-NNB$_N$ (kg N ha^{-1})			NHI$_N$ (%)			N losses in animal production (Gg N y^{-1})		
Regions	Baseline	Scenario	Change	Baseline	Scenario	Change	Baseline	Scenario	Change
South Asia	64	52	-18%	137	110	-20%	2	2	-1%
North America	36	35	-2%	103	87	-16%	90	90	1%
Western Europe	45	43	-5%	110	94	-14%	341	325	-5%
ESEA[1]	40	37	-7%	82	77	-5%	364	359	-1%
Eastern Europe	34	30	-12%	102	86	-16%	63	60	-5%
Oceania	38	40	7%	108	92	-15%	5	5	-1%
LAC[2]	30	35	14%	74	73	-1%	100	99	-1%
Russian Federation	34	26	-22%	101	76	-25%	35	33	-4%
Sub-Saharan Africa	33	31	-5%	158	156	-1%	3	3	0%
NENA[3]	41	40	-2%	105	79	-24%	4	4	-2%

[1]Eastern and Southeast Asia
[2]Latin America and the Caribbean
[3]Near East and North Africa

References

Aillery, M., Gollehon, N., Johansson, R., Key, N., Ribaudo, M., 2006. Managing manure to improve air and water quality. Agric. Nutr. Manage. 12 (2), 14–19.

Alexandratos, N., Bruinsma, J., 2012. World agriculture towards 2030/2050: the 2012 revision. Tech. rep., ESA Working paper Rome, FAO, Rome, Italy.

Anastasiadis, S., Kerr, S., 2013. Mitigation and heterogeneity in management practices on New Zealand dairy farms. Motu Economic and Public Policy Research Motu Working Paper 13-11.

Arriaga, H., Pinto, M., Calsamiglia, S., Merino, P., 2009. Nutritional and management strategies on nitrogen and phosphorus use efficiency of lactating dairy cattle on commercial farms: An environmental perspective. Journal of Dairy Science 92 (1), 204–215.

Ascott, M., Gooddy, D., Wang, L., Stuart, M., Lewis, M., Ward, R., Binley, A. M., 2017. Global patterns of nitrate storage in the vadose zone. Nature Communications 8 (1), 1416.

Azevedo, L. B., Henderson, A. D., van Zelm, R., Jolliet, O., Huijbregts, M. A. J., 2013. Assessing the importance of spatial variability versus model choices in life cycle impact assessment: The case of freshwater eutrophication in europe. Environmental Science & Technology 47 (23), 13565–13570.

Bahr, E., Chamba-Zaragocin, D., Fierro-Jaramillo, N., Witt, A., Makeschin, F., 2015. Modeling of soil nutrient balances, flows and stocks revealed effects of management on soil fertility in south ecuadorian smallholder farming systems. Nutrient Cycling in Agroecosystems 101 (1), 55–82.

Bai, Z., Lee, M. R., Ma, L., Ledgard, S., Oenema, O., Velthof, G. L., Ma, W., Guo, M., Zhao, Z., Wei, S., 2018a. Global environmental costs of China's thirst for milk. Global Change Biology 24 (5), 2198–2211.

Bai, Z., Ma, L., Jin, S., Ma, W., Velthof, G. L., Oenema, O., Liu, L., Chadwick, D., Zhang, F., 2016. Nitrogen, phosphorus, and potassium flows through the manure management chain in china. Environmental Science & Technology 50 (24), 13409–13418.

Bai, Z., Ma, L., Qin, W., Chen, Q., Oenema, O., Zhang, F., 2014. Changes in pig production in china and their effects on nitrogen and phosphorus use and losses. Environmental Science & Technology 48 (21), 12742–12749.

Bai, Z., Ma, W., Ma, L., Velthof, G. L., Wei, Z., Havlík, P., Oenema, O., Lee, M. R., Zhang, F., 2018b. China's livestock transition: Driving forces, impacts, and consequences. Science Advances 4 (7), eaar8534.

Bannink, A., Šebek, L., Dijkstra, J., 2010. Phosphorus and calcium utilization and requirements in farm animals. In: Phosphorus and Calcium Utilization and Requirements in Farm Animals, cabi Edition. Vitti, D.M.S.S., Kebreab, E., p. 192.

Bassanino, M., Grignani, C., Sacco, D., Allisiardi, E., 2007. Nitrogen balances at the crop and farm-gate scale in livestock farms in Italy. Agriculture, Ecosystems & Environment 122 (3), 282–294.

Basset-Mens, C., Kelliher, F. M., Ledgard, S., Cox, N., 2009. Uncertainty of global warming potential for milk production on a New Zealand farm and implications for decision making. The International Journal of Life Cycle Assessment 14 (7), 630–638.

Batjes, N., 2011. Global distribution of soil phosphorus retention potential. Wageningen, ISRIC-World Soil Information (with dataset), ISRIC Report 6, 42.

Beausang, C., Hall, C., Toma, L., 2017. Food waste and losses in primary production: Qualitative insights from horticulture. Resources, Conservation and Recycling 126, 177–185.

Beig, G., Maji, S., Panicker, A., Sahu, S., 2017. 25 - reactive nitrogen and air quality in india. In: Adhya, T. K., Aneja, V. P., Raghuram, N., Pathak, H., Kulshrestha, U., Sharma, C., Singh, B. (Eds.), The Indian Nitrogen Assessment. Elsevier, pp. 403–426.

Bertoli, S., Goujon, M., Santoni, O., 2016. The CERDI-seadistance database. https://halshs.archives-ouvertes.fr/halshs-01288748/document/, [online: accessed 10-Mars-2018].

Billen, G., Lassaletta, L., Garnier, J., 2014. A biogeochemical view of the global agro-food system: Nitrogen flows associated with protein production, consumption and trade. Global Food Security 3 (3-4), 209–219.

Bioteau, T., Burton, C., Guiziou, F., Martinez, J., 2009. Qualitative assessment of manure management in main livestock production systems and a review of gaseous emissions factors of manure throughout EU27.

Bittman, S., Dedina, M., Howard, C., Oenema, O., Sutton, M., 2014. Options for ammonia mitigation: Guidance from the unece task force on reactive nitrogen.

Bodirsky, B. L., Popp, A., Lotze-Campen, H., Dietrich, J. P., Rolinski, S., Weindl, I., Schmitz, C., Müller, C., Bonsch, M., Humpenöder, F., et al., 2014. Reactive nitrogen requirements to feed the world in 2050 and potential to mitigate nitrogen pollution. Nature Communications 5, 3858.

Borhan, M. S., Capareda, S. C., Mukhtar, S., Faulkner, W. B., McGee, R., Parnell, C. B., 2011. Greenhouse gas emissions from ground level area sources in dairy and cattle feedyard operations. Atmosphere 2 (3), 303–329.

Bouwman, L., Goldewijk, K. K., Van Der Hoek, K. W., Beusen, A. H. W., Van Vuuren, D. P., Willems, J., Rufino, M. C., Stehfest, E., 2013. Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900–2050 period. Proceedings of the National Academy of Sciences 110 (52), 20882–20887.

Brancoli, P., Rousta, K., Bolton, K., 2017. Life cycle assessment of supermarket food waste. Resources, Conservation and Recycling 118, 39–46.

Bretz, R., 1999. SETAC LCA Workgroup: Data availability and data quality. The International Journal of Life Cycle Assessment 4 (1), 15–15.

Britz, W., Leip, A., 2009. Development of marginal emission factors for N losses from agricultural soils with the DNDC–CAPRI meta-model. Agriculture, Ecosystems & Environment 133 (3–4), 267–279.

Brock, C., Hoyer, U., Leithold, G., Hülsbergen, K.-J., 2012. The humus balance model (HU-MOD): a simple tool for the assessment of management change impact on soil organic matter levels in arable soils. Nutrient Cycling in Agroecosystems 92 (3), 239–254.

Brockhaus, M., Djoudi, H., 2008. Adaptation at the interface of forest ecosystem goods and services and livestock production systems in northern mali. Center for International Forestry Research (CIFOR) Information Brief (19).

Brouwer, J., Powell, J., 1998. Increasing nutrient use efficiency in West-African agriculture: the impact of micro-topography on nutrient leaching from cattle and sheep manure. Agriculture, Ecosystems & Environment 71 (1–3), 229–239.

Brunner, P. H., Ma, H.-W., 2009. Substance Flow Analysis. Journal of Industrial Ecology 13 (1), 11–14.

Campolongo, F., Cariboni, J., Saltelli, A., 2007. An effective screening design for sensitivity analysis of large models. Environmental modelling & software 22 (10), 1509–1518.

Canada, S., 2003. Manure storage in canada. farm environmental management in canada.

Canty, A., Ripley, B., 2014. boot: Bootstrap R (S-Plus) Functions. R package version, 1–3.

Carslaw, D. C., Rhys-Tyler, G., 2013. New insights from comprehensive on-road measurements of NOx, NO_2 and NH_3 from vehicle emission remote sensing in London, UK. Atmospheric Environment 81, 339–347.

Cederberg, C., Mattsson, B., 2000. Life cycle assessment of milk production — a comparison of conventional and organic farming. Journal of Cleaner Production 8 (1), 49–60.

Chen, M., Sun, F., Shindo, J., 2016. China's agricultural nitrogen flows in 2011: environmental assessment and management scenarios. Resources, Conservation and Recycling 111, 10–27.

CML, 2003. Cml-ia characterization factors. update information version 4.2. Tech. rep., CML.

Conant, R. T., Berdanier, A. B., Grace, P. R., 2013. Patterns and trends in nitrogen use and nitrogen recovery efficiency in world agriculture. Global Biogeochemical Cycles 27 (2), 558–566.

Conijn, J., Bindraban, P., Schröder, J., Jongschaap, R., 2018. Can our global food system meet food demand within planetary boundaries? Agriculture, Ecosystems & Environment 251, 244–256.

Cooper, J., Carliell-Marquet, C., 2013. A substance flow analysis of phosphorus in the UK food production and consumption system. Resources, Conservation and Recycling 74 (0), 82–100.

Corbala-Robles, L., Sastafiana, W., Volcke, E., Schaubroeck, T., et al., 2018. Life cycle assessment of biological pig manure treatment versus direct land application- a trade-off story. Resources, Conservation and Recycling 131, 86–98.

Cordell, D., Drangert, J.-O., White, S., 2009. The story of phosphorus: Global food security and food for thought. Global Environmental Change 19 (2), 292–305.

Cosenza, A., Mannina, G., Vanrolleghem, P. A., Neumann, M. B., 2013. Global sensitivity analysis in wastewater applications: A comprehensive comparison of different methods. Environmental modelling & software 49, 40–52.

Cosme, N., Jones, M. C., Cheung, W. W., Larsen, H. F., 2017. Spatial differentiation of marine eutrophication damage indicators based on species density. Ecological indicators 73, 676–685.

Costanza, J. K., Marcinko, S. E., Goewert, A. E., Mitchell, C. E., 2008. Potential geographic distribution of atmospheric nitrogen deposition from intensive livestock production in north carolina, USA. Science of The Total Environment 398 (1), 76–86.

Csathó, P., Radimszky, L., 2012. Sustainable Agricultural NP Turnover in the 27 European Countries. In: Lichtfouse, E. (Ed.), Organic Fertilisation, Soil Quality and Human Health. Vol. 9 of Sustainable Agriculture Reviews. Springer Netherlands, pp. 161–186.

Curran, M. A., 2013. Life Cycle Assessment: a review of the methodology and its application to sustainability. Current Opinion in Chemical Engineering 2 (3), 273–277.

Curran, M. A., 2014. Strengths and limitations of life cycle assessment. In: Klöpffer, W. (Ed.), Background and Future Prospects in Life Cycle Assessment. Springer Netherlands, pp. 189–206.

Dan, T., Hoa, T., Hung, L., Tri, B., Hoa, H., Hien, L., Tri, N., 2003. Area-wide integration (AWI) of specialized crop and livestock activities in vietnam. http://www.fao.org/tempref/docrep/nonfao/lead/x6157e/x6157e00.pdf/, [Online: accessed 2016-10-03].

Daverede, I. C., Kravchenko, A. N., Hoeft, R. G., Nafziger, E. D., Bullock, D. G., Warren, J. J., Gonzini, L. C., 2003. Phosphorus Runoff. Journal of Environmental Quality 32 (4), 1436–1444.

De Boer, I. J., 2003. Environmental impact assessment of conventional and organic milk production. Livestock Production Science 80 (1–2), 69–77.

De Haan, C., Gerber, P., Opio, C., 2010. Structural change in the livestock sector. In: Steinfeld, Henning, H. A. M. S. F. N., Laurie, E. (Eds.), Livestock in a changing landscape. Vol. 1 of Drivers, Consequences and Responses. Island Press, pp. 35–50.

de Oliveira, G. L. T., Schneider, M., 2016. The politics of flexing soybeans: China, brazil and global agro-industrial restructuring. The Journal of Peasant Studies 43 (1), 167–194.

De Vries, M., De Boer, I., 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livestock Science 128 (1–3), 1–11.

de Vries, W., Kros, J., Kroeze, C., Seitzinger, S. P., 2013. Assessing planetary and regional nitrogen boundaries related to food security and adverse environmental impacts. Current Opinion in Environmental Sustainability 5 (3), 392–402.

Dentener, F., 2006. Global maps of atmospheric nitrogen deposition, 1860, 1993, and 2050. Data set. Available on-line (http://daac. ornl. gov/) from Oak Ridge National Laboratory Distributed Active Archive Center, Oak Ridge, TN, USA.

do Carmo Horta, M., Torrent, J., 2007. The olsen p method as an agronomic and environmental test for predicting phosphate release from acid soils. Nutrient Cycling in Agroecosystems 77 (3), 283–292.

Dollé, J.-B., Smati, M., 2005. Les effluents et boues des industries laitières: Procédés de traitement et valorisation agronomique. Tech. rep., Institut de l'Elevage, France.

Dolman, M., Sonneveld, M., Mollenhorst, H., de Boer, I., 2014. Benchmarking the economic, environmental and societal performance of Dutch dairy farms aiming at internal recycling of nutrients. Journal of Cleaner Production 73 (0), 245–252.

Domburg, P., Edwards, A. C., Sinclair, A. H., Chalmers, N. A., 2000. Assessing nitrogen and phosphorus efficiency at farm and catchment scale using nutrient budgets. Journal of the Science of Food and Agriculture 80 (13), 1946–1952.

Doney, S. C., Mahowald, N., Lima, I., Feely, R. A., Mackenzie, F. T., Lamarque, J.-F., Rasch, P. J., 2007. Impact of anthropogenic atmospheric nitrogen and sulfur deposition on ocean acidification and the inorganic carbon system. Proceedings of the National Academy of Sciences 104 (37), 14580.

Dube, P., Vanotti, M., Szogi, A., García-González, M., 2016. Enhancing recovery of ammonia from swine manure anaerobic digester effluent using gas-permeable membrane technology. Waste Management 49, 372–377.

Dumas, M., Frossard, E., Scholz, R. W., 2011. Modeling biogeochemical processes of phosphorus for global food supply. Chemosphere 84 (6), 798–805.

D'Haene, K., Magyar, M., De Neve, S., Pálmai, O., Nagy, J., Németh, T., Hofman, G., 2007. Nitrogen and phosphorus balances of Hungarian farms. European Journal of Agronomy 26 (3), 224–234.

EEA, CLRTAP, 2016. EMEP/EEA air pollutant emission inventory guidebook 2016. https://www.eea.europa.eu/publications/emep-eea-guidebook-2016/, [Online: accessed 2017-01-15].

Elduque, A., Javierre, C., Elduque, D., Fernández, Á., 2015. Lci databases sensitivity analysis of the environmental impact of the injection molding process. Sustainability 7 (4), 3792–3800.

EPA, 2004. National Emission Inventory - Ammonia Emissions from Animal Husbandry Operations. https://www3.epa.gov/ttnchie1/ap42/ch09/related/nh3inventorydraft_jan2004.pdf/, [Online: accessed 2018-01-15].

Erisman, J., Bleeker, A., Galloway, J., Sutton, M. S., 2007. Reduced nitrogen in ecology and the environment. Environmental Pollution 150 (1), 140–149.

Erisman, J. W., Sutton, M. A., Galloway, J., Klimont, Z., Winiwarter, W., 2008. How a century of ammonia synthesis changed the world. Nature Geosci 1 (10), 636–639.

Erisman, J. W., van Grinsven, H., Grizzetti, B., Bouraoui, F., Powlson, D. S., Sutton, M. A., Bleeker, A., Reis, S., 2011. The european nitrogen problem in a global perspective. In: Sutton, M., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H., Grizzetti, B. (Eds.), The European Nitrogen Assessment. Cambridge University Press, pp. 405–433.

Erisman, W. J., Leach, A., Bleeker, A., Atwell, B., Cattaneo, L., Galloway, J., 2018. An integrated approach to a nitrogen use efficiency (NUE) indicator for the food production–consumption chain. Sustainability 10 (4).

European Commission, 2010. ILCD handbook International Reference Life Cycle Data System. Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment: Background document. European Commission Joint Research Centre Institute for Environment and Sustainability.

European Commission, 2015. The end of milk quotas. https://ec.europa.eu/agriculture/milk-quota-end_en.

Eurostat, 2013. Nutient Budgets - Methodology and Handbook. Version 1.0.2. http://ec.europa.eu/eurostat/documents/2393397/2518760/Nutrient_Budgets_Handbook_(CPSA_AE_109)_corrected3.pdf/4a3647de-da73-4d23-b94b-e2b23844dc31.

Eurostat, 2014. Agri-environmental indicator - risk of pollution by phosphorus. http://ec.europa.eu/eurostat/statistics-explained/index.php/{Agri}-environmental_indicator_-_risk_of_pollution_by_phosphorus.

FAO, 2003. Technical conversion factors for agricultural commodities. http://www.fao.org/fileadmin/templates/ess/documents/methodology/tcf.pdf, online: accessed on 2017-01-03.

FAO, 2011a. Energy-Smart food for people and climate. Issue paper, [Online: accessed on 2013-05-07].

FAO, 2011b. Global food losses and food waste–Extent, Causes and Prevention. http://www.fao.org/3/a-i2697e.pdf/, [Online: accessed 10-January-2017].

FAO, 2011. World Livestock 2011. Livestock in food security. http://www.fao.org/docrep/014/i2373e/i2373e.pdf/, [online: accessed 2017-05-26].

FAO, 2016. Environmental performance of large ruminant supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. http://www.fao.org/3/a-i6494e.pdf/, [Online: accessed 15-December-2016].

FAO, 2017. Global livestock environmental assessment model. reference documentation. version 2. http://www.fao.org/fileadmin/user_upload/gleam/docs/GLEAM_Version_2.0_Revision_1.pdf/, [online: accessed on 2017-06-10].

FAO, 2018a. Global Livestock Environmental Assessment Model. Reference documentation. Version 2. hhttp://www.fao.org/fileadmin/user_upload/gleam/docs/GLEAM_2.0_Model_description.pdf/, [Online: accessed on 10-September-2018].

FAO, 2018b. Nutrient flows and associated environmental impacts in livestock supply chains. guidelines for assessment (version 1). http://www.fao.org/3/CA1328EN/ca1328en.pdf/, [Online: accessed 20-October-2018].

FAO, 2018c. World livestock 2018: Transforming the livestock sector through the sustainable development goals. http://www.fao.org/3/CA1201EN/ca1201en.pdf/, [Online: accessed 22-December-2018].

FAOSTAT, 2017. Statistical Databases. Statistics Division. Food and Agriculture Organization of the United Nations. http://faostat3.fao.org/home/E/, [online: accessed 2017-01-10].

FAOSTAT, 2018. Statistical Databases. Statistics Division. Food and Agriculture Organization of the United Nations. http://faostat3.fao.org/home/E/, [online: accessed 2018-01-10].

Fealy, R., Schröder, J. J., 2008. Assessment of manure transport distances and their impact on economic and energy costs. In: Conference Proceedings (642) for the International Fertiliser Society Conference. IFS.

Feedpedia, 2012. Feedpedia. an on-line encyclopedia of animal feeds. Animal feed resources Information system. http://www.feedipedia.org/, [Online: accessed 2014-11-01].

Ferretti, F., Saltelli, A., Tarantola, S., 2016. Trends in sensitivity analysis practice in the last decade. Science of the Total Environment 568, 666–670.

Fischer, G., Nachtergaele, F. O., Prieler, S., Teixeira, E., Tóth, G., Van Velthuizen, H., Verelst, L., Wiberg, D., 2012. Global agro-ecological zones (GAEZ v3. 0)-model documentation. http://www.fao.org/nr/gaez/en/, [Online: accessed 2013-10-10].

Freeman, H., Thornton, P. K., van de Steeg, J. A., Mcleod, A., 2006. Future scenarios of livestock systems in developing countries. Animal production and animal science worldwide. WAAP Book of the Year, 219–232.

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., 2005. The ecoinvent Database: Overview and Methodological Framework (7 pp). The International Journal of Life Cycle Assessment 10 (1), 3–9.

Gale, H. F., Hansen, J., Jewison, M., 2015. China's growing demand for agricultural imports. USDA-ERS Economic Information Bulletin (136).

Galloway, J. N., Aber, J. D., Erisman, J. W., Seitzinger, S. P., Howarth, R. W., Cowling, E. B., Cosby, B. J., 2003. The Nitrogen Cascade. BioScience 53 (4), 341–356.

Galloway, J. N., Burke, M., Bradford, G. E., Naylor, R., Falcon, W., Chapagain, A. K., Gaskell, J. C., McCullough, E., Mooney, H. A., Oleson, K. L., et al., 2007. International trade in meat: The tip of the pork chop. AMBIO: A Journal of the Human Environment 36 (8), 622–629.

Galloway, J. N., Townsend, A. R., Erisman, J. W., Bekunda, M., Cai, Z., Freney, J. R., Martinelli, L. A., Seitzinger, S. P., Sutton, M. A., 2008. Transformation of the Nitrogen Cycle: Recent Trends, Questions, and Potential Solutions. Science 320 (5878), 889–892.

Ganeshamurthy, A., Rupa, T., Kalaivanan, D., Radha, T., 2017. 9 - Nitrogen management paradigm in horticulture systems in India. In: Adhya, T. K., Aneja, V. P., Raghuram, N., Pathak, H., Kulshrestha, U., Sharma, C., Singh, B. (Eds.), The Indian Nitrogen Assessment. Elsevier, pp. 133–147.

Gao, X., Parsonage, S., Tenuta, M., Baron, K., Hanis-Gervais, K., Nelson, A., Tomasiewicz, D., Mohr, R., 2017. Nitrogen fertilizer management practices to reduce N_2O emissions from irrigated processing potato in Manitoba. American Journal of Potato Research 94 (4), 390–402.

Gao, Z., Lin, Z., Yang, Y., Ma, W., Liao, W., Li, J., Cao, Y., Roelcke, M., 2014. Greenhouse gas emissions from the enteric fermentation and manure storage of dairy and beef cattle in China during 1961–2010. Environmental Research 135, 111–119.

Garrone, P., Melacini, M., Perego, A., Sert, S., 2016. Reducing food waste in food manufacturing companies. Journal of cleaner production 137, 1076–1085.

Gerber, J. P., Uwizeye, A., Schulte, P. O. R., Opio, I. C., de Boer, J. M. I., 2014. Nutrient use efficiency: A valuable approach to benchmark the sustainability of nutrient use in global livestock production? Current Opinion in Environmental Sustainability 9–10 (9), 122–130.

Gerber, P., Chilonda, P., Franceschini, G., Menzi, H., 2005. Geographical determinants and environmental implications of livestock production intensification in Asia. Bioresource Technology 96 (2), 263–276.

Gerber, P., Menzi, H., 2006. Nitrogen losses from intensive livestock farming systems in Southeast Asia: A review of current trends and mitigation options. International Congress Series 1293 (0), 253–261.

Gerber, P., Robinson, T., Wassenaar, T., Steinfeld, H., 2010. Livestock in geographical transition. In: Steinfeld, H., Harold, A. M., Fritz, S., Laurie, E. N. (Eds.), Livestock in a changing landscape. Vol. 1 of Drivers, Consequences and Responses. Island Press, pp. 51–66.

Gerber, P., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A., Tempio, G., 2013. Tackling climate change through livestock – a global assessment of emissions and mitigation opportunities. Food and Agriculture Organization of United Nations, Rome, Italy.

Gibbs, H. K., Rausch, L., Munger, J., Schelly, I., Morton, D. C., Noojipady, P., Soares-Filho, B., Barreto, P., Micol, L., Walker, N. F., 2015. Brazil's soy moratorium. Science 347 (6220), 377–378.

Godinot, O., Carof, M., Vertès, F., Leterme, P., 2014. SyNE: An improved indicator to assess nitrogen efficiency of farming systems. Agricultural Systems 127 (0), 41–52.

Gourley, C. J., Aarons, S. R., Hannah, M. C., Awty, I. M., Dougherty, W. J., Burkitt, L. L., 2015. Soil phosphorus, potassium and sulphur excesses, regularities and heterogeneity in grazing-based dairy farms. Agriculture, Ecosystems & Environment 201, 70–82.

Gourley, C. J., Aarons, S. R., Powell, J. M., 2012a. Nitrogen use efficiency and manure management practices in contrasting dairy production systems. Agriculture, Ecosystems & Environment 147 (0), 73–81.

Gourley, C. J. P., Dougherty, W. J., Weaver, D. M., Aarons, S. R., Awty, I. M., Gibson, D. M., Hannah, M. C., Smith, A. P., Peverill, K. I., 2012b. Farm-scale nitrogen, phosphorus, potassium and sulfur balances and use efficiencies on Australian dairy farms. Animal Production Science 52 (10), 929–944.

Groen, E. A., Bokkers, E. A., Heijungs, R., de Boer, I. J., 2017. Methods for global sensitivity analysis in life cycle assessment. The International Journal of Life Cycle Assessment 22 (7), 1125–1137.

Groen, E. A., Heijungs, R., 2017. Ignoring correlation in uncertainty and sensitivity analysis in life cycle assessment: what is the risk? Environmental Impact Assessment Review 62, 98–109.

Groen, E. A., Heijungs, R., Bokkers, E. A., de Boer, I. J. M., 2014a. Sensitivity analysis in life cycle assessment. In: Proceedings of the Life Cycle Assessment Food Conference (LCA Food

2014). pp. 482–488.

Groen, E. A., Heijungs, R., Bokkers, E. A. M., De Boer, I. J. M., 2014b. Methods for uncertainty propagation in life cycle assessment. Environmental Modelling & Software 62, 316–325.

Groen, E. A., van Zanten, H. H. E., Heijungs, R., Bokkers, E. A. M., De Boer, I. J. M., 2016. Sensitivity analysis of greenhouse gas emissions from a pork production chain. Journal of Cleaner Production 129, 202–211.

Guardia, G., Cangani, M. T., Sanz-Cobena, A., Junior, J. L., Vallejo, A., 2017. Management of pig manure to mitigate NO and yield-scaled N_2O emissions in an irrigated mediterranean crop. Agriculture, Ecosystems & Environment 238, 55–66.

Guinée, J. B., Heijungs, R., Huppes, G., Kleijn, R., Koning, A. d., Oers, L. V., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H., Bruijn, H. d., Duin, R. V., Huijbregts, M., 2002. Handbook on life cycle assessment operational guide to the ISO standards. The international journal of life cycle assessment 7 (5), 311–313.

Gupta, P., Jha, A., Koul, S., Sharma, P., Pradhan, V., Gupta, V., Sharma, C., Singh, N., 2007. Methane and nitrous oxide emission from bovine manure management practices in india. Environmental Pollution 146 (1), 219–224.

Hafner, S. D., Pacholski, A., Bittman, S., Burchill, W., Bussink, W., Chantigny, M., Carozzi, M., Génermont, S., Häni, C., Hansen, M. N., et al., 2018. The ALFAM2 database on ammonia emission from field-applied manure: Description and illustrative analysis. Agricultural and Forest Meteorology 258, 66–79.

Halberg, N., van der Werf, H. M., Basset-Mens, C., Dalgaard, R., de Boer, I. J., 2005a. Environmental assessment tools for the evaluation and improvement of European livestock production systems. Livestock Production Science 96 (1), 33–50.

Halberg, N., Verschuur, G., Goodlass, G., 2005b. Farm level environmental indicators; are they useful?: An overview of green accounting systems for European farms. Agriculture, Ecosystems & Environment 105 (1–2), 195–212.

Hamilton, H. A., Ivanova, D., Stadler, K., Merciai, S., Schmidt, J., van Zelm, R., Moran, D., Wood, R., 2018. Trade and the role of non-food commodities for global eutrophication. Nature Sustainability 1 (6), 314.

Harris, I., Jones, P. D., Osborn, T. J., Lister, D. H., 2014. Updated high-resolution grids of monthly climatic observations–the CRU TS3. 10 dataset. International journal of climatology 34 (3), 623–642.

Hart, M. R., Quin, B. F., Nguyen, M., 2004. Phosphorus runoff from agricultural land and direct fertilizer effects. Journal of Environmental Quality 33 (6), 1954–1972.

Hashimoto, S., Moriguchi, Y., 2004. Proposal of six indicators of material cycles for describing society's metabolism: from the viewpoint of material flow analysis. Resources, Conservation and Recycling 40 (3), 185–200.

Haydon, D. T., Kao, R. R., Kitching, R. P., 2004. The UK foot-and-mouth disease outbreak—the aftermath. Nature Reviews Microbiology 2 (8), 675.

Heffer, P., Gruère, A., Roberts, T., 2017. Assessment of fertilizer use by crop at the global level. International Fertilizer Association (IFA) and International Plant Nutrition Institute (IPNI).

Heijungs, R., 1996. Identification of key issues for further investigation in improving the reliability of life-cycle assessments. Journal of Cleaner Production 4 (3-4), 159–166.

Heijungs, R., Kleijn, R., 2001. Numerical approaches towards life cycle interpretation five examples. The International Journal of Life Cycle Assessment 6 (3), 141–148.

Henderson, B., Falcucci, A., Mottet, A., Early, L., Werner, B., Steinfeld, H., Gerber, P., 2017. Marginal costs of abating greenhouse gases in the global ruminant livestock sector. Mitigation and Adaptation Strategies for Global Change 22 (1), 199–224.

Hendriks, C., Kranenburg, R., Kuenen, J., Van den Bril, B., Verguts, V., Schaap, M., 2016. Ammonia emission time profiles based on manure transport data improve ammonia modelling across north western europe. Atmospheric Environment 131, 83–96.

Herrero, M., Thornton, P. K., 2013. Livestock and global change: Emerging issues for sustainable food systems. Proceedings of the National Academy of Sciences 110 (52), 20878–20881.

Herridge, D., Peoples, M., Boddey, R., 2008. Global inputs of biological nitrogen fixation in agricultural systems. Plant Soil 311 (1-2), 1–18.

HLPE, 2014. Food losses and waste in the context of sustainable food systems. a report by the high level panel of experts on food security and nutrition of the committee on world food security. http://www.fao.org/3/a-i3901e.pdf/, [Online: accessed 2016-01-20].

Horst, H., Huirne, R., Dijkhuizen, A., 1997. Risks and economic consequences of introducing classical swine fever into the netherlands by feeding swill to swine. Revue Scientifique et Technique (International Office of Epizootics) 16 (1), 207–214.

Hou, Y., Velthof, G. L., Oenema, O., 2015. Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: a meta-analysis and integrated assessment. Global Change Biology 21 (3), 1293–1312.

Hristov, A., Hazen, W., Ellsworth, J., 2006. Efficiency of Use of Imported Nitrogen, Phosphorus, and Potassium and Potential for Reducing Phosphorus Imports on Idaho Dairy Farms. Journal of Dairy Science 89 (9), 3702–3712.

Huang, J., Xu, J., Xia, Z., Liu, L., Zhang, Y., Li, J., Lan, G., Qi, Y., Kamon, M., Sun, X., et al., 2015. Identification of influential parameters through sensitivity analysis of the tough+ hydrate model using lh-oat sampling. Marine and Petroleum Geology 65, 141–156.

Huang, W., Qiao, F., Liu, H., Jia, X., Lohmar, B., 2016. From backyard to commercial hog production: Does it lead to a better or worse rural environment? China Agricultural Economic Review 8 (1), 22–36.

Hutchings, N. J., Nielsen, O.-K., Dalgaard, T., Mikkelsen, M. H., Børgesen, C. D., Thomsen, M., Ellermann, T., Højberg, A., Mogensen, L., Winther, M., 2014. A nitrogen budget for Denmark; developments between 1990 and 2010, and prospects for the future. Environmental Research Letters 9 (11), 115012.

IMO, 2014. Greenhouse Gas Study 2014. International Maritime Organization, London, UK, [online: accessed 10-Mars-2018].

IPCC, 2006. 2006 Inter-government Panel for Climate Change Guidelines for National Greenhouse Gas Inventories. prepared by the National Greenhouse Gas Inventories Programme. https://www.ipcc-nggip.iges.or.jp/public/2006gl/, [Online: accessed 2013-05-20].

ISO 14040, 2006. Environmental Management–Life Cycle Assessment—Principles and Framework. Tech. rep., International Organization for Standardization, Geneva, Switzerland.

ISO 14044, 2006. Environmental Management–Life Cycle Assessment—Requirements and Guidelines. Tech. rep., International Organization for Standardization, Geneva, Switzerland.

Jonker, J., Kohn, R., High, J., 2002. Dairy Herd Management Practices that Impact Nitrogen Utilization Efficiency. Journal of Dairy Science 85 (5), 1218–1226.

Jordan, P., Menary, W., Daly, K., Kiely, G., Morgan, G., Byrne, P., Moles, R., 2005. Patterns and processes of phosphorus transfer from Irish grassland soils to rivers—integration of laboratory and catchment studies. Journal of Hydrology 304 (1–4), 20–34.

Kastner, T., Schaffartzik, A., Eisenmenger, N., Erb, K.-H., Haberl, H., Krausmann, F., 2014. Cropland area embodied in international trade: Contradictory results from different approaches. Ecological Economics 104, 140–144.

Kirchherr, J., Reike, D., Hekkert, M., 2017. Conceptualizing the circular economy: An analysis of 114 definitions. Resources, Conservation and Recycling 127, 221–232.

Klop, G., Ellis, J., Bannink, A., Kebreab, E., France, J., Dijkstra, J., 2013. Meta-analysis of factors that affect the utilization efficiency of phosphorus in lactating dairy cows. Journal of Dairy Science 96 (6), 3936–3949.

Kohn, R., Dou, Z., Ferguson, J., Boston, R., 1997. A Sensitivity Analysis of Nitrogen Losses from Dairy Farms. Journal of Environmental Management 50 (4), 417–428.

Kratz, S., Halle, I., Rogasik, J., Schnug, E., 2004. Nutrient balances as indicators for sustainability of broiler production systems. British Poultry Science 45 (2), 149–157.

Krueger, T., Page, T., Hubacek, K., Smith, L., Hiscock, K., 2012. The role of expert opinion in environmental modelling. Environmental Modelling & Software 36, 4–18.

Larson, G., Fuller, D. Q., 2014. The evolution of animal domestication. Annual Review of Ecology, Evolution, and Systematics 45, 115–136.

Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J., Garnier, J., 2014. 50 year trends in nitrogen use efficiency of world cropping systems: the relationship between yield and nitrogen input to cropland. Environmental Research Letters 9 (10), 105011.

Lassaletta, L., Bouwman, L., Estellés, F., Westhoek, H., Beusen, A., Doelman, J., Calvet, S., Stehfest, E., van Grinsven, H., 2016. Modelling nitrogen use efficiency by world pig production systems in 2050 under contrasting production and dietary scenarios. In: Proceedings of the 2016 International Nitrogen Initiative Conference, "Solutions to improve nitrogen use efficiency for the world", 4-8 December 2016, Melbourne, Australia. International Nitrogen Initiative.

Latham, J., Cumani, R., Rosati, I., Bloise, M., 2014. FAO global land cover (GLC-SHARE) beta-release 1.0 database. http://www.fao.org/uploads/media/glc-share-doc.pdf/, [Online accessed

2018-01-10].

Lauer, M., Hansen, J. K., Lamers, P., Thrän, D., 2018. Making money from waste: The economic viability of producing biogas and biomethane in the idaho dairy industry. Applied Energy 222, 621–636.

Leach, A. M., Galloway, J. N., Bleeker, A., Erisman, J. W., Kohn, R., Kitzes, J., 2012. A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. Environmental Development 1 (1), 40–66.

Leip, A., Billen, G., Garnier, J., Grizzetti, B., Lassaletta, L., Reis, S., Simpson, D., Sutton, M. A., de Vries, W., Weiss, F., 2015. Impacts of european livestock production: nitrogen, sulphur, phosphorus and greenhouse gas emissions, land-use, water eutrophication and biodiversity. Environmental Research Letters 10 (11), 115004.

Leip, A., Britz, W., Weiss, F., de Vries, W., 2011. Farm, land, and soil nitrogen budgets for agriculture in Europe calculated with CAPRI. Environmental Pollution 159 (11), 3243–3253.

Leip, A., Weiss, F., LESSCHEN, J., Westhoek, H., 2013. The nitrogen footprint of food products in the European Union. The Journal of Agricultural Science, 1–14.

Leip, A., Weiss, F., Lesschen, J., Westhoek, H., 2014. The nitrogen footprint of food products in the european union. The Journal of Agricultural Science 152 (S1), 20–33.

Liu, C., Hotta, Y., Santo, A., Hengesbaugh, M., Watabe, A., Totoki, Y., Allen, D., Bengtsson, M., 2016. Food waste in japan: Trends, current practices and key challenges. Journal of Cleaner Production 133, 557–564.

Liu, Q., Wang, J., Bai, Z., Ma, L., Oenema, O., 2017. Global animal production and nitrogen and phosphorus flows. Soil Research 55 (6), 451–462.

Liu, Y., Villalba, G., Ayres, R. U., Schroder, H., 2008. Global Phosphorus Flows and Environmental Impacts from a Consumption Perspective. Journal of Industrial Ecology 12 (2), 229–247.

Lu, X., Mao, Q., Gilliam, F. S., Luo, Y., Mo, J., 2014. Nitrogen deposition contributes to soil acidification in tropical ecosystems. Global Change Biology 20 (12), 3790–3801.

Ma, L., Ma, W., Velthof, G., Wang, F., Qin, W., Zhang, F., Oenema, O., 2010. Modeling nutrient flows in the food chain of China. Journal of environmental quality 39 (4), 1279–1289.

MacDonald, J. M., McBride, W. D., 2009. The transformation of US livestock agriculture scale, efficiency, and risks. Economic Information Bulletin No. 43.

MacLeod, M., Gerber, P., Mottet, A., Tempio, G., Falcucci, A., Opio, C., Vellinga, T., Henderson, B., Steinfeld, H., 2013. Greenhouse gas emissions from pig and chicken supply chains - A global life cycle assessment. Tech. rep., Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Mahowald, N., Jickells, T. D., Baker, A. R., Artaxo, P., Benitez-Nelson, C. R., Bergametti, G., Bond, T. C., Chen, Y., Cohen, D. D., Herut, B., Kubilay, N., Losno, R., Luo, C., Maenhaut, W., McGee, K. A., Okin, G. S., Siefert, R. L., Tsukuda, S., 2008. Global distribution of atmospheric phosphorus sources, concentrations and deposition rates, and anthropogenic impacts.

Global Biogeochemical Cycles 22 (4), GB4026.

Makkar, H. P. S., 2017. Opinion paper: Food loss and waste to animal feed. animal 11 (7), 1093–1095.

McDowell, R. W., Sharpley, A. N., 2001. Approximating phosphorus release from soils to surface runoff and subsurface drainage. Journal of Environmental Quality 30 (2), 508–520.

Meena, V. S., Meena, S. K., Verma, J. P., Meena, R. S., Jat, L. K., 2015. Current trend to mitigate climate change and sustainable agriculture: a book review. Journal of Cleaner Production 102 (0), 548–549.

Middelaar, C. E., Cederberg, C., Vellinga, T. V., Werf, H. M. G., Boer, I. J. M., 2013. Exploring variability in methods and data sensitivity in carbon footprints of feed ingredients. The International Journal of Life Cycle Assessment 18 (4), 768–782.

Mink, T., Aldrich, E. L., Leon, L. A., 2015. Anaerobic biodigester technology in methane capture and manure management in Mexico - the history and current situation. Tech. rep., Mexico Low Emissions Development Program (MLED).

Misselbrook, T., Powell, J., Broderick, G., Grabber, J., 2005. Dietary Manipulation in Dairy Cattle: Laboratory Experiments to Assess the Influence on Ammonia Emissions. Journal of Dairy Science 88 (5), 1765–1777.

Moldanova, J., Grennfelt, P., Jonsson, A., Simpson, D., Spranger, T., Aas, W., Munthe, J., Rabl, A., 2011. Nitrogen as a threat to european air quality. In: Sutton, M., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., Grinsven, H. v., Grizzetti, B. (Eds.), The European Nitrogen Assessment. Cambridge University Press, pp. 405–433.

Mottet, A., de Haan, C., Falcucci, A., Tempio, G., Opio, C., Gerber, P., 2017. Livestock: On our plates or eating at our table? A new analysis of the feed/food debate. Global Food Security 14, 1–8.

Mu, W., Groen, E., van Middelaar, C., Bokkers, E., Hennart, S., Stilmant, D., de Boer, I. J. M., 2017. Benchmarking nutrient use efficiency of dairy farms: The effect of epistemic uncertainty. Agricultural Systems 156, 25–33.

Mu, W., van Middelaar, C., Bloemhof, J., Oenema, J., de Boer, I., 2016. Nutrient balance at chain level: a valuable approach to benchmark nutrient losses of milk production systems. Journal of Cleaner Production 112, 2419–2428.

Mueller, N. D., Gerber, J. S., Johnston, M., Ray, D. K., Ramankutty, N., Foley, J. A., 2012. Closing yield gaps through nutrient and water management. Nature 490, 254.

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., Zhang, H., 2013. Anthropogenic and natural radiative forcing. In: Stocker, T. F., Qin, D., Plattner, G.-K., Tignor, M. M., Allen, S. K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P. (Eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, pp. 661–740.

Nadeau, E., Englund, J.-E., Gustafsson, A. H., 2007. Nitrogen efficiency of dairy cows as affected by diet and milk yield. Livestock Science 111 (1–2), 45–56.

Navarro, J., Bryan, B. A., Marinoni, O., Eady, S., Halog, A., 2016. Mapping agriculture's impact by combining farm management handbooks, life-cycle assessment and search engine science. Environmental Modelling & Software 80, 54–65.

Nevens, F., Verbruggen, I., Reheul, D., Hofman, G., 2006. Farm gate nitrogen surpluses and nitrogen use efficiency of specialized dairy farms in Flanders: Evolution and future goals. Agricultural Systems 88 (2–3), 142–155.

Nori, S., Gemini, M., 2011. The common agricultural policy vis-à-vis European pastoralists: principles and practices. Pastoralism: Research, Policy and Practice 1 (1), 27.

Nyfeler, D., Huguenin-Elie, O., Suter, M., Frossard, E., Connolly, J., Lüscher, A., 2009. Strong mixture effects among four species in fertilized agricultural grassland led to persistent and consistent transgressive overyielding. Journal of Applied Ecology 46 (3), 683–691.

Öborn, I., Edwards, A., Witter, E., Oenema, O., Ivarsson, K., Withers, P., Nilsson, S., Richert Stinzing, A., 2003. Element balances as a tool for sustainable nutrient management: a critical appraisal of their merits and limitations within an agronomic and environmental context. European Journal of Agronomy 20 (1–2), 211–225.

OECD, 2001. Environmental Indicators for Agriculture: Methods and Results. Agriculture and Food, oecd ed. Edition. Vol. 3. Organisation For Economic Co-Operation and Development, Paris, France.

OECD, 2008. Environmental Performance of Agriculture in OECD Countries since 1990, oecd ed. Edition. OECD, Paris, France.

OECD/EUROSTAT, 2007. Gross Nitrogen Balances-Handbook. http://www.oecd.org/greengrowth/sustainable-agriculture/40820234.pdf 20, 2010.

OECD/FAO, 2015. OECD-FAO agricultural outlook 2015-2024. http://www.fao.org/3/a-i4738e.pdf/, [Online: accessed 19-January-2017].

Oenema, J., Burgers, S., van Keulen, H., van Ittersum, M., 2013. Stochastic uncertainty and Sensitivities of nitrogen flows on dairy farms in the Netherlands. In: Transitions in nutrient management on commercial pilot farms in the Netherlands. Wageningen UR Livestock Research, Wageningen, The Netherlands, p. 125.

Oenema, J., Burgers, S., van Keulen, H., van Ittersum, M., 2015. Stochastic uncertainty and sensitivities of nitrogen flows on dairy farms in The Netherlands. Agricultural Systems 137 (0), 126–138.

Oenema, J., van Ittersum, M., van Keulen, H., 2012. Improving nitrogen management on grassland on commercial pilot dairy farms in the Netherlands. Agriculture, Ecosystems & Environment 162 (0), 116–126.

Oenema, O., 2006. Nitrogen budgets and losses in livestock systems. International Congress Series 1293 (0), 262–271.

Oenema, O., Bannink, A., Sommer, S., Van Groenigen, J., Velthof, G., 2008. Chapter 12 -

Gaseous Nitrogen Emissions from Livestock Farming Systems. In: Hatfield, J., Follett, R. (Eds.), Nitrogen in the Environment (Second Edition). Academic Press, San Diego, pp. 395–441.

Oenema, O., Ju, X., de Klein, C., Alfaro, M., del Prado, A., Lesschen, J. P., Zheng, X., Velthof, G., Ma, L., Gao, B., Kroeze, C., Sutton, M., 2014. Reducing nitrous oxide emissions from the global food system. Current Opinion in Environmental Sustainability 9–10 (0), 55–64.

Oenema, O., Kros, H., de Vries, W., 2003. Approaches and uncertainties in nutrient budgets: implications for nutrient management and environmental policies. European Journal of Agronomy 20 (1–2), 3–16.

Oenema, O., Tamminga, S., 2005. Nitrogen in global animal production and management options for improving nitrogen use efficiency. Science in China Series C: Life Sciences 48 (2), 871–887.

Oenema, O., Wrage, N., Velthof, G. L., van Groenigen, J. W., Dolfing, J., Kuikman, P. J., 2005. Trends in global nitrous oxide emissions from animal production systems. Nutrient Cycling in Agroecosystems 72 (1), 51–65.

Oita, A., Malik, A., Kanemoto, K., Geschke, A., Nishijima, S., Lenzen, M., 2016. Substantial nitrogen pollution embedded in international trade. Nature Geoscience 9 (2), 111.

Ondersteijn, C., Beldman, A., Daatselaar, C., Giesen, G., Huirne, R., 2002. The Dutch Mineral Accounting System and the European Nitrate Directive: implications for N and P management and farm performance. Agriculture, Ecosystems & Environment 92 (2–3), 283–296.

Oomen, G., Lantinga, E., Goewie, E., Van der Hoek, K., 1998. Mixed farming systems as a way towards a more efficient use of nitrogen in European Union agriculture. Environmental Pollution 102 (1, Supplement 1), 697–704.

Opio, C., Gerber, P., Mottet, A., Tempio, G., MacLeod, M., Vellinga, T., Henderson, B., Steinfeld, H., 2013. Greenhouse gas emissions from ruminant supply chains: A global Life Cycle Assessment. Tech. rep., Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Özbek, F. Şaban., Leip, A., 2015. Estimating the gross nitrogen budget under soil nitrogen stock changes: A case study for turkey. Agriculture, Ecosystems & Environment 205, 48–56.

Papargyropoulou, E., Lozano, R., K. Steinberger, J., Wright, N., Ujang, Z. b., 2014. The food waste hierarchy as a framework for the management of food surplus and food waste. Journal of Cleaner Production 76, 106–115.

Parodi, A., Leip, A., De Boer, I. J. M., Slegers, P. M., Ziegler, F., Temme, E. H. M., Herrero, M., Tuomisto, H., Valin, H., Van Middelaar, C. E., Van Loon, J. J. A., van Zanten, H. H. E., 2018. The potential of future foods for sustainable and healthy diets. Nature Sustainability 1 (12), 782–789.

Payen, S., Ledgard, S. F., 2017. Aquatic eutrophication indicators in LCA: Methodological challenges illustrated using a case study in new zealand. Journal of Cleaner Production 168, 1463–1472.

Peoples, M., Brockwell, J., Herridge, D., Rochester, I., Alves, B., Urquiaga, S., Boddey, R., Dakora, F., Bhattarai, S., Maskey, S., et al., 2009. The contributions of nitrogen-fixing crop legumes to the productivity of agricultural systems. Symbiosis 48 (1-3), 1–17.

Philippe, F.-X., Cabaraux, J.-F., Nicks, B., 2011. Ammonia emissions from pig houses: Influencing factors and mitigation techniques. Agriculture, Ecosystems & Environment 141 (3-4), 245–260.

Pianosi, F., Beven, K., Freer, J., Hall, J. W., Rougier, J., Stephenson, D. B., Wagener, T., 2016. Sensitivity analysis of environmental models: A systematic review with practical workflow. Environmental Modelling & Software 79, 214–232.

Plaizier, J., Legesse, G., Ominski, K., Flaten, D., 2014. Whole-farm budgets of phosphorus and potassium on dairy farms in Manitoba. Canadian Journal of Animal Science 94 (1), 119–128.

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Science 360 (6392), 987–992.

Powell, J., Gourley, C., Rotz, C., Weaver, D., 2010. Nitrogen use efficiency: A potential performance indicator and policy tool for dairy farms. Environmental Science & Policy 13 (3), 217–228.

Powell, J., MacLeod, M., Vellinga, T., Opio, C., Falcucci, A., Tempio, G., Steinfeld, H., Gerber, P., 2013. Feed–milk–manure nitrogen relationships in global dairy production systems. Livestock Science 152 (2–3), 261–272.

Pujol, G., Iooss, B., Janon, A., Lemaitre, P., Gilquin, L., Le Gratiet, L., Touati, T., Ramos, B., Fruth, J., De Veiga, S., 2015. Sensitivity: Sensitivity analysis. R package version 1.11.1. R Development Core Team.

R Core Team, 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rai, M., Reeves, T. G., Pandey, S., Collette, L., 2011. Save and Grow: A Policymaker's Guide to Sustainable Intensification of Smallholder Crop Production. Food and Agriculture Organization of the United Nations, Rome, Italy.

Redding, M. R., Devereux, J., Phillips, F., Lewis, R., Naylor, T., Kearton, T., Hill, C., Weidemann, S., 2015. Field measurement of beef pen manure methane and nitrous oxide reveals a surprise for inventory calculations. Journal of Environmental Quality 44 (3), 720–728.

Reetz Jr, H. F., Heffer, P., Bruulsema, T. W., 2015. 4r nutrient stewardship: A global framework for sustainable fertilizer management. Managing Water and Fertilizer for Sustainable Agricultural Intensification, 65.

Reis, S., Bekunda, M., Howard, C. M., Karanja, N., Winiwarter, W., Yan, X., Bleeker, A., Sutton, M. A., 2016. Synthesis and review: Tackling the nitrogen management challenge: from global to local scales. Environmental Research Letters 11 (12), 120205.

Reuter, H. I., Nelson, A., Jarvis, A., 2007. An evaluation of void-filling interpolation methods for SRTM data. International Journal of Geographical Information Science 21 (9), 983–1008.

Ripple, W. J., Smith, P., Haberl, H., Montzka, S. A., McAlpine, C., Boucher, D. H., 2013. Ruminants, climate change and climate policy. Nature Climate Change 4 (1), 2.

Robinson, T., Thornton, P., Franceschini, G., Kruska, R., Chiozza, F., Notenbaert, A., Cecchi, G., Herrero, M., Epprecht, M., Fritz, S., You, L., Conchedda, G., Sere, L., 2011. Global livestock production systems. Food and Agriculture Organization of the United Nations (FAO) and International Livestock Research Institute (ILRI), Rome, Italy.

Rotz, C. A., Asem-Hiablie, S., Place, S., Thoma, G., 2019. Environmental footprints of beef cattle production in the united states. Agricultural Systems 169, 1–13.

Ryan, W., Hennessy, D., Murphy, J., Boland, T., Shalloo, L., 2011. A model of nitrogen efficiency in contrasting grass-based dairy systems. Journal of Dairy Science 94 (2), 1032–1044.

Röös, E., Patel, M., Spångberg, J., Carlsson, G., Rydhmer, L., 2016. Limiting livestock production to pasture and by-products in a search for sustainable diets. Food Policy 58, 1–13.

Saltelli, A., Annoni, P., 2010. How to avoid a perfunctory sensitivity analysis. Environmental Modelling & Software 25 (12), 1508–1517.

Saltelli, A., Chan, K., Scott, E. M., 2000. Sensitivity analysis, john wiley & sons, ltd Edition. Vol. 1. John Wiley & Sons, Ltd, Chichester, New York, Weinheim, Brisbane, Singapore, Toronto.

Saltelli, A., Ratto, M., Andres, T., Campolongo, F., Cariboni, J., Gatelli, D., Saisana, M., Tarantola, S., 2008. Global sensitivity analysis: the primer. John Wiley & Sons.

Saltelli, A., Tarantola, S., Chan, K.-S., 1999. A quantitative model-independent method for global sensitivity analysis of model output. Technometrics 41 (1), 39–56.

Sanz-Cobena, A., Lassaletta, L., Estellés, F., Del Prado, A., Guardia, G., Abalos, D., Aguilera, E., Pardo, G., Vallejo, A., Sutton, M., 2014. Yield-scaled mitigation of ammonia emission from n fertilization: the spanish case. Environmental Research Letters 9 (12), 125005.

Sarrazin, F., Pianosi, F., Wagener, T., 2016a. Global sensitivity analysis of environmental models: convergence and validation. Environmental Modelling & Software 79, 135–152.

Sarrazin, F., Pianosi, F., Wagener, T., 2016b. Global Sensitivity Analysis of environmental models: Convergence and validation. Environmental Modelling & Software 79, 135–152.

Sattari, S. Z., Bouwman, A. F., Giller, K. E., van Ittersum, M. K., 2012. Residual soil phosphorus as the missing piece in the global phosphorus crisis puzzle. Proceedings of the National Academy of Sciences 109 (16), 6348–6353.

Sattari, S. Z., Bouwman, A. F., Martinez Rodriguez, R., Beusen, A. H. W., van Ittersum, M. K., 2016. Negative global phosphorus budgets challenge sustainable intensification of grasslands. Nat Commun 7.

Schader, C., Muller, A., Scialabba, N. E.-H., Hecht, J., Isensee, A., Erb, K.-H., Smith, P., Makkar, H. P., Klocke, P., Leiber, F., et al., 2015. Impacts of feeding less food-competing feedstuffs to livestock on global food system sustainability. Journal of the Royal Society Interface 12 (113), 20150891.

Schaffner, M., Bader, H.-P., Scheidegger, R., 2009. Modeling the contribution of pig farming to pollution of the Thachin River. Clean Technologies and Environmental Policy 12 (4), 407–425.

Schröder, J., Aarts, H., ten Berge, H., van Keulen, H., Neeteson, J., 2003. An evaluation of whole-farm nitrogen balances and related indices for efficient nitrogen use. European Journal of Agronomy 20 (1–2), 33–44.

Schröder, J., Neeteson, J., 2008. Nutrient management regulations in The Netherlands. Geoderma 144 (3–4), 418–425.

Schröder, J., Scholefield, D., Cabral, F., Hofman, G., 2004. The effects of nutrient losses from agriculture on ground and surface water quality: the position of science in developing indicators for regulation. Environmental Science & Policy 7 (1), 15–23.

Schröder, J., Smit, A., Cordell, D., Rosemarin, A., 2011. Improved phosphorus use efficiency in agriculture: A key requirement for its sustainable use. Chemosphere 84 (6), 822–831.

Schulte, R., Herlihy, M., 2007. Quantifying responses to phosphorus in Irish grasslands: Interactions of soil and fertiliser with yield and P concentration. European Journal of Agronomy 26 (2), 144–153.

Schulte, R., Melland, A., Fenton, O., Herlihy, M., Richards, K., Jordan, P., 2010. Modelling soil phosphorus decline: Expectations of Water Framework Directive policies. Environmental Science & Policy 13 (6), 472–484.

Schulte, R., Richards, K., Daly, K., Kurz, I., McDonald, E., Holden, N., 2006. Agriculture, Meteorology and Water Quality in Ireland: a Regional Evaluation of Pressures and Pathways of Nutrient Loss to Water. Biology & Environment: Proceedings of the Royal Irish Academy 106 (2), 117–133.

Segato, S., Marchesini, G., Andrighetto, I., 2010. A comparison of nitrogen use efficiency and surplus in two dairy farms typologies. Italian Journal of Animal Science 8 (3s), 178–180.

Senthilkumar, K., Nesme, T., Mollier, A., Pellerin, S., 2012. Conceptual design and quantification of phosphorus flows and balances at the country scale: The case of France. Global Biogeochemical Cycles 26 (2).

Sharpley, A., 1999. Agricultural phosphorus, water quality, and poultry production: are they compatible? Poultry Science 78 (5), 660–673.

Sharpley, A., Jarvie, H. P., Buda, A., May, L., Spears, B., Kleinman, P., 2013. Phosphorus Legacy: Overcoming the Effects of Past Management Practices to Mitigate Future Water Quality Impairment. Journal of Environmental Quality 42 (5), 1308–1326.

Simon, J.-C., Grignani, C., Jacquet, A., Le Corre, L., Pagès, J., 2000. Typologie des bilans d'azote de divers types d'exploitation agricole: recherche d'indicateurs de fonctionnement. Agronomie 20 (2), 175–195.

Sin, G., Gernaey, K. V., Neumann, M. B., van Loosdrecht, M. C., Gujer, W., 2011. Global sensitivity analysis in wastewater treatment plant model applications: Prioritizing sources of uncertainty. Water Research 45 (2), 639–651.

Smil, V., 1999. Nitrogen in crop production: An account of global flows. Global Biogeochem. Cycles 13 (2), 647–662.

Sobol, 2001. Global sensitivity indices for nonlinear mathematical models and their monte carlo estimates. Mathematics and Computers in Simulation 55 (1), 271–280.

Spears, R., Kohn, R., Young, A., 2003a. Whole-farm nitrogen balance on Western dairy farms. Journal of Dairy Science 86 (12), 4178–4186.

Spears, R., Young, A., Kohn, R., 2003b. Whole-Farm Phosphorus Balance on Western Dairy Farms. Journal of Dairy Science 86 (2), 688–695.

Springmann, M., Clark, M., Mason-D'Croz, D., Wiebe, K., Bodirsky, B. L., Lassaletta, L., de Vries, W., Vermeulen, S. J., Herrero, M., Carlson, K. M., et al., 2018. Options for keeping the food system within environmental limits. Nature 562 (7728), 519.

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., de Vries, W., de Wit, C. A., 2015. Planetary boundaries: Guiding human development on a changing planet. Science 347 (6223), 1259855.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., De Haan, C., 2006. Livestock's long shadow. Food and Agriculture Organization of United Nations, Rome, Italy.

Stevens, C. J., Dise, N. B., Mountford, J. O., Gowing, D. J., 2004. Impact of nitrogen deposition on the species richness of grasslands. Science 303 (5665), 1876.

Strokal, M., Ma, L., Bai, Z., Luan, S., Kroeze, C., Oenema, O., Velthof, G., Zhang, F., 2016. Alarming nutrient pollution of chinese rivers as a result of agricultural transitions. Environmental Research Letters 11 (2), 024014.

Strokal, M., Yang, H., Zhang, Y., Kroeze, C., Li, L., Luan, S., Wang, H., Yang, S., Zhang, Y., 2014. Increasing eutrophication in the coastal seas of china from 1970 to 2050. Marine Pollution Bulletin 85 (1), 123–140.

Suh, S., Yee, S., 2011. Phosphorus use-efficiency of agriculture and food system in the US. Chemosphere 84 (6), 806–813.

Sutton, M. A., Bleeker, A., Howard, C., Bekunda, M., Grizzetti, B., Vries, W. d., Grinsven, H. v., Abrol, Y., Adhya, T., Billen, G., Davidson, E. A., Datta, A., Diaz, R., Erisman, J., Liu, X. J., Oenema, O., Palm, C., Raghuram, N., Reis, S., Scholz, R. W., Sims, J. T., Westhoek, H., Zhang, F. S., 2013a. Our Nutrient World: the challenge to produce more food and energy with less pollution. Global Overview of Nutrient Management. NERC/Centre for Ecology & Hydrology, Edinburgh, UK.

Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., Van Grinsven, H., Grizzetti, B., 2011. The European nitrogen assessment: sources, effects and policy perspectives. Cambridge University Press.

Sutton, M. A., Reis, S., Riddick, S. N., Dragosits, U., Nemitz, E., Theobald, M. R., Tang, Y. S., Braban, C. F., Vieno, M., Dore, A. J., et al., 2013b. Towards a climate-dependent paradigm of ammonia emission and deposition. Philosophical Transactions of the Royal Society of London B: Biological Sciences 368 (1621), 20130166.

Swaney, D. P., Howarth, R. W., Hong, B., 2018. Nitrogen use efficiency and crop production: Patterns of regional variation in the United States, 1987–2012. Science of The Total Environment 635, 498–511.

Takata, M., Fukushima, K., Kino-Kimata, N., Nagao, N., Niwa, C., Toda, T., 2012. The effects of recycling loops in food waste management in japan: based on the environmental and economic evaluation of food recycling. Science of the Total Environment 432, 309–317.

Thieme, O., Makkar, H. P., 2017. Utilisation of loss and waste during the food-production cycle as livestock feed. Animal Production Science 57 (4), 601–607.

Thien Thu, C. T., Cuong, P. H., Hang, L. T., Chao, N. V., Anh, L. X., Trach, N. X., Sommer, S. G., 2012. Manure management practices on biogas and non-biogas pig farms in developing countries – using livestock farms in Vietnam as an example. Journal of Cleaner Production 27 (0), 64–71.

Thomassen, M., Dolman, M., van Calker, K., de Boer, I., 2009. Relating life cycle assessment indicators to gross value added for Dutch dairy farms. Ecological Economics 68 (8–9), 2278–2284.

Tóth, G., Guicharnaud, R.-A., Tóth, B., Hermann, T., 2014. Phosphorus levels in croplands of the European Union with implications for P fertilizer use. European Journal of Agronomy 55 (0), 42–52.

Tóth, G., Jones, A., Montanarella, L., 2013a. The LUCAS topsoil database and derived information on the regional variability of cropland topsoil properties in the European Union. Environ Monit Assess 185 (9), 7409–7425.

Tóth, G., Jones, A., Montanarella, L., 2013b. LUCAS Topsoil Survey: methodology, data and results. Publications Office of the European Union. EUR26102.

Ulén, B., Bechmann, M., Fölster, J., Jarvie, H. P., Tunney, H., 2007. Agriculture as a phosphorus source for eutrophication in the north-west European countries, Norway, Sweden, United Kingdom and Ireland: a review. Soil Use and Management 23, 5–15.

UN, 2018. The Paris Agreement. http://unfccc.int/paris_agreement/items/9485.php/, online: accessed 2018-03-16.

Uusitalo, L., Lehikoinen, A., Helle, I., Myrberg, K., 2015. An overview of methods to evaluate uncertainty of deterministic models in decision support. Environmental Modelling & Software 63, 24–31.

Uwizeye, A., Gerber, P. J., Groen, E. A., Dolman, M. A., Schulte, R. P. O., de Boer, I. J. M., 2017. Selective improvement of global datasets for the computation of locally relevant environmental indicators: A method based on global sensitivity analysis. Environmental Modelling & Software 96, 58–67.

Uwizeye, A., Gerber, P. J., Opio, C. I., Tempio, G., Mottet, A., Makkar, H. P., Falcucci, A., Steinfeld, H., de Boer, I. J. M., 2019. Nitrogen flows in global pork supply chains and potential improvement from feeding swill to pigs. Resources, Conservation and Recycling 146, 168–179.

Uwizeye, A., Gerber, P. J., Schulte, R. P. O., de Boer, I. J. M., 2016a. A comprehensive framework to assess the sustainability of nutrient use in global livestock supply chains. Journal of cleaner production 129, 647–658.

Uwizeye, A., Groen, E. A., Gerber, P. J., Schulte, R. P., De Boer, I. J. M., 2016b. Reducing uncertainty at minimal cost: a method to identify important input parameters and prioritize data collection. In: Book of Abstracts: Putting LCA into Practice. The 10th international conference on Life Cycle Assessment of Food 2016. LCA Food 2016, pp. 1154–1159.

Van der Hoek, K. W., 1998. Nitrogen efficiency in global animal production. Environmental Pollution 102 (1, Supplement 1), 127–132.

van Gijlswijk, R., Coenen, P., Pulles, T., Van Der Sluijs, J., 2004. Uncertainty assessment of NOx, SO_2 and NH_3 emissions in the Netherlands. Apeldoorn: TNO, [Online accessed 2015-10-11].

Van Middelaar, C., Dijkstra, J., Berentsen, P., De Boer, I., 2014. Cost-effectiveness of feeding strategies to reduce greenhouse gas emissions from dairy farming. Journal of Dairy Science 97 (4), 2427–2439.

van Zanten, H. H. E., Herrero, M., Van Hal, O., Röös, E., Muller, A., Garnett, T., Gerber, P. J., Schader, C., De Boer, I. J. M., 2018. Defining a land boundary for sustainable livestock consumption. Global Change Biology 24 (9), 4185–4194.

van Zanten, H. H. E., Meerburg, B. G., Bikker, P., Herrero, M., de Boer, I. J. M., 2016. Opinion paper: The role of livestock in a sustainable diet: a land-use perspective. animal 10 (4), 547–549.

Vellinga, T., Blonk, H., Marinussen, M., van Zeist, W., de Boer, I., Starmans, D., 2013. Methodology Used in FeedPrint: a Tool Quantifying Greenhouse Gas Emissions of Feed Production and Utilization, wageningen ur livestock research Edition. Vol. Report 674. Wageningen UR Livestock Research, Wageningen, The Netherlands.

Velthof, G., Oudendag, D., Witzke, H., Asman, W., Klimont, Z., Oenema, O., 2009. Integrated assessment of nitrogen losses from agriculture in EU-27 using MITERRA-EUROPE. Journal of Environmental Quality 38 (2), 402–417.

Velthof, G. L., Hou, Y., Oenema, O., 2015. Nitrogen excretion factors of livestock in the European Union: a review. Journal of the Science of Food and Agriculture 95 (15), 3004–3014.

Verheijen, L., Wiersema, D., Hulshoff Pol, L., de Wit, J., 1996. Management of waste from animal product processing. Food and Agriculture Organization of United Nations, Rome, Italy.

Vonk, J., van der Sluis, S., Bannink, A., van Bruggen, C., Groenestein, C., Huijsmans, J., van der Kolk, J., Lagerwerf, L., Luesink, H., Voshaar, S. O., Velthof, G., 2018. Methodology for estimating emissions from agriculture in the netherlands–update 2018. calculations of CH_4, NH_3, N_2O, NOx, PM10, PM2.5 and CO_2 with the National Emission Model for Agriculture (NEMA). http://edepot.wur.nl/443801/, [Online: accessed 15-June-2018].

Vu, T., Tran, M., Dang, T., 2007. A survey of manure management on pig farms in Northern Vietnam. Livestock Science 112 (3), 288–297.

Wall, D., Jordan, P., Melland, A., Mellander, P., Mechan, S., Shortle, G., 2013. Forecasting the decline of excess soil phosphorus in agricultural catchments. Soil Use and Management 29, 147–154.

Watson, C., Bengtsson, H., Ebbesvik, M., Loes, A.-K., Myrbeck, A., Salomon, E., Schröder, J., Stockdale, E., 2002. A review of farm-scale nutrient budgets for organic farms as a tool for management of soil fertility. Soil Use and Management 18, Issue supplement, 264–273.

Weiler, V., Udo, H. M., Viets, T., Crane, T. A., De Boer, I. J., 2014. Handling multi-functionality of livestock in a life cycle assessment: the case of smallholder dairying in Kenya. Current Opinion in Environmental Sustainability 8, 29–38.

Westendorf, M., Dong, Z., Schoknecht, P., 1998. Recycled cafeteria food waste as a feed for swine: nutrient content digestibility, growth, and meat quality. Journal of Animal Science 76 (12), 2976–2983.

Wilkes, A., Reisinger, A., Wollenberg, E., Van Dijk, S., 2017. Measurement, reporting and verification of livestock GHG emissions by developing countries in the UNFCCC: current practices and opportunities for improvement. https://cgspace.cgiar.org/rest/bitstreams/146542/retrieve/, [Online: Accessed 2017-12-13].

Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., Jonell, M., Clark, M., Gordon, L. J., Fanzo, J., Hawkes, C., Zurayk, R., Rivera, J. A., De Vries, W., Majele Sibanda, L., Afshin, A., Chaudhary, A., Herrero, M., Agustina, R., Branca, F., Lartey, A., Fan, S., Crona, B., Fox, E., Bignet, V., Troell, M., Lindahl, T., Singh, S., Cornell, S. E., Srinath Reddy, K., Narain, S., Nishtar, S., Murray, C. J. L., 2019. Food in the anthropocene: the EAT–lancet commission on healthy diets from sustainable food systems. The Lancet 393 (10170), 447–492.

Wolf, P., Groen, E. A., Berg, W., Prochnow, A., Bokkers, E. A., Heijungs, R., de Boer, I. J., 2017. Assessing greenhouse gas emissions of milk production: which parameters are essential? The International Journal of Life Cycle Assessment 22 (3), 441–455.

World Bank, 2009. Minding the stock: Bringing public policy to bear on livestock sector development. Report No 44010-GLB[Online: accessed 2019-01-10].

Wu, H., Yuan, Z., Zhang, Y., Gao, L., Liu, S., 2014a. Life cycle phosphorus use efficiency of the farming system in Anhui Province, Central China. Resources, Conservation and Recycling 83 (0), 1–14.

Wu, H., Yuan, Z., Zhang, Y., Gao, L., Liu, S., Geng, Y., 2014b. Data uncertainties in anthropogenic phosphorus flow analysis of lake watershed. Journal of Cleaner Production 69 (0), 74–82.

Xue, X., Landis, A. E., 2010. Eutrophication Potential of Food Consumption Patterns. Environmental Science & Technology 44 (16), 6450–6456.

Yamaji, K., Ohara, T., Akimoto, H., 2004. Regional-specific emission inventory for NH_3, N_2O, and CH_4 via animal farming in South, Southeast, and East Asia. Atmospheric Environment 38 (40), 7111–7121.

Yao, Y., Zhao, Y., 2007. Conflict analysis based on discernibility and indiscernibility. In: Foundations of Computational Intelligence, 2007. FOCI 2007. IEEE Symposium on. IEEE, pp. 302–307.

Yuan, Z., Wu, H., He, X., Liu, X., 2014. A bottom–up model for quantifying anthropogenic phosphorus cycles in watersheds. Journal of Cleaner Production 84, 502–508.

Zhu, B., Kros, J., Lesschen, J. P., Staritsky, I. G., de Vries, W., 2016. Assessment of uncertainties in greenhouse gas emission profiles of livestock sectors in Africa, Latin America and Europe.

Regional Environmental Change 16 (6), 1571–1582.

zu Ermgassen, E. K., Kelly, M., Bladon, E., Salemdeeb, R., Balmford, A., 2018. Support amongst uk pig farmers and agricultural stakeholders for the use of food losses in animal feed. PloS one 13 (4), e0196288.

zu Ermgassen, E. K., Phalan, B., Green, R. E., Balmford, A., 2016. Reducing the land use of EU pork production: where there's swill, there's a way. Food Policy 58, 35–48.

Summary

The global livestock sector is rapidly transforming. Over the past few decades, many livestock systems over the world have evolved from local, small-scale mixed crop-livestock systems to global and demand-driven supply chains, in which feed and animal production stages are often disconnected. These changes, driven by economic opportunities, have altered the way livestock production impacts global nitrogen and phosphorus flows and emissions. These emissions take place in several stages of the supply chains, namely feed production, animal production and processing of animal products and threaten water, soil and air quality, but also climate, biodiversity and human health. Achieving better nutrient management is thus an important aspect of improving environmental performance in the livestock sector. Improving the efficiency of nutrient use has been identified as the main strategy to reduce environmental pressures while achieving global food security and sustainability.

To reduce nutrient losses in livestock supply chains, there is a need for methods and indicators that determine these losses or the other way around, determine the nutrient use efficiency (NUE). Most studies that evaluate NUE focus on animal, farm or regional level. For global livestock supply chains, however, that run across national and continental boundaries, such approaches over-look nutrient losses associated with off-farm activities, such as the production of feed. Some studies assess nutrient losses and NUE at a chain level, but they do not consider the entire supply chain and do not consider the effect of nutrient recycling and stock changes on NUE, or do not identify hotspots of nutrient loss along the chain that are required to support targeted nutrient improvement pathways towards sustainable nutrient use. The two objectives of this thesis, therefore, were to develop a framework of indicators to assess nutrient flows and emissions along global livestock supply chains, while identifying data, which can be improved to enhance the accuracy of the results, and to assess the impacts of the global livestock supply chains on the nitrogen flows, while exploring the improvement options.

Evaluating nutrient use and flows in livestock supply chains requires a framework and data to estimate flows, emissions and relevant indicators from each production stage. To develop such a framework, **Chapter 2** first reviewed existing studies on nutrient use in the livestock sector. The review showed that four methods were used previously to analyse nutrient use in the livestock sector, namely a nutrient balance, nutrient use efficiency, material flow analysis and life cycle assessment. Among these methods, nutrient use efficiency appeared a suitable approach to benchmark nutrient management at the animal level, and to some extent at the farm level. The analysis showed that integrating the life cycle approach into NUE, therefore, could allow for the

computation of supply chain level NUE, which was proposed as a valuable indicator of nutrient management sustainability.

To this end, in **Chapter 3**, a comprehensive framework of indicators, based on the life-cycle approach, was developed to assess the efficiency of nitrogen and phosphorus use. The framework represents nutrient flows in the typical livestock supply chain from the "cradle-to-primary-processing-gate", including crop/pasture production, animal production and primary processing stage as well as the transportation of feed materials, live animals or animal products. It encompassed three indicators, including the life-cycle nutrient use efficiency (life-cycle-NUE), life-cycle net nutrient balance (life-cycle-NNB) and nutrient hotspot index (NHI). The framework was tested for a case study of mixed dairy supply chains in Europe. The proposed indicators were found to be suitable to describe different aspects of nitrogen and phosphorus dynamics and, therefore, were all needed.

This framework of indicators developed requires detailed data such as nutrient inputs into soils, herd parameters, climate, emission factors, and manure management, to estimate nutrient flows and three nutrient use indicators. These data are highly variable at the global scale, resulting in large uncertainties due to the differences in geographical representation, time boundaries, farming technology and completeness. In **Chapter 4**, a method was proposed to identify the important inputs parameters that contribute significantly to the variance of the results. This method, which relies on a global sensitivity analysis is tested for the cases studies of mixed cattle dairy systems in the Netherlands and Rwanda, using the Global Environmental Assessment Model (GLEAM) dataset. The results showed that uncertainties of a few important input parameters, such as manure deposited on grasslands, applied manure and synthetic fertilizer, milk production and emission factors, could explain most of the variance of N use indicators. We subsequently fixed non-important and substituted important parameters in GLEAM with new field survey data, which substantially improved the results of N use indicators. This method can be applied to any environmental modelling using global datasets to improve their relevance by prioritizing important parameters for additional data collection.

In **Chapter 5**, the framework of indicators was applied to assess N use, flows and emissions, in the global pork supply chains and to evaluate the effects of feeding swill to pigs as a strategy to integrate better livestock in a circular bio-economy. Results showed that N emissions into the environment amount to around 14.7 Tg N y^{-1}. More than half of these emissions take place in the backyard system, although this system contributed only 27% to total pork production. Industrial systems emitted 23% of total N emissions but contributed more than half of the global pork production (56%). Intermediate systems contributed around 19% to both pork production and N emissions. We found that most of N emissions are in the form of NO_3^- and organic N lost to surface and groundwater, with large implications for aquatic eutrophication. Backyard and intermediate systems, with relatively high connectivity between animal and crop production were more efficient than industrial systems. These results showed that the efficiency of N use and the magnitude of N losses per unit of area depend chiefly on the region (agro-ecological and economic context), on the origin of feed, and on manure management systems. The results also showed that the substitution of swill for grains and soybeans could improve N use indicators and abate N emissions. Applied on a global scale to industrial systems, this strategy was estimated to save 31 Mt of soybeans and 20 Mt of grains on dry matter basis, equivalent to 16 M ha of land use. Implementing swill feeding, however, would require innovative policies to guide the collection, treatment, and usage of swill, and ensure safety and traceability.

In **Chapter 6**, the global nitrogen use and flows were evaluated for livestock supply chains using the spatially explicit Global Livestock Environmental Assessment Model and its database. The results showed that, globally, livestock supply chains are responsible for around one-third of human-induced N emissions of which 63% take place in 2 regions (i.e. South Asia and East and Southeast Asia), and 61% at the feed production stage. These emissions are in the form of NO_3^- (28 Tg N y^{-1}), NH_3 (26 Tg N y^{-1}), NO_x (8 Tg N y^{-1}) and N_2O (2 Tg N y^{-1}). The magnitude and concentration of N losses imply that there is both urgent need to reduce these emissions and the opportunity to design targeted mitigation interventions. The wide range of values calculated for N use indicators further indicates that good practices are available and already implemented in parts of the value chains. Mitigation options proposed include improvement of feed fertilisation, and manure management through the adoption of innovative technology and best practices. These improvement pathways can be effective because N emissions are concentrated in few regions, supply chains and steps along the chain and the wide variability of N use indicators offers opportunity to design mitigation interventions. The adoption of good practices would likely require additional investments, knowledge transfer and additional solutions to improve simultaneously the socio-economic conditions of farmers worldwide. The design and implementation of interventions should consider potential trade-offs and synergies with other sustainability dimensions, such as climate change, resource scarcity and food security.

In **Chapter 7**, the development of the framework of indicators, modelling challenges and data quality were discussed. The discussion revealed that the three indicators proposed in this framework: Life-cycle-NUE_N, Life-cycle-NNB_N and NHI_N provide a comprehensive analysis of nutrient use, flows and emissions in global livestock supply chains. The discussion revealed that livestock supply chains play a role in the net transfer of soil fertility from grassland to cropland and in shifting N embodied in feed between countries, which may be lost through unregulated disposals of manure. The chapter discussed the potential improvement options but emphasised the need to consider rebound effect related to the improvement of NUE, which may result in a consumption surge. The chapter discussed nutrients challenges in connection to the overall sustainability of the livestock sector, which uses of a large number of natural resources such as land, freshwater, often with low efficiency and contributes to global human-induced greenhouse emissions. Because the livestock supply chains are embedded in the economy and culture of societies. They contribute to rural development, human diets, trade balances, risk management and other relevant development outcomes, while building resilience and adaptation to climate change. Livestock can also negatively affect these outcomes, e.g. contributing to public health issues (diets, zoonoses, Anti-microbial resistance), and offering poor conditions to livestock producers and animals themselves. Addressing N challenges will require the consideration of potential trade-offs and synergies with these wider sustainability dimensions (e.g. poverty eradication, nutrition, human health) and it will also need to be done in conjunction with other interventions that address the growth of the livestock sector. The chapter ends up by calling for a global initiative with a strong representation of livestock sector scientists and stakeholders to tackle the N pollution.

Résumé

Le secteur mondial de l'élevage se transforme rapidement. Au cours des dernières décennies, de nombreux systèmes d'élevage ont été développés, passant de systèmes locaux et de petite échelle, souvent mixtes culture-élevage à des filières d'élevage orientées vers le marché, dans lesquelles la production de l'aliment est souvent déconnectée de la production animale. Ces changements, motivés par les opportunités économiques, ont modifié la manière dont la production animale affecte les flux et les émissions d'azote et de phosphore à l'échelle mondiale. Ces émissions ont lieu en plusieurs étapes des chaînes d'approvisionnement de l'élevage, à savoir la production d'aliments, la production animale et la transformation des produits d'origine animale. Ces émissions menacent la qualité de l'eau, du sol et de l'air, par conséquent le climat, la biodiversité et la santé humaine. La meilleure gestion des nutriments est donc un aspect important de l'amélioration des performances environnementales dans le secteur de l'élevage. L'amélioration de l'efficience de l'utilisation des nutriments a été identifié comme la principale stratégie pour réduire les pressions sur l'environnement tout en assurant la sécurité alimentaire et la durabilité des systèmes mondiales.

Pour réduire les pertes des nutriments dans les chaînes d'approvisionnement de l'élevage, il est nécessaire de disposer de méthodes et d'indicateurs permettant de déterminer ces pertes ou, inversement, de déterminer l'efficience d'utilisation des nutriments (NUE). La plupart des études évaluant NUE se concentrent sur les animaux, les exploitations agricoles ou les régions. Pour les chaînes d'approvisionnement mondiales de l'élevage, toutefois, qui traversent les frontières nationales et continentales, de telles approches négligent les pertes de nutriments associées aux activités se déroulant en dehors des exploitations agricoles, telles que la production d'aliments pour animaux. Certaines études évaluent les pertes de nutriments et NUE au niveau de la chaîne, mais ils ne considèrent pas l'ensemble de la chaîne d'approvisionnement et ne prennent pas en compte l'effet du recyclage de nutriments et les variations des stocks sur NUE, ou ne permettent pas d'identifier où les points sensibles des pertes de nutriments sont concentrés le long de la chaîne. Ces points sont nécessaires pour formuler les solutions ciblées d'amélioration de l'utilisation des nutriments pour une gestion durable de leur utilisation. Les deux objectifs de cette thèse étaient donc de développer un cadre méthodologique avec indicateurs permettant d'évaluer les flux et les émissions de nutriments tout au long des chaînes d'approvisionnement mondiales de l'élevage, tout en identifiant des données pouvant être améliorées pour augmenter l'exactitude des résultats et d'évaluer les impacts des chaînes d'approvisionnement de l'élevage à l'échelle mondiale sur les flux d'azote, tout en explorant les options d'amélioration.

L'évaluation de l'utilisation et des flux de nutriment dans les chaînes d'approvisionnement d'élevage nécessite un cadre méthodologique et des données permettant d'estimer les flux, les émissions et les indicateurs pertinents à chaque étape de la production. Pour développer un tel cadre méthodologique, Le **Chapitre 2** a tout d'abord passé en revue les études existantes sur l'utilisation des nutriments dans le secteur de l'élevage. L'étude bibliographique a montré que quatre méthodes étaient précédemment utilisées pour analyser l'utilisation des nutriments dans le secteur de l'élevage, à savoir le bilan de nutriments, l'efficience de l'utilisation des nutriments (NUE), l'analyse des flux de matières et l'analyse du cycle de vie. Parmi ces méthodes, l'efficience d'utilisation des nutriments est apparue comme une approche appropriée pour évaluer la gestion des nutriments au niveau des animaux et, dans une certaine mesure, au niveau de la ferme. L'analyse a montré que l'intégration de l'approche du cycle de vie dans NUE pourrait donc permettre de calculer le NUE au niveau de la chaîne d'approvisionnement. Cette approche a été proposé comme un indicateur pertinent de la durabilité de la gestion de flux de nutriments.

À cette fin, dans le **Chapitre 3**, un cadre méthodologique d'indicateurs pertinents, basé sur l'approche du cycle de vie, a été développé pour évaluer l'efficience de l'utilisation de l'azote et du phosphore. Le cadre méthodologique représente les flux de nutriments dans la chaîne d'approvisionnement de l'élevage typique de la "*de l'extraction des matières premières à la transformation primaire des produits*", y compris la production des cultures agricoles et fourragères, la production animale et la première transformation des produits d'origine animale, ainsi que le transport de l'aliment des animaux vivants ou des produits d'origine animale. Il englobait trois indicateurs, dont l'efficience d'utilisation des nutriments au niveau cycle de vie (Life-cycle-NUE), le bilan net des nutriments au niveau cycle de vie (Life-cycle-NNB) et l'indice des points sensibles aux pertes de nutriments (NHI). Le cadre méthodologique a été testé pour une étude de cas des systèmes de production mixte de vaches laitières en Europe. Les indicateurs proposés se sont avérés appropriés pour décrire différents aspects de la dynamique de l'azote et du phosphore et ont donc été tous pertinents.

Ce cadre méthodologique d'indicateurs développé nécessite des données détaillées telles que les apports en nutriments dans les sols, les paramètres des troupeaux, le climat, les facteurs d'émission et la gestion du fumier, afin d'estimer les flux de nutriments et les trois indicateurs de leur utilisation. Ces données sont très variables à l'échelle globale, ce qui entraîne de grandes incertitudes en raison des différences de représentation géographique, de limites de temps, de technologie de production et d'exhaustivité. Dans le **Chapitre 4**, une méthode a été proposée pour identifier les facteurs d'entrée importants du modèle qui contribuent de manière significative à la variance des résultats. Cette méthode, qui repose sur une analyse de sensibilité globale, est testée pour les études de cas de systèmes de production mixte de vaches laitières aux Pays-Bas et au Rwanda, en utilisant une base de données du modèle globale d'évaluation environnementale de l'élevage (GLEAM). Les résultats ont montré que les incertitudes de quelques facteurs d'entrée importants, tels que le lisier déposé dans les prairies, le lisier épandu et les engrais synthétiques, la production de lait et les facteurs d'émission, pourraient expliquer l'essentiel de la variance des indicateurs d'utilisation de l'azote. Par la suite, nous avons fixé les facteurs non–importants, dont la variabilité n'influe pas la variance des indicateurs, et substitué les facteurs importants dans la base de données de GLEAM avec de nouvelles données issues d'enquête de terrain, ce qui a considérablement amélioré les résultats des indicateurs d'utilisation de l'azote. Cette méthode peut être appliquée à toute modélisation environnementale utilisant des jeux de données globaux pour améliorer leur pertinence en hiérarchisant les facteurs d'entrée importants pour la collecte de données supplémentaires.

Au **Chapitre 5**, le cadre méthodologique d'indicateurs a été appliqué pour évaluer l'utilisation, les flux et les émissions d'azote dans les chaînes d'approvisionnement mondiales du porc et pour évaluer les effets de l'alimentation des porcs en eaux grasses en tant que meilleure stratégie d'intégration de l'élevage dans une bio-économie circulaire. Les résultats ont montré que les émissions d'azote dans l'environnement s'élevaient à environ 14,7 Tg N par an. Plus de la moitié de ces émissions se produisent dans le système de basse-cour, bien que ce système ne représente que 27% de la production totale de viande de porc. Les systèmes industriels émettaient 23% des émissions totales d'azote mais représentaient plus de la moitié de la production mondiale de viande de porc (56%). Les systèmes intermédiaires représentaient environ 19% de la production porcine et des émissions d'azote. Nous avons constaté que la plupart des émissions d'azote se présentaient sous forme de ${NO_3}^-$ et d'azote organique perdu dans les eaux de surface et les eaux souterraines, ce qui avait des incidences importantes sur l'eutrophisation des milieux aquatiques. Les systèmes de basse-cour et intermédiaires, avec une connectivité relativement élevée entre la production animale et la production des cultures, étaient plus efficaces que les systèmes industriels. Ces résultats ont montré que l'efficacité de l'utilisation de l'azote et l'ampleur des pertes d'azote par unité de surface dépendent principalement de la région (contexte agro-écologique et économique), de l'origine des aliments pour animaux et des systèmes de gestion du fumier. Les résultats ont également montré que la substitution des eaux grasses aux céréales et au soja pourrait améliorer les indicateurs d'utilisation de l'azote et réduire les émissions d'azote. Appliquée à l'échelle mondiale aux systèmes industriels, cette stratégie devrait permettre d'économiser 31 Mt de soja et 20 Mt de céréales en poids de matière sèche, ce qui correspond à 16 millions d'hectares d'utilisation des terres. Cependant, la mise en place d'une alimentation basée sur les eaux grasses nécessiterait des politiques innovantes pour guider la collecte, le traitement et l'utilisation des eaux grasses, ainsi que pour assurer la sécurité et la traçabilité.

Dans le **Chapitre 6**, l'utilisation et les flux d'azote dans le monde ont été évalués pour les chaînes d'approvisionnement de l'élevage à l'aide du modèle global d'évaluation de l'environnementale de l'élevage (GLEAM) qui est spatialement explicite et de sa base de données. Les résultats ont montré que, dans le monde, l'élevage est responsable d'environ un tiers des émissions d'azote d'origine anthropogénique, dont 63% ont lieu dans deux régions (Asie du Sud et Asie de l'Est et du Sud-Est) et 61% ont lieu durant la production d'aliments. Ces émissions se présentent sous la forme de: ${NO_3}^-$ (28 Tg N an^{-1}), NH_3 (26 Tg N an^{-1}), NO_x (8 Tg N an^{-1}) et N_2O (2 Tg N an^{-1}). L'ampleur et la concentration des pertes en azote impliquent qu'il est à la fois urgent de réduire ces émissions et qu'il est possible de concevoir des interventions d'atténuation ciblées. La large gamme de valeurs calculées pour les indicateurs d'utilisation de l'azote indique également que de bonnes pratiques sont disponibles et déjà mises en œuvre dans certaines parties des chaînes d'approvisionnement. Les options d'atténuation proposées comprennent l'amélioration de l'utilisation des engrais de synthèse et du lisier durant la production de l'aliment pour animaux et la gestion du fumier grâce à l'adoption de technologies innovantes et de meilleures pratiques de production. Ces voies d'amélioration peuvent être efficaces car les émissions d'azote sont concentrées dans quelques régions, systèmes d'élevage, et étapes tout au long de la chaîne, et la grande variabilité des indicateurs d'utilisation de l'azote offre la possibilité de concevoir ces interventions d'atténuation. L'adoption de bonnes pratiques de production nécessiterait probablement des investissements supplémentaires, un transfert de connaissances et des solutions supplémentaires pour améliorer simultanément les conditions socio-économiques des producteurs dans le monde. La conception et la mise en œuvre des interventions doivent prendre en compte les compromis et synergies potentiels avec d'autres dimensions de la durabilité, telles que le changement climatique, la rareté des ressources naturelles et la sécurité alimentaire.

Dans le **Chapitre 7**, le développement du cadre méthodologique d'indicateurs, les défis de la modélisation et la qualité des données ont été discutés. La discussion a révélé que les trois indicateurs proposés dans ce cadre à savoir: Life-cycle-NUE_N, Life-cycle-NNB_N et NHI_N fournissent une analyse complète de l'utilisation, des flux et des émissions de nutriments dans les chaînes d'approvisionnement mondiales de l'élevage. La discussion a révélé que les chaînes d'approvisionnement de l'élevage jouent un rôle dans le transfert net de la fertilité des sols des prairies aux terres cultivées et dans l' échange international de l'azote contenu dans les aliments pour animaux entre pays, cet azote peut être perdu à travers des déversement non réglementée du lisier dans l'environnement. Le chapitre a examiné les options d'amélioration potentielles, mais a souligné la nécessité de prendre en compte l'effet de rebond lié à l'amélioration du NUE, ce qui pourrait entraîner une hausse de la consommation. Le chapitre a abordé les problèmes de nutriments en liaison avec la durabilité totale du secteur de l'élevage, qui utilise un grand nombre de ressources naturelles telles que la terre, l'eau fraiche, souvent avec un faible rendement et contribue aux émissions anthropologique des gaz à effet de serre. Parce que les chaînes d'approvisionnement de l'élevage sont enracinées dans l'économie et la culture des sociétés. Elles contribuent au développement rural, à l'alimentation humaine, à la balance commerciale, à la gestion des risques et à d'autres résultats pertinents pour le développement, tout en renforçant la résilience et l'adaptation au changement climatique. L'élevage peut également avoir un impact négatif sur ces résultats, par exemple en contribuant aux problèmes de santé publique (régimes alimentaires, zoonoses, résistance antimicrobienne) et en offrant de mauvaises conditions aux éleveurs et aux animaux eux-mêmes. Pour relever les défis liés à l'azote, il faudra prendre en compte les compromis possibles et les synergies avec ces dimensions plus larges de la durabilité (par exemple, l'éradication de la pauvreté, la nutrition, la santé humaine) et cela devra également être fait en conjonction avec d'autres interventions qui traitent de la croissance du secteur de l'élevage. Le chapitre conclut en appelant à une initiative mondiale avec une forte représentation de scientifiques et d'acteurs du secteur de l'élevage pour lutter contre la pollution par l'azote.

Acknowledgements

My PhD research was a fantastic journey of learning and scientific development. The completion of this PhD is a result of the ability to set clear daily goals, celebrate small successes, and learn from failures. This milestone is also due to several people and organisations that supported and helped me to achieve this scientific research.

I want to express my gratitude to the Walsh Fellowship and the Livestock Environmental Assessment and Performance (LEAP) Partnership, and Food and Agriculture Organization of United Nations for funding this project and facilitating my research at FAO.

I would wish to express my sincere appreciation and gratitude to my scientific mentors and role models Prof. Dr Imke de Boer, Prof. Dr Pierre Gerber and Prof. Dr Rogier Schulte for accepting and mentoring me through this transdisciplinary research. Thank you for trusting my independence to work as an external PhD student and for your flexibility to schedule our many remote meetings.

To my promotor Imke, thank you for guidance, advice and scientific insights. Thank you for the fruitful discussion about several issues, for asking me insightful questions that helped me to achieve this PhD research and for reading and editing my manuscript. I am grateful to you and Paul for opening your house and hosting me during my several visits at Wageningen. I enjoyed discovering the Dutch culture and gastronomy and appreciated all our discussions over dinner. You inspired me and shaped my critical thinking and scientific analysis.

To my promotor and daily supervisor Pierre, thank you for arranging my stay at FAO and for the opportunity to contribute to the debate and scientific development to shape the sustainability of the livestock sector. I enjoyed our discussions on Skype or during our many face-to-face catch-ups. Your insights, scientific advice and guidance made this PhD journey more adventurous and beautiful, and helped me to become a perfect scientist. Thank you for your trust in my independence, and for teaching me how to write concisely and short. I am happy to be your first PhD student and congratulate you for your promotion as an extraordinary professor.

To my co-promotor and daily supervisor Rogier, our long and lovely adventures started in Wexford and Kilkenny, Ireland. Thank you for the opportunity to present my PhD project to farmers and livestock stakeholders in Kildalton and for my first TV interview with Duncan Stewart. Thank you for teaching me the crop and soil sciences, which were crucial pieces in nutrient modelling. Your appreciation of the quality and diligence of my work encouraged me to keep focused and always do my best to achieve outcomes at the highest level of science. Our Skype or face-to-face meetings were still fascinating, and I am grateful for your edits and your accuracy in revising my manuscript. Congratulations on your promotion as Professor at Wageningen University.

I would also wish to express my sincere appreciation to my leaders and supervisors at FAO: Berhe Tekola, Henning Steinfeld, and Carolyn Opio. Berhe, thank you for welcoming me in AGA family. Henning, I am grateful for your insights and your fascinating effort to shape the sustainability of the livestock sectors. Carolyn, thank you for your trust and for allowing me to co-lead the development of nitrogen module of GLEAM.

I want to thank my co-authors Evelyne Groen, Mark Dolman, Anne Mottet, Carolyn Opio, Giuseppe Tempio, Alessandra Falcucci, Harinder Makkar, Flavia Casu, and Felix Teillard, for our fruitful collaboration and knowledge sharing along the way.

I want to thank my colleagues at FAO, in GLEAM team and LEAP secretariat: Carolyn, Alessandra, Giuseppe, Juliana, Flavia, Félix, Monica, Giuseppina, and Camillo. Thank you for our brainstorming meetings and your conviviality and knowledge over the last years. I enjoyed our teamwork and our approach to winning together. Félix, thank you for showing me around on my first day at FAO, for being the awesome roommate and colleague, you shared all coding tips and pieces with me, which boosted my modelling. Alessandra and Giuseppe, thank you for your insights on GLEAM and for being patient in my multitude request to extract data and re-run the model. Monica, thank you for design the beautiful maps. Flavia, thank you for your energy, enthusiasm and dedication to support the update of GLEAM. Camillo, thank you for your insights on life cycle assessment and our passionate discussion on LEAP and livestock sustainability. Special thanks go to all scientists and LEAP steering committee members for our collaboration to achieve the scientific consensus on the methods to assess the environmental performance of the livestock.

I want to thank my FAO colleagues in Rwanda: Alexis, Maiga, Jeanne d'Arc, and Otto for facilitating the logistic of my fieldwork and providing me with a desk to work during my visits in Rwanda. Special thanks also go to all dairy farmers in Rwanda who welcomed me in their farms and passionately shared their dairy practices and information with me.

My journey at FAO has been fascinating and formidable thanks to many colleagues who shared their knowledge and time during a meeting, coffee-break or lunch, especially Félix, Giuseppe, Julianna, Pierre, Rubén, Jean Marie Vianney, Viviane, Stephane, Carine, Clarisse, Meshak, Uwe, Nathanael, Nessie, Viola, Maryia, Faustine, Lazare, Rebecka, Renata, Kosuke, Bernardin, Stanley, and Roland. I want to thank Claudia and Ginevra for designing the beautiful cover. Clarisse, thank you for your friendship, for opening your house to me and for supporting me all these years. Pierre and Jean Marie Vianney, thank you for your friendship and conviviality.

I want to thank my colleagues at Animal Production Systems group: Imke, Corina, Aart, Linda, Heelen, Simon, Eddie, Fokje, Ymkje, Lia, Theo, Raimon, Daniel, Charles, Ollie, Wenjuan, Akke, Iris, Laura, Pim, Simon, Kabebe, Salome and all APS PhD colleagues. Despite that, I spent little time in Wageningen; I enjoyed discussing and catching up with all of you. I am happy to be part of this incredible group with a multitude of expertise. Aart, thank you for organising the visit to your home and showing me different farms in your neighbourhood.

To my paranymphs Viviane and Aart, thank you for your support and help to organise the PhD ceremony.

I want to thank my friends Jean-Jacques, Ernest, Fidèle, and Jean Baptiste for your support and encouragement during this PhD journey. Jean-Jacques, thanks for opening your home in Paris to me and for supporting me during my active search for the PhD project.

I want to thank the family of Moise and Clarisse for all the support and our friendship.

I am grateful my parents Pélagie and Jean Damascène, my brothers Jean Pierre, Jean Claude and Roger Pacifique, my cousins and their families: Alphonsine, Clarisse, Pacifique and Jean de Dieu. Thank you for your support along the way. To my parents, living away from you has always been challenging, but I appreciate your understanding and your incredible support and encouragement in my scientific career.

To Joselyne, my wife, you energise and support me each day to be the best human being. Thank you for raising our lovely son Dylan and always boosting my energy to achieve this doctorate. I also thank you for your patience especially during the days, where I focused more on my research and when I was travelling far away for missions or conferences. To my friend and brother Father Eugène, thank you for your friendship and being part of our family. Thanks go to my friends in Rome: Beata, Yvonne, Cadette, Melaine, Patrick, Hibert, Cyusa, Adolphe, Valens, and Crépin.

Last, but not least, I thank all scientists and several individuals who have contributed to this research directly or indirectly but are not mentioned here by name.

About the author

Aimable Uwizeye was born in Karongi - Bwishyura (Rwanda) on December, 22th 1981. He completed his Doctorate in Veterinary Medicine at Cheikh Anta Diop University (*École Inter-États des Sciences et Médecine Vétérinaires de Dakar*) in Senegal (2008). After his veterinary studies, he worked for two years at the Rwanda Animal Resources and Development Authority, under the Ministry of Agriculture and Animal Resources, as professional in charge of veterinary services and livestock development in the Western Province of Rwanda. He graduated with a double Master degrees under the Erasmus Mundus - Agris Mundus Program in sustainable development in Agriculture. His first Master at the University of Catania focused on Livestock Production and Farming Systems. His second Master at Montpellier SupAgro focused on Agronomy and agri-food with specialisation in innovative systems and techniques for sustainable development in Agriculture, focussing on Livestock Production in Sub-Tropical and Tropical Regions. His master thesis was carried out within the Lethbridge Research Centre in Alberta, Canada and focused on the effect of active dried and killed yeast on ruminal acidosis in cattle, resulting in two scientific publications.

After graduation, he started as an external PhD student within the Animal Production Systems group of Wageningen University in collaboration with the Food and Agriculture Organization of United Nations (FAO) and the Teagasc – the Irish Agriculture and Food Development Authority. His PhD was directed at the development of methods and indicators for nutrient use efficiency in global livestock supply chains and financed by the Teagasc Walsh Fellowship Scheme and the Livestock Environmental Assessment Performance (LEAP) Partnership, hosted at Animal Production and Health Division of FAO. During his PhD, he was a technical officer of the LEAP Secretariat, where he provided technical and scientific contributions during the development of the LEAP guidelines to assess the environmental performance of livestock supply chains. At FAO, he co-led the update of the Global Livestock Environmental Assessment Model (GLEAM). He has been an advisor to the towards the establishment of an International Nitrogen Management Systems (INMS) project, Newton-Bhabha Virtual Centre on Nitrogen Efficiency of Whole Cropping Systems and UKRI GCRF South Asian Nitrogen Hub. He has been a member of the Task Force for Reactive Nitrogen and American Geophysical Union. Since completing his PhD research, he has been working as a Livestock specialist at FAO.

Selected Publications

Refereed scientific journals

Uwizeye, A., Gerber, P.J., Opio, C.I., Tempio, G., Mottet, A., Makkar, H.P.S., Falcucci, A., Steinfeld, H., de Boer, I.J.M. 2019. Nitrogen flows in global pork supply chains and potential benefits of feeding swill to pigs. Resources, Conservation and Recycling, 146, 168-179.

Leip, A., Ledgard, S., Uwizeye, A., Palhares, J., Aller, F., Amon, B., Cordovil, C.M.d.S., De Camillis, C., Dong, H., Fusi, A., Helin. J., Hörtenhuber, S., Hristov, A.N., Koelsch, R., Liu, C., Masso, C., Nkongolo, N.V., Patra, A.K. Redding, M.R., Rufino, M.C., Sakrabani, R., Thoma, G., Vertès, F., Wang, Y. The value of manure - manure as coproduct in life cycle assessment. Journal of Environmental Management. 241, 293-304.

Lassaletta, L., Estellés, F., Beusen, A.H., Bouwman, L., Calvet, S., van Grinsven, H.J., Doelman, J.C., Stehfest, E., Uwizeye, A. and Westhoek, H., 2019. Future global pig production systems according to the Shared Socioeconomic Pathways. Science of The Total Environment. 665, 739-751.

Uwizeye, A., Gerber, P.J., Groen, E.A., Dolman, M.A., Schulte, R.P.O., de Boer, I.J.M. (2017). Selective improvement of global datasets for the computation of locally relevant environmental indicators: a method based on global sensitivity analysis. Journal of Environmental Modelling and Software, 96, 58-67.

Uwizeye, A., Gerber, P.J., Schulte, R.P.O., de Boer, I.J.M. 2016. A comprehensive framework to assess the sustainability of nutrient use in global livestock supply chains. Journal of Cleaner Production, 129, 647-658.

Gerber, P.J., Uwizeye, A., Schulte, R.P.O., Opio, C.I., de Boer, I.J.M. (2014). Nutrient use efficiency: a valuable approach to benchmark the sustainability of nutrient use in global livestock production? Current Opinion in Environmental Sustainability, 9, 122-130.

Vyas, D., Uwizeye, A., Yang, W.Z., Beauchemin, K.A. (2014). The importance of yeast viability for reducing the effects of ruminal acidosis in beef heifers during and following an imposed acidosis challenge. Animal Feed Science and Technology, 197, 103-113.

Vyas D., Uwizeye A., Mohammed R., Yang W.Z., Walker N.D., Beauchemin K.A., 2014. The effects of active dried and killed dried yeast on subacute ruminal acidosis, ruminal fermentation, and nutrient digestibility in beef heifers. J Anim Sci. 2014 Feb; 92(2):724-32. doi: 10.2527/jas.2013-7072. Epub 2014 Jan 7.

Vanham, D., Leip, A., Galli, A., Kastner, T., Bruckner, M., Uwizeye, A., van Dijk, K., Ercin, E., Dalin, C., Brandão, M., Bastianoni, S., Fang, K., Leach, A., Chapagain, A., Van der Velde, M., Sala, S., Pant, R., Mancini, L., Monforti-Ferrario, F., Carmona-Garcia, G., Marques, A., Weiss, F., Hoekstra, A.Y. Environmental footprint family to address local to planetary sustainability and deliver on the SDGs. *Submitted.*

Book chapter

Leip, A., Uwizeye, A., 2018. Nitrogen Footprints, Editor(s): Brian Fath, Encyclopedia of Ecology (Second Edition), Elsevier, 2019, Pages 370-382, ISBN 9780444641304, https://doi.org/10.1016/B978-0-12-409548-9.10753-5.

Refereed conference papers

Teillard, F., De Camillis, C., Uwizeye, A. and Opio, C.I. 2016. The LEAP principles for the assessment of livestock impacts on biodiversity. Sustainable value chains for sustainable food systems, In: Alexandre Meybeck and Suzanne Redfern (Eds). Sustainable value chains for sustainable food systems. A workshop of the FAO/UNEP Programme on Sustainable Food Systems. Food and Agriculture Organization of United Nations. Rome, 8-9 June 2016.

Uwizeye, A., Gerber, P.J., Schulte, R.P.O., de Boer, I.J.M. 2016. Assessing three nitrogen use efficiency indicators for pig supply chains in East and Southeast Asia. Proceedings of the 2016 International Nitrogen Initiative Conference, "Solutions to improve nitrogen use efficiency for the world. 4 – 8 December 2016, Melbourne, Australia. www.ini2016.com

Uwizeye, A., Groen, E.A., Gerber, P.J., Schulte, R.P.O., de Boer, I.J.M. 2016. Reducing uncertainty at minimal cost: a method to identify important input parameters and prioritize data collection. Book of Abstracts of the 10th international conference on Life Cycle Assessment of Food. 10th international conference on Life Cycle Assessment of Food, Dublin, 19-21 October 2016.

Uwizeye, A., Tempio, G., Gerber, P.J., Schulte, R.P.O., de Boer, I.J.M. 2014. A framework to assess life cycle nitrogen use efficiency along livestock supply chains. In Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2014), San Francisco, California, USA, 8-10 October 2014 (pp. 1398-1407). American Center for Life Cycle Assessment.

Abstracts in conference proceedings

Uwizeye, A., Gerber, P.J., Opio, C.I., Falcucci, A., Tempio, G., Teillard, F., Steinfeld, H., Schulte, R.P.O., de Boer, I.J.M. 2018. The role of globalizing livestock supply chains in the disruption of global nitrogen cycles. AGU Fall meeting. American Geophysical Union. Washington, D.C. 10-14 Dec 2018.

De Camillis, C., Opio, C.I., Merènyi, M., Uwizeye, A., Diaz, E.A., Brown, D., Clark, H., Castillo-Fernandez, L., de Athayde, A., de Schryver, A. and de Rosa, P., (2016). Livestock Environmental

Assessment and Performance (LEAP) Partnership. In Book of Abstracts of the 10th International Conference on Life Cycle Assessment of Food 2016.

Uwizeye, A., Gerber, P.J., Schulte, R.P.O., de Boer, I.J.M. 2015. Towards a common nutrient use efficiency assessment method for livestock supply chains: a case of study of mixed dairy supply chains in Western Europe. IDF World Dairy Summit 2015. Vilnius, Lithuania.

Gerber, P.J., Mottet, A., Uwizeye, A., Falcucci, A., Tempio, G. 2014. Natural resource use efficiency gap analysis, insights from global modelling. EAAP Conference. Book of Abstract, p. 93.

Mohammed, R., Vyas, D., Uwizeye, A., Yang, W.Z., Beauchemin, K.A., Walker, N.D. 2013. Changes in the relative population size of target ruminal bacteria following a grain-induced challenge in beef cattle receiving viable and nonviable active dried yeast. 2013 Abstract Book. J. Dairy Sci. 96 (E-Suppl. 1):[p. 273]

Uwizeye, A., Mohammed, R., Yang, W.Z., Beauchemin, K.A., Walker, N.D. 2013. Influence of yeast viability on rumen fermentation parameters and nutrient digestibility in beef heifers. 2013 Abstract Book. J. Dairy Sci. 96(E-Suppl. 1):[p. 497]

Vyas, D., Uwizeye, A., Yang, W.Z., Beauchemin, K.A., Walker, N.D. 2013. Importance of yeast viability for reducing the effects of ruminal acidosis in beef heifers during and following an imposed acidosis challenge. 2013 Abstract Book. J. Dairy Sci. 96(E-Suppl. 1):[p. 668]

Uwizeye, A., 2010. Climate change and Animal husbandry. Abstract. Proceedings of the 5th Pan Commonwealth Veterinary Conference. 21-25 March 2011, Accra, Ghana. Journal of Commonwealth Veterinary Association. Special issue July 2011, Vol 27 No.2.

Technical report

FAO (2018). Guidelines for environmental quantification of nutrient flows and impact assessment in livestock supply chains. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO (2018). Nitrogen inputs to agricultural soils from livestock manure. New Statistics. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO (2017). Greenhouse gas emissions and fossil energy demand from pigs supply chains. Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO (2016). Environmental performance of large ruminant supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

Education certificate

Completed training and supervision plan[1]

The Basic Package (3 ECTS)

- WIAS Introduction Course (2013)
- WGS course on "Ethics and Philosophy in Life Sciences" (2014)

International conferences (9 ECTS)

- 6th International Nitrogen Conference, Kampala, Uganda (2013)
- LCA Food conference, San Francisco, USA (2014)
- SETAC Conference, Barcelona, Spain (2015)
- World Dairy Summit, Vilnius, Lithuania (2015)
- LCA Food conference, Dublin, Ireland (2016)
- 7th International Nitrogen Conference, Melbourne, Australia (2016)
- The Joint Nitrogen VJC, New Delhi, India (2017)

Seminars and workshops (5.10 ECTS)

- Symposium, 'Internationale kansen bij verduurzaming van de veehouderij' (2013)
- Establishing agriculture's greeen credentials /Proof & Prospects, Kildalton, Ireland (2013)
- 10th Task Force of Reactive Nitrogen, Lisbon, Portugal (2015)
- OECD/TFRN workshop "The nitrogen cascade and policy - towards integrated solutions", Paris, France (2016)
- Multi-stakeholder Meeting of Global Agenda for Sustainable Livestock, Addis Abeba, Ethiopia (2017)
- 55th Session Working Group on Strategies and Review, UNECE Convention on Long-range Transboundary Air Pollution, Palais des Nations, Geneva, Switzerland (2017)
- Task Force for Reactive Nitrogen, Agriculture and Air Pollution, Geneva, Switzerland (2017)
- Footprint family workshop: Footprints of food production and consumption in the EU, JRC, Ispra, Italy (2017)
- GCP - INI - Global N_2O budget: Regional and global N input datasets and global N_2O, New Orleans, USA (2017)

Presentations (10.5 ECTS)

- Establishing agriculture's green credentials - Proof & Prospects, Kildaton, Ireland, Oral (2013)
- 6th International Nitrogen Conference, Kampala, Uganda, Oral (2013)
- LCA Food conference, San Francisco, USA, Poster (2014)
- 10th Task Force of Reactive Nitrogen, Lisbon, Portugal, Oral (2015)
- World Dairy Summit, Vilnius, Lithuania, Oral (2015)
- OECD/TFRN workshop, N cascade and policy, Paris, France, Oral (2015)
- LCA of Food conference, Dublin, Ireland, oral (2016)
- 7th International Nitrogen Conference, Melbourne, Australia, Oral (2016)
- The Joint Nitrogen VJC, New Delhi, India, Oral (2017)
- Task Force for Reactive Nitrogen Agriculture and Air Pollution, Geneva, Switzerland, Oral (2017)
- GCP-INI - Global N_2O budget New Orleans, USA, Oral (2017)

In-Depth courses (6.1 ECTS)

- "Statistics for the life science", WIAS (23-30 May 2013) (PhD Course)
- "Environmental Impact Assessment from livestock sector", Wageningen University (2015)
- "Climate change and Food Security", FAO (2015)
- "Vulnerability assessment and analysis", FAO (2017)

Scientific discussion groups (1.5 ECTS)

- FAO Climate Change Study cycle (2014 - 2018)

Professional Skills Support Courses (7.5 ECTS)

- Techniques for Writing and Presenting a Scientific Paper, Wageningen University (2014)
- Emotional Intelligence fundamentals, FAO (2015)
- Programming for everybody (PYTHON) by Coursera (2015)
- Essentials of Entrepreneurship: Thinking & Action by Coursera (2015)
- The data scientist's toolbox by Coursera (2016)
- Inspiring and Motivating Individuals by Coursera (2018)

Research Skills Training (13 ECTS)

- Preparing PhD research proposal
- External training period, FAO, Rome, Italy
- Reviewing scientific papers for conferences and journals (IJAS, Sustainability, Agronomy for sustainable development)
- Development of Livestock Environmental Assessment and Performance Partnership guidelines for quantification of environmental impacts from livestock sector (feed, poultry, pig, small-ruminants, large-ruminants, biodiversity, nutrient use, water, soil carbon, feed additives): http://www.fao.org/partnerships/leap/en/ (2013 – 2018)

Didactic Skills Training (1 ECTS)

- Supervision of one MSc Student from Wageningen University (2017)

Organisation of seminars and courses (2 ECTS)

- Session organiser, co-chair and rapporteur: SETAC Conference, Barcelona, Spain (2015)
- Seminar: Food losses, a sustainable animal feed? Rome, Italy (2017)

Membership of boards and committees

- Member of Task Force for Reactive Nitrogen, Expert Panel Nitrogen and Food (2016 - now)

Total: 57.2 ECTS

[1]With the activities listed the PhD Candidate has complied with the educational requirements set by the Graduate School of Wageningen Institute of Animal Sciences (WIAS). One ECTS equals a study load of 28 hours.

Colophon

The research described in this thesis was financially supported by the Teagasc Walsh Fellowship Scheme (Ref: 2012230), the FAO Livestock Environmental Assessment Performance (LEAP) Partnership (GCP /GLO/369/MUL), and Animal Production and Health Division (AGA) of Food and Agriculture Organization of the United Nations.

Cover design by Claudia Ciarlantini and Ginevra Virgili, Animal Production and Health Division, Food and Agriculture Organization of the United Nations, Rome, Italy.

Printed by: ProefschriftMaken | Digiforce