Recycling Plant Nutrients from Waste and By-Products

A Life Cycle Perspective

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Abstract

Chemical fertilisers contribute to greenhouse gas emissions, fossil fuel use, use of non-renewable phosphate rock and a flow of reactive nitrogen to the biosphere, exceeding the planetary boundaries. Recycling of plant nutrients from waste and by-products from society would reduce the use of chemical fertilisers. These plant nutrient sources are also of interest for organic farming, where chemical fertilisers are not allowed, especially organic farms without access to manure.

This thesis assessed the environmental impact of systems recycling plant nutrients from slaughterhouse waste, toilet waste fractions, digested food waste and mussels too small to be used in food production. The methodology used was life cycle assessment (LCA) and the functional unit was production of 1 kg plant-available nitrogen. The environmental impact categories studied were primary energy use, global warming potential (GWP), potential eutrophication and potential acidification. Flow of cadmium to arable soil, use of non-renewable phosphate rock and potential carbon sequestration were also assessed. In addition, additional functions such as phosphorus added to arable soil, energy production, removal of nitrogen and phosphorus from wastewater streams *etc.* were considered. The reference scenario for all comparisons was the production and use of chemical fertilisers.

In general, storage and spreading of the organic fertilisers contributed greatly to potential eutrophication and acidification, except in the case of meat meal fertiliser, which was in a pseudo-stable form. All investigated fertilisers gave rise to goal conflicts as none of the fertilisers reduced the impact for all impact categories studied. The urine fertiliser reduced the largest amount of impact categories and added the least amount of cadmium to arable soil. Meat meal reduced, or had similar results as the reference scenario, for all impact categories except primary energy use and potential eutrophication. For digested food waste, chemical fertiliser use was an environmentally better option for all impacts. Composting gave rise to large nitrogen emissions, thus anaerobic storage was a better environmental option for mussel treatment. Due to the large amount of phosphorus per kg nitrogen in the compost, the reference scenario used the largest amount of non-renewable phosphate rock. A need for applicable methods and data for estimating emissions in LCA of agricultural systems was identified.

Keywords: recycling, waste, organic fertilisers, life cycle assessment, primary energy use, global warming, acidification, eutrophication, cadmium

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Dedication

To Mother Earth

“Oh, Mother Earth,
With your fields of green
Once more laid down
by the hungry hand
How long can you
give and not receive
And feed this world
ruled by greed
And feed this world
ruled by greed“
Neil Young

Contents

List of Publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I Spångberg, J., Hansson, P.-A., Tidåker, P. and Jönsson, H. (2011). Environmental impact of meat meal fertilizer vs. chemical fertilizer. *Resources, Conservation and Recycling* 55, 1078-1086

II Spångberg, J., Tidåker, P. and Jönsson, H. Environmental impact of recycling nutrients in human excreta to agriculture compared with enhanced wastewater treatment (submitted to *Science of The Total Environment*)

III Chiew, Y.L., Spångberg, J., Hansson, P.-A. and Jönsson, H. Environmental impact of recycling digested food waste as a fertilizer in agriculture - a generalized case study (manuscript)

IV Spångberg, J., Jönsson, H. and Tidåker, P. (2013). Bringing nutrients from sea to land - mussels as fertiliser from a life cycle perspective. *Journal of Cleaner Production* 51, 234-244

Papers I and IV are reproduced with the permission of the publishers.

The contribution of Johanna Spångberg to the papers included in this thesis was as follows:

I Planned the paper with the co-authors. Collected data for the calculations and carried out the impact assessment with inputs from the co-authors. Wrote the paper with large inputs from the co-authors.

II Planned the paper with the co-authors. Collected data for the calculations and carried out the impact assessment. Wrote the paper with inputs from the co-authors.

III Planned the paper with the co-authors. Collected some of the data for the calculations and carried out a minor part of the impact assessment. Wrote parts of the paper, with the first author providing the majority of the impact assessment and writing.

IV Did the majority of the planning. Collected data for the calculations and carried out the impact assessment. Wrote the paper with inputs from the co-authors.

Abbreviations

ABP	Animal By-Product
ALCA	Attributional life cycle assessment
Cd	Cadmium
CLCA	Carbon dioxide
CO_2	Global warming potential
GWP	Consequential life cycle assessment
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
N	Nitrogen
P	Phosphorus
WWTP	Wastewater treatment plant

1 Introduction

One of the main concerns regarding the environmental impact of agriculture is nutrient management; to maintain good soil quality and avoid nutrient depletion in soils, to avoid emissions from the production and use of fertilisers and to avoid the use of non-renewable resources in the production of fertilisers. This is especially important as the agricultural sector is predicted to further increase due to estimated global population growth of about 35% by 2050 (UN, 2013).

About half the plant nutrient inputs in European agriculture are provided in the form of chemical fertilisers (Eurostat, 2011a; 2011b). The production of these fertilisers relies on fossil fuels and contributes about 4% of the total emissions of greenhouse gases from Swedish agriculture (Brentrup and Pallière, 2008; Jordbruksverket, 2009; Jordbruksverket, 2012a). About 2% of the total energy use in the European Union (EU) is consumed as direct energy use in agriculture, of which about 50% derives from fossil oil use (Eurostat, 2012a). There is also great indirect energy use in agriculture from use of inputs such as fertilisers, pesticides, animal feed *etc.*, which have been estimated to be larger than the direct energy use in *e.g.* Sweden and United Kingdom (Edström *et al.*, 2005; Defra, 2008a). Fertiliser production also contributes to the flow of nitrogen from the atmosphere to the biosphere, increasing the amount of reactive nitrogen in the biosphere and thus potentially increasing the risk of eutrophication of soil and water (Rockström *et al.*, 2009).

By recycling plant nutrients from waste and by-products, production of chemical fertilisers can be decreased, plant nutrients including micro-nutrients returned to arable soil and the flow of new reactive nitrogen into the biosphere decreased. Within organic farming, it is also important to find other plant nutrient sources, especially for farms without access to manure, as the use of chemical fertiliser is not permitted in organic agriculture (EC, 2008). The largest fraction of nitrogen and phosphorus deriving from agriculture is found in human excreta and in organic waste fractions from households and the food industry (Wivstad *et al.*, 2009). However, the nutrient concentrations in organic fertilisers from waste products are often lower and thus a larger mass of material has to be handled. Organic fertilisers cause emissions of ammonia, nitrogen oxides, methane and nitrous oxide during storage and after spreading, which can contribute to global warming, eutrophication and acidification.

It is therefore a need to assess the environmental impact arising from the management and use of fertilisers deriving from different types of wastes and by-products using a life cycle perspective. Which are the environmental hot-spots from the management and use of such fertilisers and how can the environmental performance be improved?

2 Objectives and structure of the thesis

2.1 Objectives

The main objective of this thesis was to investigate the environmental impacts of recycling plant nutrients from different waste and by-products as fertilisers in agriculture. A life cycle perspective was used in the studies to identify the strengths and weaknesses of these nutrient sources regarding resource use and environmental impact. These aspects are of interest for all agricultural production systems striving to reduce their environmental impact and of special interest for organic arable farming, where chemical fertilisers are not permitted and thus other fertilisers are needed.

2.2 Structure

Three waste fractions from society were studied, namely slaughterhouse waste (**Paper I**), human excreta (**Paper II**) and food waste (**Paper III**) as well as small cultivated mussels, a by-product of seawater treatment (**Paper IV**). The environmental impacts of using these fractions as fertilisers, considering nitrogen and phosphorus content, were assessed and compared with the use of chemical fertilisers. The environmental impacts assessed were; primary energy use, global warming potential, potential eutrophication, potential acidification, flow of cadmium to arable land and use of non-renewable phosphate rock.

Paper I assessed the use of slaughterhouse waste as fertiliser. In the scenario studied, meat meal was produced from slaughterhouse waste, with animal fat as a by-product. The meat meal was then pelleted and used as a fertiliser product and the animal fat was combusted replacing fossil fuel oil. In the reference scenario, the slaughterhouse waste was instead incinerated and chemical fertiliser was produced and used. The geographical location was southern Sweden.

Paper II assessed the use of toilet waste fractions as fertiliser. In one scenario, source-separated urine and faeces (*e.g.* blackwater) were assessed as fertiliser, while in another scenario only the urine fraction was used as fertiliser. In the reference scenario, chemical fertiliser was produced and used. All scenarios included advanced removal at a wastewater treatment plant of nitrogen and phosphorus in wastewater fractions not used as fertilisers. Greywater was not included in the study. The geographical location was the periphery of Stockholm, Sweden.

Paper III assessed the use of digested food waste as fertiliser. The biogas produced replaced fossil vehicle fuel. In the reference scenario, the food waste was instead incinerated and the heat produced replaced Swedish district heating. Chemical fertiliser was produced and used in the reference scenario. The geographical location was central Sweden.

Paper IV assessed the use of mussels cultivated on the east coast of Sweden as fertiliser. The mussels in that region grow too small to be used in the food industry, due to the low salinity of the water. In the reference scenario, nitrogen and phosphorus removed with the uptake of the mussels were instead removed at a wastewater treatment plant and chemical fertiliser was produced and used.

The background to the research topic is presented in Chapter 3, while background to the methodology used is presented in Chapter 4. In Chapter 5, methodology used in **Paper I-IV** is described with a system description of the studies. The main findings are also presented in Chapter 5, which concludes with a summarising section comparing the different fertilisers in which the results are presented per kg plant-available nitrogen spread on arable soil. An overall discussion follows in Chapter 6. *Figure 1* illustrates the structure of the thesis relative to **Papers I-IV**.

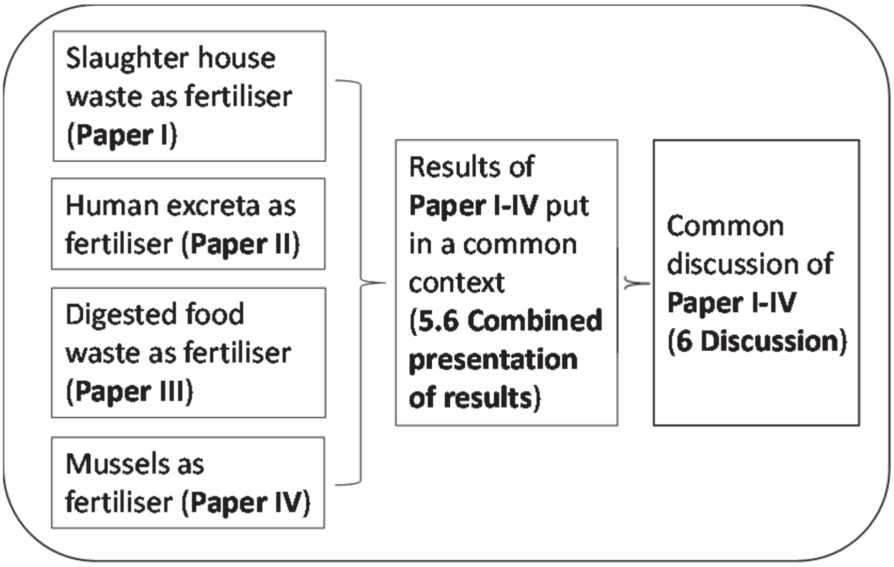

Figure 1. Structure of the thesis relative to **Papers I-IV**.

3 Background

3.1 Agriculture and the environment

According to Rockström *et al*. (2009), the planetary boundaries have been exceeded in a number of environmental categories important for a sustainable world, such as loss of biodiversity, increased climate change and excessive inputs to the nitrogen cycle. Agriculture is a sector with important impacts relating to all these categories.

3.1.1 Resource use in agriculture

Of the total energy use in the EU, about 2% is consumed as direct energy use in agriculture, of which about 50% derives from fossil oil use (Eurostat, 2012a). Apart from the direct energy use, there is also great indirect energy use in agriculture from use of inputs such as fertilisers, pesticides, animal feed *etc*. There are no data available on this indirect use at EU level, but estimates show that the indirect energy use is similar or larger than the direct energy use, depending on the farming system (Edström *et al.*, 2005; Defra, 2008a). Estimates made for Swedish agriculture show that the indirect energy use is about 13% larger than the direct energy use (Edström *et al.*, 2005). Of the indirect energy use in agriculture, production of chemical fertilisers is the main input, contributing about 50-60% of the total indirect energy use (Edström *et al*., 2005; Defra, 2008a).

Resources used in agriculture are fossil fuels and minerals, such as phosphate rock. Fossil fuels are used in most operations on the farm, such as field operations, drying of crops and heating of animal housing facilities. Phosphate rock is an essential resource in the production of phosphorus fertilisers and is a non-renewable resource. The main phosphate rock mines are located in China, the United States, Morocco, West Sahara and Russia. The reserves of a certain mineral indicate the amount of that mineral it is feasible to produce under current economic and technical conditions (USGS, 2013). If current rate of production is assumed also for the future, the phosphate rock reserves are estimated to be available between 90 and 400 years (Vaccari, 2009; Van Kauwenbergh, 2010; USGS, 2013), while estimates considering potential changes in demand estimate the availability to 60 to 130 years (Cordell and White, 2011). In addition, many actors are concerned that the quality of the phosphorus will decrease and thus become more costly to produce. Potassium, like phosphorus, derives from mine reserves in the form of potash (most commonly potassium chloride), which contains water-soluble potassium. Estimated lifetime of the potassium reserves, at current production rates, is about 280 years (USGS, 2013).

3.1.2 Emissions from agriculture

Greenhouse gas emissions are projected to increase with increasing global population (van Beek *et al*., 2010). This is seen as a major problem globally, with the EU, having committed within the Kyoto agreement to reduce its greenhouse gas emissions by 20% by 2020 compared with 1990 levels (EC, 2014), and the Swedish Parliament having adopted a vision of a climate-neutral country by 2050 (Sveriges Regering, 2012). According to National Inventory Reports, agriculture contributes 10% of the greenhouse gas emissions within the European Union and 13% of the Swedish emissions (EEA, 2012; SEPA, 2012) (*Figure 2a*). These reported emissions from agriculture include only emissions from enteric fermentation, manure management and managed soils. If emissions from organogenic soils, chemical fertiliser production, fossil fuel use and imported fodder also were to be included, Swedish agriculture would cause about 19% of the total greenhouse gases reported (*Figure 2b*). Emissions from fossil fuel use, chemical fertiliser production, manure management and managed soils are all connected to fertiliser use to some extent.

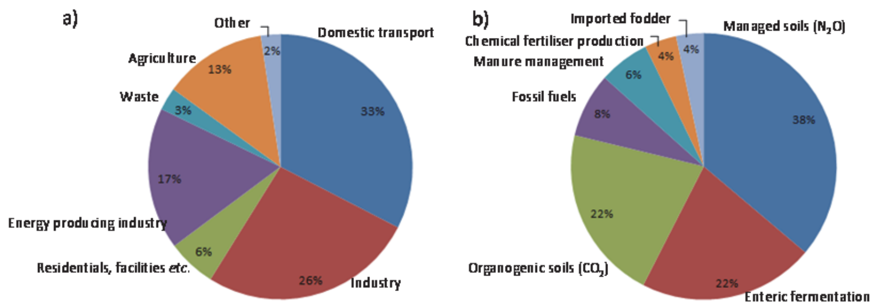

Figure 2. Contributions to a) global warming potential (GWP) from the different sectors of the Swedish society according to the Swedish National Inventory Report (SEPA, 2012) and b) to GWP within the Swedish agriculture (Brentrup and Pallière, 2008, Jordbruksverket, 2009; Jordbruksverket, 2012a; Jordbruksverket, 2013a).

Agriculture can contribute to carbon sequestration and can thus act as a sink for some greenhouse gas emissions, by building up the carbon pool of the soil. This function could be expanded by *e.g*. converting arable land to forestland or grassland, restoring wetlands, adding organic materials with fertilisers to soil, using crop rotations including diverse crops or using cover crops (Freibauer *et al.*, 2004; Lal, 2008). However, it must be borne in mind that sooner or later the level of soil organic matter reaches a certain equilibrium level, thus limiting further carbon sequestration by the soil (El-Hage Scialabba and Müller-Lindenlauf, 2010).

Agriculture is the main contributor of ammonia emissions within Europe, creating over 90% of the total emissions (Eurostat, 2012b). Ammonia is a gas that cause both acidification and eutrophication and which derives mainly from manure management in agriculture. Ammonia emissions and losses of nitrogen and phosphorus from arable soils to waters are the greatest contributors to eutrophication within agriculture. For example, there have been tremendous problems with eutrophication of the Baltic Sea in northern Europe owing to increased nutrient loads between the 1950s and 1980s. These loads have stabilised in recent years, but are still a major concern. Agriculture contributes an estimated 40% of the total anthropogenic Swedish net inputs of nitrogen and phosphorus to the surrounding seas (SEPA, 2008). Other nutrient-related problems are occurring in other parts of Europe, *e.g.* German Bight, the Wadden Sea *etc*. (EEA, 2001).

Another agricultural activity having an impact on acidification is the combustion of fossil fuels which emits nitrogen oxides, causing acidification and eutrophication, and sulphur oxides, causing acidification. Both these emissions can also give rise to photochemical ozone.

Agricultural soils can be a sink for heavy metals, with the main sources being deposition and addition of fertilisers (de Vries *et al*., 2002; Nicholson *et al*., 2003). Of these heavy metals, cadmium is of great concern as intake can cause renal and skeletal problems in humans, with one of the major intake routes being via food (mainly cereals and root crops) (EFSA, 2009). Monitoring has shown that parts of the Swedish population have cadmium levels in their urine that are at or above the levels which can potentially cause skeletal or renal effects (KemI, 2011). Historically, the main source of cadmium to arable soil was application of chemical fertilisers, but today the main source is atmospheric deposition (KemI, 2011). Cadmium in chemical fertilisers derives from phosphate rock, with sedimentary sources, the main source of phosphate rock globally, containing significantly higher concentrations than volcanic sources, *e.g.* from Russia, Finland and South Africa. The Swedish Chemicals Agency has emphasised the need to lower the national limit value for cadmium in chemical fertilisers substantially, from 100 to 12 mg per kg phosphorus, in order to reduce these health risks (KemI, 2011). There are currently no EU regulations concerning cadmium in fertilisers, but there are proposals to set a limit of 46 mg cadmium per kg phosphorus (EC, 2011). Studies have shown that the median content of cadmium in phosphate fertilisers sold in Europe is 87 mg per kg phosphorus (Nziguheba and Smolders, 2008).

3.2 Plant nutrients in agriculture

For plants to grow optimally, essential nutrients are required. Among these nutrients, nitrogen, phosphorus and potassium are needed in greater amounts and are thus called macro-nutrients. However, other macro-nutrients such as calcium, magnesium and sulphur are also needed, as are a number of micro-nutrients such as boron, iron, manganese, copper, zinc, molybdenum, chlorine *etc*. Although these are not necessarily essential to all plants, all are essential to some (IFA, 2013). Factors such as the geographical location of the soil, the soil type and its acidity (pH) determine the extent to which nutrients within the soil are available to plants (Barber, 1995). For example, a soil with high content of clay or organic matter holds water and nutrients much better than a sandy soil. Furthermore, even though nutrients are presented in the soil, the supply to plants is limited by the rate at which the soil can release these nutrients and the extent to which the nutrients are removed by the harvested crops. Thus, in all farming systems, it is highly important to have good nutrient management so as to maintain soil fertility and provide a good balance of required nutrients in order to obtain good crop yields (Watson *et al*., 2002; Dawson and Hilton, 2011). A large amount of the nutrients used in agriculture leave the farm with crops supplied to external food and feed markets, but there are also great internal flows on the farm in the form of manure, crop residues and feed (Wivstad *et al.*, 2009). Due to the large amounts of macro-nutrients needed by crop plants, adding these to the soil is of the greatest concern for the farmer. Traditionally, crop rotation and regular fallow periods, together with spreading of animal manure, allowed the soil to recover some of its fertility, but today the main method used to restore nutrients in soil is the application of chemical fertilisers (EC, 2013).

3.2.1 Plant nutrient inputs to agriculture

As part of the ‘Green Revolution’ beginning in the 1960s, the production and use of chemical fertilisers increased (Matson *et al.*, 1997). Today, about 45% of total nitrogen and phosphorus inputs within the EU originate from chemical fertilisers (Eurostat, 2011a; 2011b). The other nitrogen inputs come from gross manure input (about 39%), atmospheric deposition (about 8%), biological nitrogen fixation (about 7%) and organic fertilisers other than manure (about 1%) (Eurostat, 2011a). The other phosphorus inputs come from manure input (about 50%) and from organic fertilisers (almost 5%) (Eurostat, 2011b). Of the plant-available nitrogen added with fertiliser inputs to Swedish agriculture, about 76% is from chemical fertilisers, about 23% from manure and about 1% from organic fertilisers other than manure (not including atmospheric deposition or biological fixation) (SCB, 2012a). In addition, of the gross nitrogen input to Swedish soils, about 12% is added by biological nitrogen fixation (Eurostat, 2011a). Of the plant-available phosphorus added with fertiliser to Swedish agriculture, about 27% is from chemical fertilisers, 71% from manure and about 2% from organic fertilisers other than manure (SCB, 2012a). Potassium is an important plant nutrient, especially for grass and legume-dominated systems. However, it is a less highly prioritised plant macro-nutrient, both because it is often not the limiting nutrient in the farming system, as significant amounts are released to Swedish soils by mineral weathering, and because it is less harmful to the environment.

3.2.2 Chemical fertiliser production

Of the nitrogen fertiliser products consumed in Europe, 47% is ammonium nitrate and calcium ammonium nitrate, while some ammonia nitrate is used in the different NP and NPK fertiliser compounds commercially available (Fertilizers Europe, 2013). The corresponding figure for Sweden is around 60% (Jordbruksverket, 2012a). Ammonium nitrate is produced from the reaction of ammonia with nitric acid. Ammonia is produced by fixation of nitrogen from the air, requiring energy, with the major energy source used being natural gas, which also emits carbon dioxide (IFA, 2009). Furthermore, processes in nitric acid production cause nitrous oxide emissions (Brentrup and Pallière, 2008). In total, nitrogen fertiliser production contributes about 1% of total global greenhouse gas emissions (Brentrup and Pallière, 2008). Globally, chemical fertiliser production is the main contributor to nitrogen fixation, with Rockström *et al.* (2009) recommending a decrease of about 75% in the current level of nitrogen fixation to reach levels within safe planetary boundaries.

The phosphorus in chemical fertilisers is derived from phosphate rock, which is mined as discussed in section 1.1. Production of phosphorus fertilisers consumes about 2% of the total energy used and contributes about 1% of the GWP caused by average European nitrogen fertiliser production (Jenssen and Kongshaug, 2003; Bellarby *et al.*, 2008).

3.2.3 Plant nutrient supply on arable farms without livestock

Due to technological developments and an increased dependency on global market conditions, specialisation in agriculture has increased over recent decades (Naylor *et al.*, 2005; Defra, 2008b). This has led to a change from farm systems with a mixture of livestock and crops to an increased proportion of farm holdings specialising in only one livestock or crop. In the EU, 40% of agricultural holdings specialise in arable farming, 22% specialise in livestock and 38% are mixed farms, *i.e.* where neither livestock nor crop production dominates the activities (Eurostat, 2010). One major factor enabling such specialisation was the introduction of chemical fertilisers.

Apart from using chemical fertilisers, arable farms without access to manure can also use green manure and crop rotations including nitrogen-fixing crops, *e.g.* a legume with nitrogen-fixing bacteria (Watson *et al.*, 2002). Green manuring involves growing plants, most commonly a nitrogen-fixing green manure crop, that are subsequently incorporated into the soil to increase the organic matter content and add plant-available nutrients to the soil. Growing plant species with a deep root system is also good for supplying the upper soil layers with nutrients 'mined' from deeper layers. However, for all these fertilisation strategies addition of phosphorus is needed in the long run, at least if more products, *i.e.* more phosphorus, are transported away from the farm than to the farm.

Finding alternative plant nutrient sources to chemical fertilisers and manure is of particular interest in organic arable farming, as chemical fertilisers are not permitted in organic production.

3.3 Organic farming

About 1% of the agricultural land worldwide (including arable land, permanent crops and pastures) is under organic production (FAO, 2010; Willer and Kilcher, 2012). The countries with the greatest proportion of organic agricultural land globally are Australia, Argentina and the United States (Willer and Kilcher, 2012).

3.3.1 General principles and regulations within organic farming

Most regulations globally on organic production are grounded in the basic principles of organic farming defined by IFOAM (International Federation of Organic Agriculture Movements) (Organic World, 2013). IFOAM is an umbrella organisation of the organic world with the mission to lead, unite and assist the organic movement (IFOAM, 2013a). The basic principle stated by IFOAM is that "production should be based on ecological processes, and recycling, the systems should fit the cycles and ecological balances in nature and by designing the farming system, establish habitats and maintain genetic and agricultural diversity" (IFOAM, 2013b).

On EU level, the legal framework on organic production and labelling is provided by Council Regulation No 834/2007 (EC, 2007), with detailed rules on production, controlling and labelling in Commission Regulation No 889/2008 (EC, 2008). These EU regulations define organic production as "an overall system of farm management and food production that combines best environmental practices, a high level of biodiversity, the preservation of natural resources, the application of high animal welfare standards and a production method in line with the preference of certain consumers for products produced using natural substances and processes" (EC, 2007). Among other things, it is stated that mineral nitrogen fertilisers are not permitted (EC, 2007).

The EU legislation on organic production acts as a common minimum standard, while member states can enact their own stricter standards. In Sweden, KRAV is the largest labelling organisation for organic production, and is also an active member of IFOAM. Compared with the EU legislation the KRAV standards are stricter in some areas (KRAV, 2013). Farmers in Sweden can receive financial compensation from the government for farming under organic principles (Jordbruksverket, 2013a).

In many ways, organic agriculture can be viewed as a legalised form of agriculture striving for environmental sustainability. Many of the principles of organic farming also apply for sustainable agriculture as defined by the European Union (EU) and Swedish authorities. In the EU, 28 agri-environment indicators are stated as a tool to assess the sustainable development of agriculture (EC, 2006a). These indicators include *e.g.* area under organic farming, chemical fertiliser consumption, energy use, specialisation of agriculture, greenhouse gas emissions *etc*. A Swedish report on indicators for sustainable agriculture, issued jointly by the Swedish Environmental Protection Agency (SEPA) and the Swedish Board of Agriculture (Jordbruksverket), includes *e.g.* plant nutrient balances, soil fertility, use of pesticides and herbicides, energy use, greenhouse gas emissions and waste management (SCB *et al.*, 2012).

3.3.2 Organic production in Europe and in Sweden

Of the total agricultural land area utilised within the EU, 4.1% is now under organic production, with the total area under organic production increasing by 6-7% annually between 2006 and 2008 (Eurostat, 2009).

In Sweden, the area under organic production is currently about 425 000 hectares, corresponding to 14% of Sweden's agricultural land area (SCB, 2013a). The increase in certified organic land (including land in the qualifying period for financial compensation) was 76% between 2005 and 2009 (Sveriges Riksdag, 2010). The Swedish Government set up a number of goals on the development of organic production to be reached by 2010 (Sveriges Regering, 2006). These goals are currently under evaluation before new goals are proposed. Two of the goals for 2010 were for the area under organic production to be increased to 20% of total agricultural area and for 25% of total public food consumption to be organically produced. Neither of these two goals was fully met (Sveriges Riksdag, 2010), although organic production is likely to increase further in Sweden.

3.3.3 Plant nutrient supply on organic arable farms without access to manure

According to IFOAM (2013b), organic management should be adapted to local conditions, where "inputs should be reduced by reuse, recycling and efficient management of materials and energy in order to maintain and improve environmental quality and conserve resources". The EU legislation states that "organic farming should primarily rely on renewable resources within locally organised agricultural systems. In order to minimise the use of non-renewable resources, waste and by-products of plant and animal origin should be recycled to return nutrients to the land" (EC, 2007). Furthermore, on plant nutrient management the EU legislation 834/2007 states that "the fertility and biological activity of the soil shall be maintained and increased by multiannual crop rotation including legumes and other green manure crops, and by the application of livestock manure or organic material, both preferably composted, from organic production" (EC, 2007). If the nutritional needs of the plants cannot be met through these measures, fertilisers listed in Annex I of EU regulation 889/2008 (EC, 2008) can be used, *e.g.* manures from non-organic production, mushroom culture waste, guano, blood meal, fish meal. Approved fertilisers in the KRAV standard follow the EU regulations except that guano and manure from genetically modified animals are not permitted (KRAV, 2013). The KRAV standard also restricts the amount of heavy metals that can be added to arable soil, *e.g.* 0.75 g of cadmium per hectare and year (KRAV, 2013).

On the farms in Sweden that receive environmental compensation for organic production, 91% of the nitrogen input is from manure and 9% from other approved fertilisers. For phosphorus, the corresponding figures are 87% and 13% (SCB, 2012a). In a study which drew up plant nutrient balances for Swedish farms, it was found that the nitrogen surplus was about 17% lower for organic arable farms than for conventional arable farms (Wivstad *et al.*, 2009). This was mainly due to lower intensity of the organic production, *e.g.* lower inputs of external plant nutrients. For phosphorus, the nutrient balance results showed the opposite to those for nitrogen, with organic arable farms having a significantly greater phosphorus surplus than the corresponding conventional farms (Wivstad *et al.*, 2009). This could have been due to a short-term increase in consumption of organic fertilisers with high phosphorus content owing to temporarily low prices, because other studies report a decreasing trend in the phosphorus content of organic arable soils (Løes and Øgaard, 1997; Stockdale *et al.*, 2002; Gosling and Shepherd, 2005). In general, organic arable farms cultivate a higher percentage of ley and green manure than conventional arable farms in order to maintain the fertility of the soil (Wivstad *et al.*, 2009). A common crop is ley with nitrogen-fixing legumes, increasing the nitrogen input to the soil. The current trend among organic arable farms in Sweden is for the area of forage and seed ley to increase and the area of green manure ley to decrease (Wivstad *et al.*, 2009).

In Sweden, as in many other Western countries, the soils in many areas are already rich in phosphorus due to excessive application of fertilisers in the past (Barberis *et al.*, 1996; Gosling and Shepherd, 2005). This has made phosphorus inputs a less significant problem than nitrogen inputs for many organic arable farms, although the soil phosphorus reserves will not last forever and sustainable phosphorus sources are required for future use. Existing phosphorus fertilisers used in organic farming, apart from manure, are products based on different by-products of animal origin, such as meat and bone meal, and by-products from the starch and yeast industry (Jordbruksverket, 2012b). As the major nitrogen input to an organic arable farm without manure is through nitrogen-fixing crops, it is highly important to reduce the losses of nitrogen at the transition from ley or green manure to crop cultivation. In this context, timing and season are of particular importance (Thorup-Kristensen *et al.*, 2003; Olesen *et al.*, 2009). Due to the difficulties with timing of plant-available nitrogen supply, nitrogen has been shown to be the most limiting nutrient in most organic arable farming systems (Torstensson, 1998; Doltra *et al.*, 2011). The nitrogen fertiliser products, other than manure, approved for organic farming are animal by-products and other products based on by-products from the food industry, such as molasses from the sugar industry, vinasse from the yeast industry and by-products from ethanol production (Jordbruksverket, 2012b).

In conclusion, it is a great challenge for organic arable farms to compensate for export of nutrients from the farm. Effective plant nutrient management is obviously key to sustainable nutrient input. This is also one of the major challenges for organic production to reduce its environmental impact, as careful management of plant nutrients reduces the emissions of ammonia and nitrous oxide (El-Hage Scialabba and Müller-Lindenlauf, 2010).

3.4 Potential plant nutrient sources other than chemical fertilisers and manure

There is a range of products from nature and industry that could be potential plant nutrient sources for farming. Major flows of nitrogen and phosphorus from nature and society are shown in *Table 1*.

Table 1. *Major flows of nitrogen and phosphorus produced in Swedish urban society (and from potential mussel production on the east coast of Sweden)*

Substrate	Nitrogen (ton)	Phosphorus (ton)
Urine[1]	37 160	3 040
Faeces[1]	5 070	1 690
(Approved sewage sludge[2])	(7 810)	(4 890)
Household food waste[3]	6 520	1 110
Non-household food waste[3]	1 250	310
Slaughterhouse waste[4]	1 820	1 220
Other by-products from food industry[5]	6 580	1 790
Ash from incineration of biofuels[6]	0	7 500
Potential mussel production[7]	480	30

[1]Wivstad *et al*. (2009).
[2]Sludge with approved levels of heavy metals (SCB, 2012b).
[3]Wivstad *et al*. (2009). "Non-household food waste" including food waste from restaurants and large-scale catering.
[4]Wivstad *et al*. (2009). Including slaughterhouse waste from bone meal production and other ABP Category III materials.
[5]Wivstad *et al.* (2009). Including by-products from the sugar industry, distillers, breweries, milk industry *etc*. About 87% of the nitrogen and 72% of the phosphorus from this section are recycled as fodder.
[6]SEPA (2013). It should be noted that a large part of this ash is from mixed fuel combustion and thus the quality of the ash varies widely. About 900 ton of these is from incineration of ABP.
[7]Lindahl (2010).

Due to the fact that a large portion of the nutrients leaving agriculture accompany food products for human intake and thus end up in human excreta, urine and faeces contain the largest flows of nitrogen and phosphorus within society (*Table 1*). As the majority of the urine and faeces produced in society are mixed with other sewage fractions entering wastewater treatment plants, they become contaminated by *e.g.* industry wastewater and stormwater, and a major proportion of the nitrogen is lost during treatment and with the effluent. The remaining nitrogen and phosphorus are found in the sewage sludge. To improve nutrient recycling, the Swedish Farmers Union (LRF), among others, is promoting the installation of source-separating systems, where urine and faeces can be collected separately under more controlled forms. The urine and faeces fractions are approved for use as fertiliser in conventional production in Sweden, but not in organic production (EC, 2008; KRAV, 2013).

Sweden has high ambitions for increasing its biogas production and has set the target that by 2018, 40% of all food waste must be treated in such a way that nutrients and energy are recovered (Sveriges Regering, 2012). The digestate produced during biogas production should thus be used as a fertiliser. The potential of slaughterhouse waste is difficult to estimate precisely, as the use of this waste depends on current regulations and demand from the pet food market. Category III Animal By-Products (ABP) are allowed in pet food production, which is a more economically favourable option than incineration of the waste. Even so, a large proportion of slaughterhouse waste is currently incinerated, mainly in the form of the biofuel Biomal (Linderholm and Mattsson, 2013). Many of the by-products from the food industry are already recycled back to Swedish agriculture as animal feedstuffs (Wivstad *et al*. 2009). The potential of ash from incineration of biofuels as a fertiliser depends on the quality of the ash (Linderholm and Mattsson, 2013).

There are a number of challenges with using organic waste and by-products as fertiliser in agriculture (*Figure 3*). For example treatment techniques which ensure good fertiliser hygiene need to be developed and logistics chains from production site to arable land need to be established, including acceptance of the products by farmers and by consumers buying the agricultural products. Development of spreading equipment is important if commonly available equipment is not suitable. Lastly, all processes involved need to be economically sustainable. However, the emphasis of this thesis was on assessing the environmental impacts of using organic waste and by-products as fertilisers in agriculture.

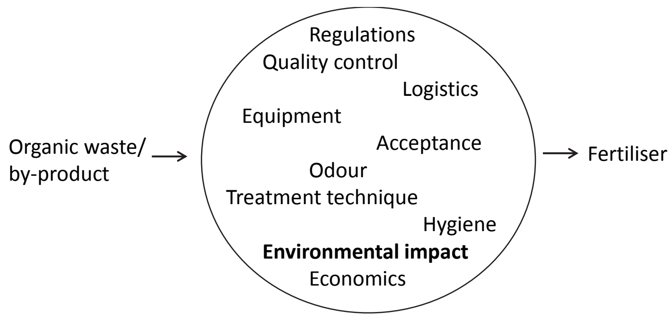

Figure 3. Challenges involved in using organic waste and by-products as fertiliser.

4 LCA methodology

Life cycle assessment (LCA) is a quantitative method which assesses the environmental impacts of a product or service over a life cycle. The concept of life cycle thinking is to consider all relevant aspects in the whole life of the product or service, from extraction of the resources for production to the end-of-use phase, *i.e.* disposal phase (Baumann and Tillman, 2004). The purpose of a LCA can be to identify environmental hot-spots in the production of a product or the use of a service, to market products or services, to compare the environmental impact of different products and services and for policy making and planning (ISO, 2006a). In order to make relevant comparisons between different LCAs, it is important that they follow the same structure and consider the same delimitations. The method is standardised according to the international standards ISO 14040 and ISO 14044 (ISO, 2006a; 2006b).

4.1 Basics of LCA

The different phases of a LCA are described in *Figure 4*. One of the most important steps in conducting a LCA is the goal and scope definition, including a clear goal of the study, well-defined level of detail and appropriate system boundaries (Ekvall *et al*., 2005; ISO, 2006a). The functional unit, which is the reference unit to which all flows and data are related, *e.g.* one kilo of a product, should also be defined in this phase. In the inventory phase, the necessary empirical data are collected. In the impact assessment step, these data are classified into different impact categories, *e.g.* global warming potential and potential acidification, and characterised according to the relative contribution of each emission or resource use. Equivalence factors, *e.g.* characterisation factors, based on science are used to convert the data collected into single values within the different impact categories (Baumann and Tillman, 2004). In the final part of the LCA, interpretation, the results are interpreted and significant issues identified. The work is carried out in an iterative way as new insights are gained during the work process (ISO, 2006a).

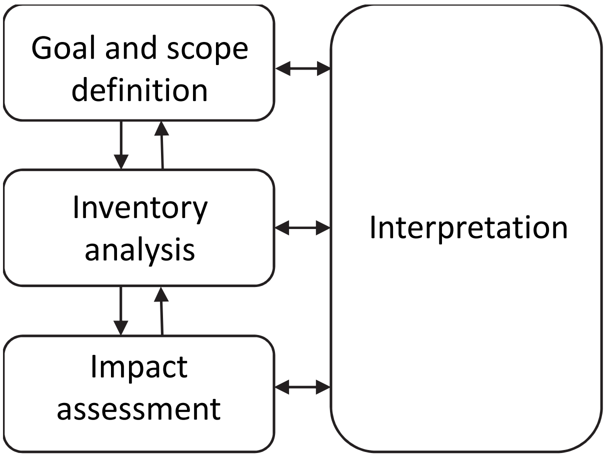

Figure 4. The different phases of a life cycle assessment (LCA) (ISO, 2006a).

The results can be presented as midpoint or endpoint results, where midpoint means stopping at the impact categories calculated and endpoint means weighing the impact categories together to one result and relating this to *e.g.* human health, where the endpoint can be Daily Adjusted Life Years (DALY) (EC, 2010). However, endpoint results are more uncertain and are less frequently calculated (Bare *et al*., 2000).

There are different approaches to LCA, the two most common being consequential LCA (CLCA) and attributional LCA (ALCA) (EC, 2010). In CLCA, a change-orientated perspective is adopted in the choice of data and in determining the effect of the products produced, following market mechanisms. Marginal data are used, *e.g.* for electricity the source chosen is what would be produced if the electricity use increased within a chosen region (EC, 2010). An ALCA strives to assess the specific impact of a product or service, using relevant data representing the existing or forecasted surrounding systems, whether a past, current or future production system. Average data are used on *e.g.* technique performance, electricity use *etc.* (Ekvall *et al.* 2005; EC, 2010). When to use which of these LCA approaches, and how, depends mainly on the goal of the study and this issue has been widely discussed in the LCA community (Finnveden *et al.*, 2009; Earles and Halog, 2011; Zamagni *et al.*, 2012). In general, CLCA is preferable for hypothetical studies on new products, for analysis of future scenarios or for policy making, while ALCA is preferable used for finding hot-spots in a production process, for environmental labelling or for comparing existing products (Baumann and Tillman, 2004;

Finnveden *et al.*, 2009). The main objection to use ALCA approach in agricultural LCAs is that it does not consider crop supply and the change in demand for other crops, land constraints, land transformation *etc.* (Schmidt, 2008).

4.2 LCA and multi-functionality

When a system or service provides more than one function in a life cycle assessment, the issue of multi-functionality and the question of how to share the environmental burden between the functions arises. These issues are especially complex when assessing reuse and recycling (EC, 2010).

Allocation by partitioning is one solution to multi-functional processes, where the burden is divided between the different products produced by *e.g.* physical or economic properties. According to the ISO standards, allocation should be avoided when possible (ISO, 2006b). This can be achieved by more refined data collection, dividing the unit process into multiple sub-processes so that the different outputs can be separated. As most multiple outputs from a production system depend on each other this can be difficult, and in these cases system expansion should be used (ISO, 2006b). System expansion can be done by expanding the system boundaries and include additional functions, so that the compared systems fulfills the same multiple functions (*Figure 5a*). Another variant of system expansion is done by subtracting alternative systems fulfilling the same functions as to reduce the functional units of the system (*Figure 5b*) (Heijungs and Guinée, 2007; EC, 2010). Substitution often leads to negative inventory flow and sometimes even overall negative results for impact parameters (Guinée *et al.*, 2002; EC, 2010). Some LCA practioners argue that system expansion should not be applied in ALCA, as the substituted activities actually do not occur and are also speculative and uncertain, and therefore more appropriate in a consequential approach (Heijungs and Guniée, 2007; Brander and Wylie, 2011). Others argue that system expansion can be used in ALCA if average data are used (Finnveden *et al.*, 2009).

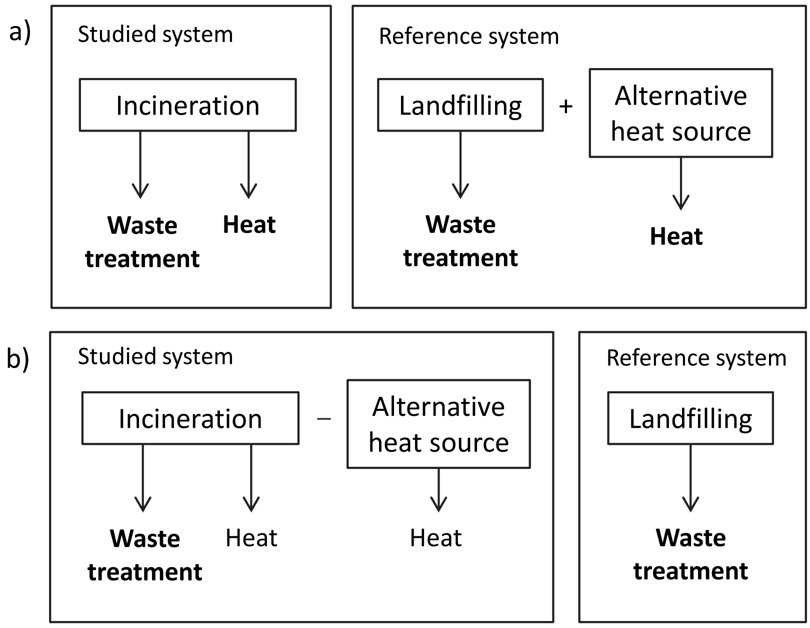

Figure 5. Illustration (after Finnveden, 1999) of handling multi-functionality in life cycle assessment (LCA) of waste treatment by a) system expansion with system enlargement and b) system expansion with subtracted function. The functions studied are noted in bold text.

As the results of a LCA study are affected by the allocation method chosen, transparency is highly important so the reference flow and how allocation is handled must be thoroughly described (Finnveden, 1999; Winkler and Bilitewski, 2007). A need for improved guidance on solving multi-functional systems in LCA has been noted by a number of LCA practitioners (Heijungs and Guniée, 2007; Lundie *et al.*, 2007; Zamagni *et al.*, 2012).

4.3 LCA and fertiliser use in agriculture

Some of the most relevant, and commonly used, environmental aspects associated with agricultural production in LCA are GWP, primary energy use, land use, depletion of abiotic resources, potential eutrophication and potential acidification. In addition to these categories, toxicity, water use, biodiversity, land use change and soil fertility should be included to cover all environmental impacts from agriculture, but are often excluded due to lack of methodological consensus, priority and time (Brentrup *et al.*, 2004; Röös *et al.*, 2013).

Activities connected with the environmental impact of fertiliser use include the production of fertilisers, depletion of abiotic resources such as phosphorus, collection and storage systems for organic fertilisers, emissions from soil after spreading of fertilisers, spreading operations, emissions of heavy metals such as cadmium to soil and effects on soil quality (Cederberg and Mattsson, 2000; Brentrup *et al.*, 2004). Finding data on processes such as fuel consumption during fertiliser spreading and the amount of heavy metals added with the fertiliser is quite straight-forward, while finding data on emissions from biological processes can be more difficult. Assessing the environmental burden of gaseous emissions, *e.g.* ammonia, methane, nitrous oxide and other nitrogen oxides emissions, occurring at different stages of decomposition of organic materials is particularly difficult, as the emissions are highly variable (Brentrup *et al.*, 2001; Payraudeau *et al.*, 2007).

According to the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol, participating parties are required to submit national inventories of anthropogenic greenhouse gas emissions on an annual basis (UNFCCC, 1998). Due to this, comprehensive guidelines and generic methods on estimating the greenhouse gas emissions from agriculture have been developed (IPCC, 2006). Spreading nitrogen fertilisers in the field cause direct nitrous oxide emissions. According to IPCC (2006), the default value for these direct nitrous oxide emissions is set to 1% nitrous oxide nitrogen per amount of total nitrogen added to soil. Volatilised ammonia and nitrogen oxides also causes indirect nitrous oxide emissions which are estimated using a default value, the same percentage as for the direct emissions, based on volatilised nitrogen becoming nitrous oxide nitrogen (IPCC, 2006). Emission factors considering more regional conditions, such as climate, type of soil *etc.* have been developed for some regions and countries. For the calculations in the national inventory of Sweden, specific national values are used for application of mineral fertilisers and manure (SEPA, 2013) and for potential eutrophication and acidification, site-specific characterisation factors have been developed (Potting and Hauschild, 2006; Finnveden *et al.*, 2009). However, specific national factors are not frequently used, as studies often involve a range of geographical sites within the system and national data can be difficult to find (Potting and Hauschild, 2006; Bare, 2009). IPCC also gives default values for nitrogen leaching, although these are very rough estimates with the same value for all soil types and fertilisers except drylands, which is set to zero (IPCC, 2006). More accurate calculations of losses from a certain amount of nitrogen and phosphorus added to an arable soil requires a model considering local conditions such as soil type, precipitation, soil organic matter *etc.* (Brentrup *et al.*, 2000; EC, 2010). A number of data on emissions from management of organic fertilisers can also be found in various field studies (*e.g.* Rodhe *et al.*, 2004; Karlsson and Rodhe, 2002; Amon *et al.*, 2006).

5 Plant nutrient sources studied - methodology and results

5.1 Methodology used

Attributional LCA methodology was used in all studies in this thesis (**Papers I-IV**). As **Papers I-IV** assessed fertiliser use based on available nitrogen added to arable soil, which did not have a direct impact on marginal effects such as change in demand for other crops, land constraints *etc.*, the choice of LCA approach was justified. Scenarios with a more consequential perspective for certain processes were assessed in the sensitivity analysis in **Papers I-III**. All studies included waste treatment in addition to the fertiliser production, *i.e.* system expansion with system enlargement including additional functions was used in all papers, **Papers I-IV**. In **Papers I** and **III** system expansion with subtracted functions was also used. Average data were used in all studies.

5.2 Appended papers

Paper I	Environmental impact of meat meal fertilizer vs. chemical fertilizer.
Paper II	Environmental impact of recycling nutrients in human excreta to agriculture compared with enhanced wastewater treatment.
Paper III	Environmental impact of recycling digested food waste as fertilizer in agriculture - a generalized case study.
Paper IV	Bringing nutrients from sea to land - mussels as fertiliser from a life cycle perspective.

5.3 Meat meal

5.3.1 Outline of the study

In the scenario studied in **Paper I**, meat meal was produced and pelleted into a fertiliser product. The burden from the generation of slaughterhouse waste, *i.e.* ABP, was not included in the study, as this was considered to be produced in the same amount and way regardless of future treatment. In the fertiliser production process, animal fat was also produced and was combusted, replacing combustion of fossil fuel oil (*Figure 6*). In the reference scenario, the slaughterhouse waste was incinerated, after addition of formic acid to prevent degradation of the material during transport. The incineration of slaughterhouse waste replaced incineration of biofuels. The fertiliser produced and used in this reference scenario was chemical fertiliser. The main functional unit of the study was the production of 1 kg of spring wheat, with the additional function of treatment of 0.59 kg of ABP. The generation of ABP, pelleting of meat meal fertiliser and production and incineration of Biomal were assumed to take place in southern Sweden, while the meat meal production and incineration of animal fat were assumed to take place in Denmark.

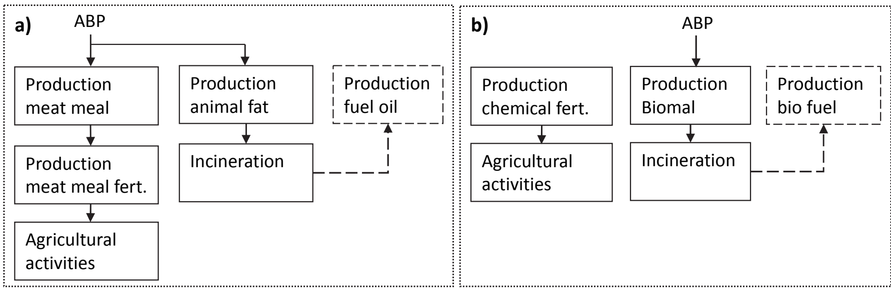

Figure 6. System description of scenarios using animal by-products (ABP) studied in **Paper I**: a) production of meat meal fertiliser (MM) and b) reference scenario (MMR) with incineration of the ABP and use of chemical fertiliser (fert.=fertiliser).

5.3.2 Main findings

The results clearly showed the importance of the infrastructure used, *i.e.* the fuels replaced in the different scenarios. As the whole fraction of slaughterhouse waste was incinerated in the reference scenario (MMR), thus containing a larger amount of energy, the energy saving was larger in this scenario. On the other hand, the meat meal fertiliser scenario (MM) replaced a fossil fuel. This meant greater savings in carbon dioxide emissions, which resulted in lower greenhouse gas emissions for the MM scenario than the reference scenario. The effects of these replaced fuels had a great influence on the final results. However, the production of meat meal fertiliser was the largest contributor to energy use and greenhouse gas emissions in the MM scenario. The results on potential acidification and eutrophication were dominated by the impacts from field operations (including leakage from soil) with total results that were similar for both scenarios. The use of non-renewable phosphorus was larger in the reference scenario, while the flow of cadmium to soil was approximately the same for both scenarios. A scenario where the incineration of slaughterhouse waste replaced incineration of coal instead of a biofuel in the MMR scenario reduced the net GWP to lower than that in the MM scenario.

5.4 Human excreta

5.4.1 Outline of the study

Two scenarios using toilet waste fractions as fertiliser were studied in **Paper II**. In one of the scenarios (TB), the urine and the faeces (blackwater) were both source-separated, and the nutrients were recycled back to arable land (*Figure 7*). In the other scenario (TU), only the urine fraction was source-separated. In both scenarios the source-separated fractions were stored according to guidelines on safe use of urine and faeces (Schönning and Stenström, 2004; WHO, 2006). In a reference scenario (TR), chemical fertilisers were produced and used. All scenarios, except the TB scenario, included treatment of nitrogen and phosphorus at a wastewater treatment plant (WWTP) for the toilet waste fractions not source-separated, so that the same amounts of nitrogen and phosphorus were removed from wastewater in all scenarios. Components included for the WWTP treatment were carbon source, *e.g.* methanol, precipitation chemicals and energy used for advanced removal of nitrogen and phosphorus to reach reduction levels specified by BSAP (SEPA, 2009). Treatment of greywater was not included. The main functional unit in **Paper II** was the production and spreading of a fertiliser containing 1 kg of plant-available nitrogen after spreading. Additional functions of the system were application of 0.15 kg of phosphorus to arable soil and treatment and removal of 1.21 kg of nitrogen and 0.15 kg of phosphorus from human excreta.

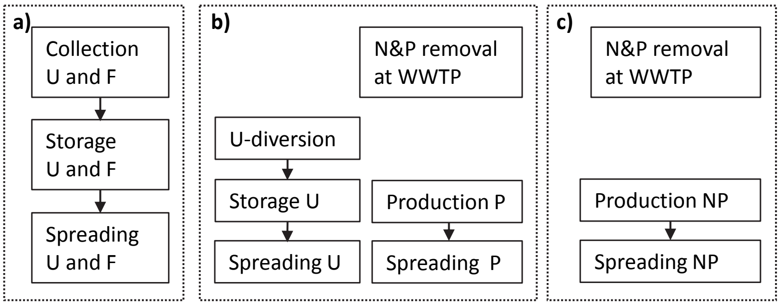

Figure 7. System description of scenarios using toilet waste fractions studied in **Paper II**: a) blackwater toilet fraction scenario (TB), b) urine toilet fraction scenario (TU) and c) toilet fraction reference scenario (TR) (U=Urine, F=Faeces, P=Phosphate rock fertiliser, NP=Chemical fertiliser).

5.4.2 Main findings

For all impact categories except energy use, the use of blackwater as fertiliser caused a larger impact than the use of urine. This was mainly due to the larger volumes of substrate that had to be handled in TB, and also because the blackwater needed a longer storage time to meet the criteria on safe use. Compared with the reference scenario, the toilet waste fraction scenarios (TB and TU) used less energy and caused lower emissions of greenhouse gases. This was mainly due to the great energy and chemical use required for advanced removal of nitrogen and phosphorus at the WWTP. On the other hand, the results on potential eutrophication and acidification were larger for the toilet waste fraction scenarios than the reference scenario. This was explained by the large emissions of ammonia during storage and after spreading of blackwater and urine. TU added significantly lower amounts of cadmium to arable soil than the other scenarios and TB used the smallest amount of non-renewable phosphate rock fertiliser. When more recently developed technology for nitrogen removal, the Annamox process, was assumed to be used at the WWTP, primary energy use was lower for TU than TB and was also strongly reduced in the reference scenario, although not to a lower level than in the TU and TB scenarios.

5.5 Digested food waste

5.5.1 Outline of the study

Paper III assessed the use of digested food waste as fertiliser. In one scenario (DF), source-separated food waste from households and non-households, *e.g.* restaurants and catering institutions, was collected in paper bags and sent to a biogas plant for biogas production. Two digestate fractions were produced from the biogas process, one liquid and one solid (*Figure 8*). The liquid fraction was stored temporarily in a large tank at the biogas plant before transport to lagoons beside the field, from where it was used as fertiliser by the farmers in spring. The solid fraction was temporarily stored in a container at the biogas plant before it was sent to be stored in a concrete container beside the field. The solid fraction was spread as a fertiliser by the farmers in autumn. The biogas was used as vehicle fuel, which was assumed to replace use of natural gas as vehicle fuel. In a reference scenario (DR), the food waste was collected mixed with other combustible waste from the households and non-households and sent to an incineration plant. The heat produced at the incineration plant was assumed to replace Swedish average district heating. In the reference scenario, chemical fertiliser was produced and used. The main functional unit in **Paper III** was the production and spreading of a fertiliser containing 1 kg of plant-available nitrogen after spreading. Additional functions were application of 0.24 kg of phosphorus to arable soil and 291 kg of food waste treated.

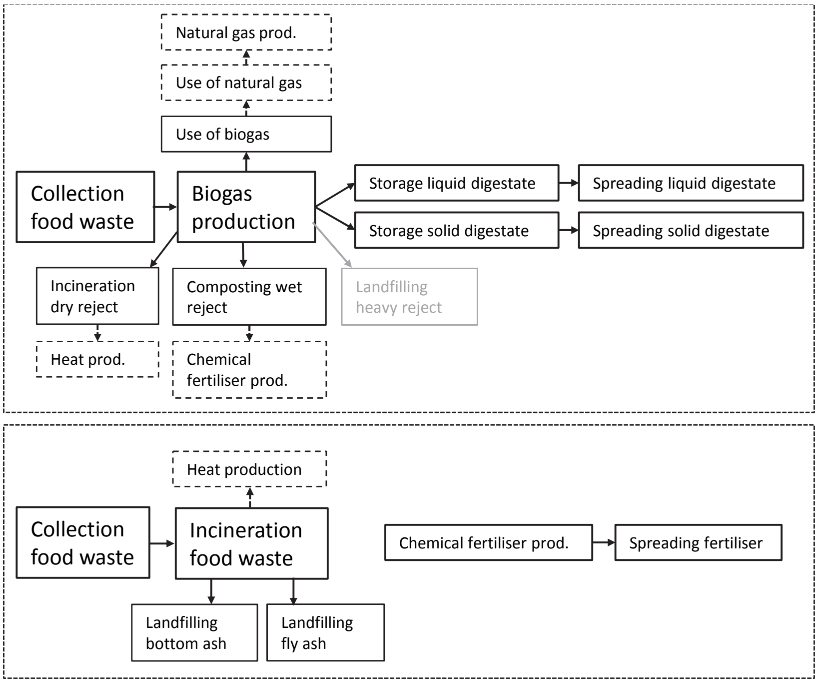

Figure 8. System description of scenarios using digested food waste studied in **Paper III**: a) digestate fertiliser scenario (DF) and b) reference scenario (DR) (prod. = production). Box in light grey includes only transport and no treatment.

5.5.2 Main findings

Both the DF and DR scenario gave negative results for primary energy use, *i.e.* a net avoidance of primary energy, due to the avoided energy sources. As the primary energy use was larger for collection of the food waste and biogas production than for the incineration process, the net avoided primary energy was larger for the reference scenario. For GWP results, methane emissions from biogas production, storage and spreading of the digestates and collection of food waste contributed significantly in the DF scenario. Although a larger amount of greenhouse gases was avoided in the DF scenario, where natural gas was avoided, than the reference scenario, the total GWP result was significantly larger for the DF scenario. For acidification and eutrophication too, the DF scenario resulted in higher total emissions than the reference scenario. This was mainly due to the emissions from storage and spreading of the digestates in the DF scenario and also the collection of food waste, as the reference scenario involved fewer waste bins and a smaller amount of food waste transported. On assuming that BAT (Best Available Technology) for methane losses in biogas and upgrading plants was applied, paper bags in the collection system were replaced with second-hand carrier bags and digestate management was improved, the DF scenario obtained similar results to the reference scenario for primary energy and GWP.

5.6 Mussels

5.6.1 Outline of the study

The Baltic Sea suffers from eutrophication problems and Sweden is required to reduce its nutrient load to the Baltic Sea according to the Baltic Sea Action Plan (BSAP) (HELCOM, 2011). Cultivation of mussels could be one way to meet these reductions. Due to the low salinity of the water on the east coast of Sweden, mussels cultivated grow too small to be used as food. However, the nutrients taken up by the mussels are removed from the sea when the mussels are harvested and, when brought back to land, as a second function, they can serve as *e.g.* fertiliser in agriculture. In **Paper IV**, two mussel scenarios were studied. In one scenario (MC), the mussels where composted to reduce odour and allow usage when needed by the farmer (*Figure 9*). In the other scenario (MA), the mussels were stored under anaerobic conditions in water to reduce degradation, and thus emissions of ammonia. This was a theoretical scenario as such storage is not currently implemented. In two reference scenarios, MCR and MAR, chemical fertilisers were produced and used. The main functional unit used in **Paper IV** was to supply arable land with 1 kg of plant-available nitrogen after spreading. Additional functions were application of 0.88 kg of phosphorus and 225 kg of liming effect (calcium oxide). The liming effect was added to the functional unit as the mussels contributed a significant soil liming effect and this is a valuable function for agriculture. In these comparisons also an additional function of removal of nitrogen and phosphorus at a WWTP was included. The removal included was relative to the nutrient reduction in the Baltic Sea in the corresponding mussel scenario. The use of mussels as fertiliser was also compared with the use of meat meal in **Paper IV**, but these results are not presented in this thesis.

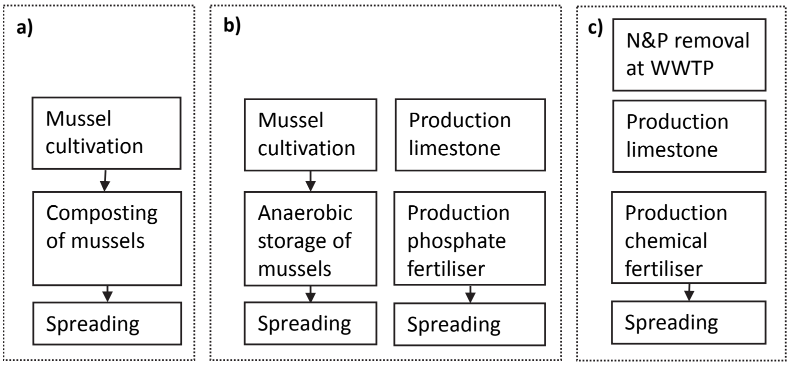

Figure 9. System description of scenarios using mussels studied in **Paper IV**: a) mussel composting scenario (MC), b) mussel anaerobic storage scenario (MA) and c) mussel reference scenarios (MCR and MAR). Two reference scenarios were needed due to the different amounts of nitrogen (N) and phosphorus (P) removed in the MA and MC scenarios.

5.6.2 Main findings

The emissions from composting of the mussels contributed significantly to the total results in all impact categories except energy use, with significantly larger potentially acidifying and greenhouse gas emissions for composting of mussels than storing them anaerobically. Since more mussels were needed in the MC scenario than the MA scenario to fulfil the functional unit, more nutrients were removed from the sea in the MC scenario. Due to this larger removal of nutrients, the total result for potential eutrophication was significantly smaller for the MC scenario than the MA scenario, where both scenarios gave negative results, *i.e.* results below zero. Compared with the reference scenarios, including nitrogen and phosphorus removal at a WWTP, the MA and MC scenarios had larger or similar results for eutrophication, acidification and GWP, while the primary energy use was lower. As the liming product and the chemical fertiliser used in the MA and reference scenarios contained significant amounts of cadmium, the compost scenario added the smallest amount of cadmium to soil. The MC scenario also used the smallest amount of non-renewable phosphate fertiliser.

5.7 Combined presentation of results

The results of **Papers I-IV** are presented in combination in this section for each impact category, and also for potential carbon sequestration. The base unit in all studies except in the meat meal study (**Paper I**) was 1 kg of plant-available nitrogen, *i.e.* 1 kg of nitrogen that can replace 1 kg chemical fertiliser nitrogen, after spreading. Since the functional unit and the system boundaries differed between the studies, the results cannot be directly compared. The meat meal study had a functional unit of 1 kg of wheat produced, so these results were here recalculated to 1 kg of plant-available nitrogen, after spreading. In addition, losses of nitrogen and phosphorus from soil were omitted as none of the other studies included these. The reference scenarios presented in this section all included chemical fertiliser.

Table 2. *Abbreviations for the scenarios used in the thesis*

Abbreviation	Scenario
MM	Meat Meal
MMR	Meat Meal Reference
TB	Toilet Blackwater
TU	Toilet Urine
TR	Toilet Reference
DF	Digestate Fertiliser
DR	Digestate Reference
MC	Mussels Composted
MCR	Mussels Composted Reference
MA	Mussels Anaerobic
MAR	Mussels Anaerobic Reference

5.7.1 Primary energy use

Overall, the two fertilisers based on toilet waste fractions (TB and TU; **Paper II**) and the two mussel fertilisers (MC and MA; **Paper IV**) used less primary energy than their reference scenarios. The toilet waste fractions reduced the primary energy use to the largest part (*Figure 10*).

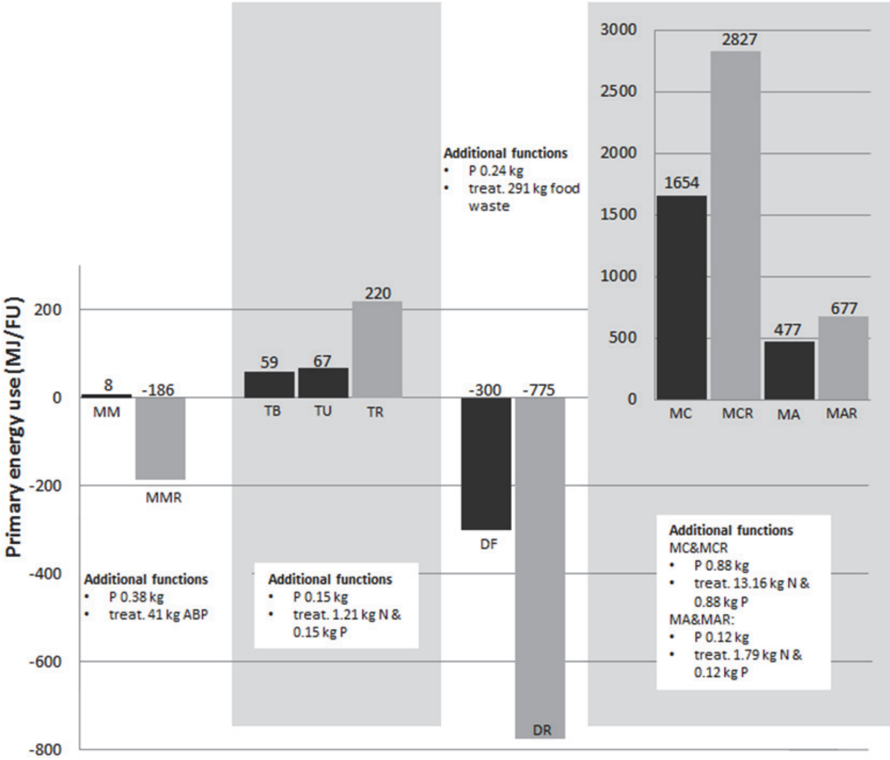

Figure 10. Primary energy use in all scenarios studied in **Papers I-IV**. Given in text is the additional functions included (treat.= treatment of).

The major influences on primary energy results for the meat meal and digestate scenarios (MM, MMR, DF and DR; **Paper I** and **Paper III**) were the avoided energy systems. In the MM scenario the avoided fuel oil and the relatively energy-consuming production of meat meal contributed most and almost balanced each other out. As the whole slaughterhouse waste fraction was used for energy recovery in the MMR scenario, the energy saving was large for this scenario. For the food waste scenarios (DF and DR; **Paper III**), about the same amount of energy were recovered, but as collection of the source-separated food waste and the biogas production were relatively energy demanding, the DR scenario avoided a larger amount primary energy use than the DF scenario. In the TU, TR, MAR and MCR scenarios (**Paper II** and **IV**), the main contributors to primary energy use were the removal of N and P at the WWTP. In the toilet waste fraction reference scenario (TR), treatment at the WWTP contributed almost 80% of the primary energy use. In the blackwater scenario (TB), the main contributors were the collection system, flushing (including water and electricity use) and transport of the blackwater fraction to the field.

The main contribution to primary energy use in the mussel composting scenario (MC; **Paper IV**) was the production of materials for mussel cultivation, as a large amount of mussels was needed for the production of 1 kg of plant-available nitrogen. The anaerobic storage of mussels (MA scenario), also used relatively large amounts of primary energy and, in addition, the production and transport of limestone contributed significantly to the primary energy use. In spite of the large use of primary energy in the MA and MC scenarios, the reference (MCR and MAR) scenarios had larger results for primary energy use.

5.7.2 Greenhouse gas emissions

Of all fertilisers investigated, meat meal fertiliser (MM; **Paper I**), toilet waste fractions (TB and TU; **Paper II**) and to some extent anaerobically stored mussels (MA; **Paper IV**) all resulted in lower GWP than their reference scenarios (*Figure 11*).

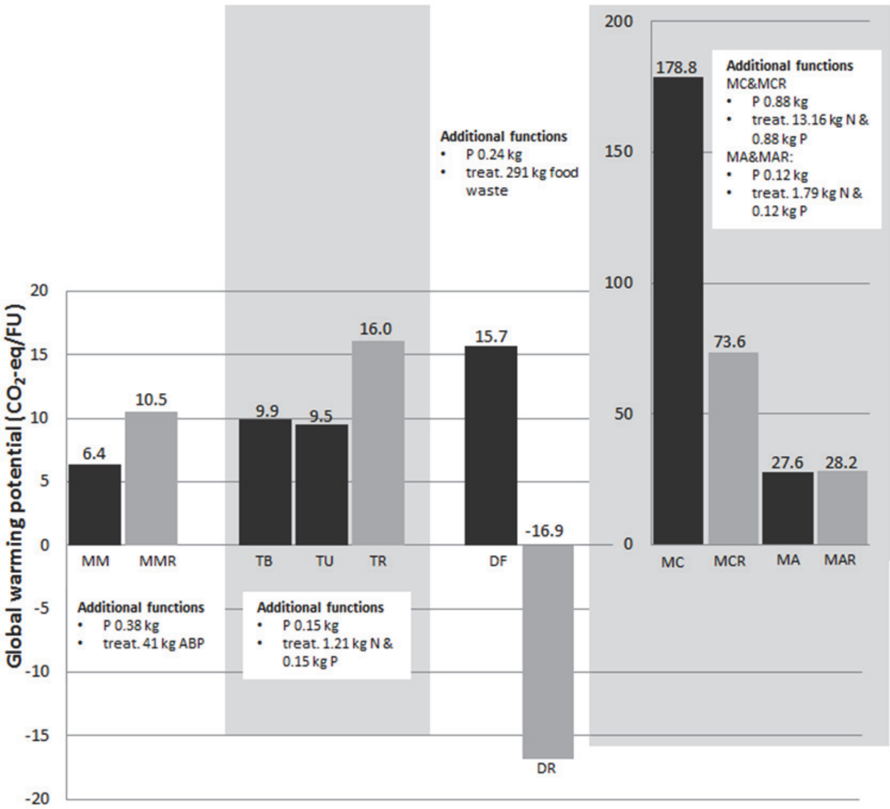

Figure 11. GWP in all scenarios studied in **Papers I-IV**. Given in text are the additional functions included (treat.=treatment of).

Avoided energy systems in the meat meal and digestate scenarios (**Paper I** and **III**) and the removal of nitrogen and phosphorus at the WWTP in the TU, TR, MAR and MCR scenarios (**Paper II** and **IV**), contributed significantly also to the results of GWP. In the MM scenario (**Paper I**), production of meat meal almost balanced out the avoided GWP from production and use of the avoided fuel oil. For both meat meal (MM and MMR) scenarios, nitrous oxide emissions from soil contributed significantly to the net result. As the incineration of ABP in the reference (MMR) scenario replaced a biofuel, GWP was not avoided from the added energy system. Instead, production of chemical fertiliser and nitrous oxide emissions from soil were the main contributors. In the DF scenario (**Paper III**), GWP from biogas production and digestate handling together was almost as large as the avoided GWP from the replaced natural gas. Collection and transport also contributed significantly. The GWP avoided from replaced heat production in the reference (DR) scenario, was significantly larger than the other contributions. For all toilet waste fraction scenarios except TR (**Paper II**), the main contributor to GWP was the nitrous oxide emissions after spreading. In the TR scenario, the removal of nitrogen and phosphorus at a WWTP was a larger contributor, resulting in larger GWP than in the other toilet waste scenarios. In the composted mussel scenario (MC; **Paper IV**), production of materials for mussel cultivation and emissions from composting were the main contributors. In the MA scenario, limestone production was the main contributor. For the two reference scenarios, MCR and MAR, chemical fertiliser production and removal of nitrogen and phosphorus at the WWTP were the main contributors.

5.7.3 Potential eutrophication

All fertilisers investigated contributed to larger net results on potential eutrophication than their reference scenarios, although, the results were similar in the meat meal scenario (MM; **Paper I**) (*Figure 12*). This was due to the ammonia emissions from storage and after spreading of the fertilisers except for the meat meal fertiliser as meat meal is pseudo-stable, *i.e.* stable due to low moisture content. There were no significant difference in eutrophying emissions from combustion of the fuels in the meat meal scenarios (MM and MMR; **Paper I**). Thus, the meat meal scenarios contributed insignificantly to potential eutrophication. A larger volume stored in the TB scenario than the TU scenario caused larger eutrophying emissions for the TB scenario. In the DR scenario, incineration was the main contributor to potential eutrophication. In the mussel reference scenarios (MCR and MAR; **Paper IV**), the same amounts of nitrogen and phosphorus as removed from the sea in the MC and MA scenarios, respectively, were removed at the WWTP. Due to the potentially eutrophying emissions at composting and anaerobic storage, *e.g.* ammonia emissions, the MC and MA scenarios avoided less net potential eutrophication than the MCR and MAR scenarios.

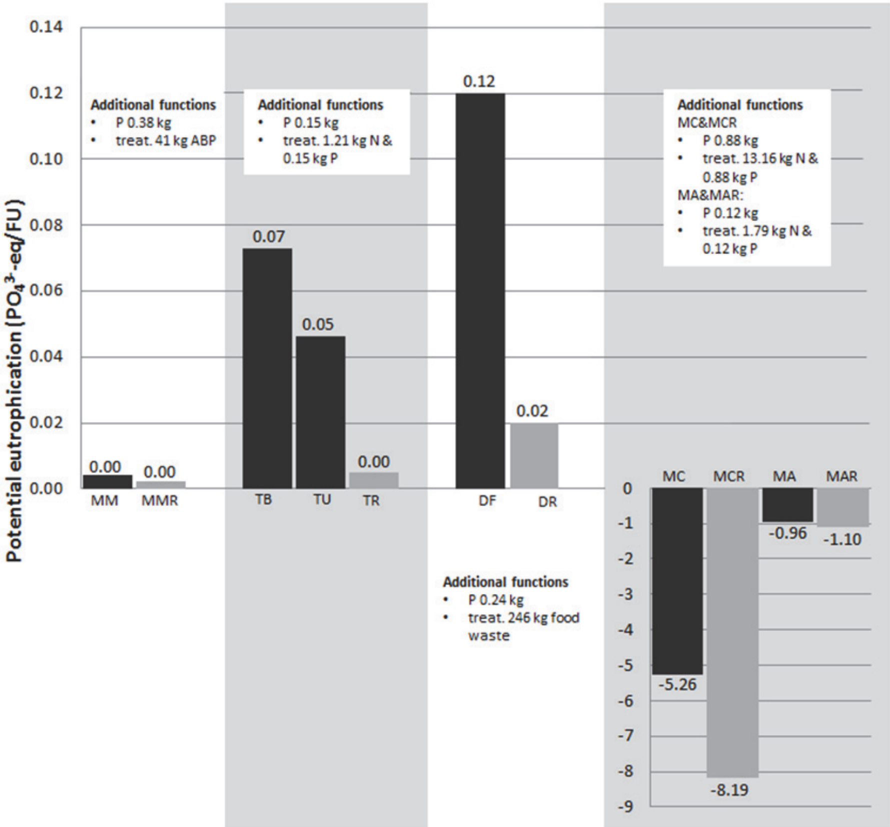

Figure 12. Potential eutrophication in all scenarios studied in **Papers I-IV**. Given in text is the additional functions included (treat.=treatment of).

5.7.4 Potential acidification

All fertilisers investigated contributed to larger potential acidification than their reference scenarios, except the meat meal fertiliser (MM; **Paper I**), which followed the same trend as for the results on eutrophication due to that ammonia emissions from storage and after spreading also contribute to potential acidification (*Figure 13*). The largest contributions in the MM scenario derived from vehicle operations, e.g. transport of the meat meal fertiliser and spreading, and the energy used at the meat meal production plant. In the reference (MMR) scenario, the avoided emissions from the biofuels replaced and the emissions from chemical fertiliser production contributed the most to potential acidification. For the mussel reference (MC and MA; **Paper IV**) scenarios, the nutrient removal at the WWTP was the major contributor in the MCR scenario and the chemical fertiliser production the major contributor in the MAR scenario.

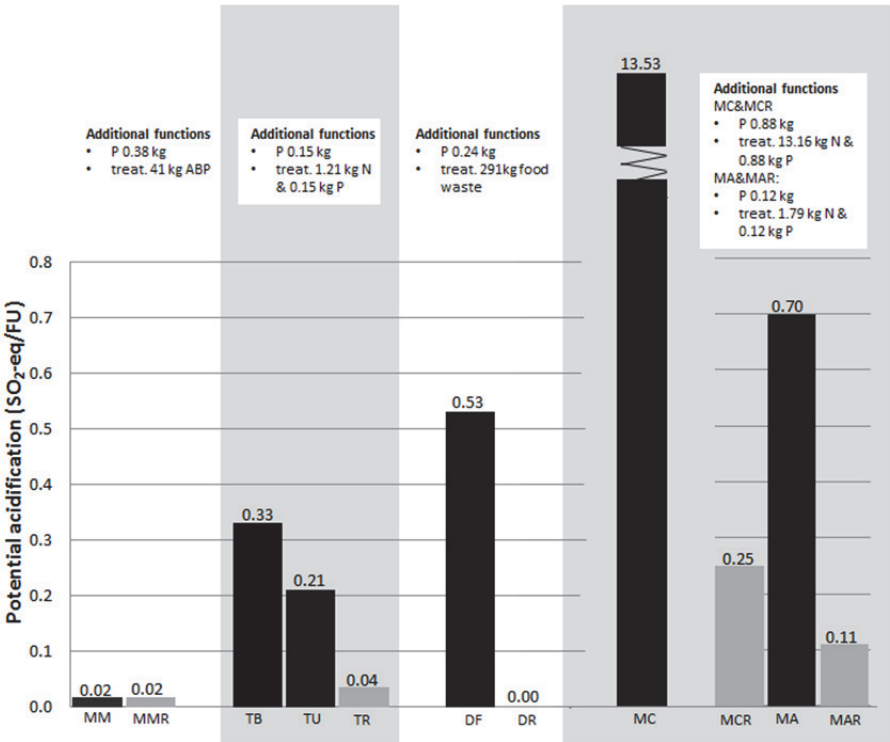

Figure 13. Potential acidification in all scenarios studied in **Papers I-IV**. Given in text is the additional functions included (treat.=treatment of).

5.7.5 Flows of non-renewable phosphate fertiliser, cadmium to arable soil and potential carbon sequestration

Composted mussels contributed the largest amount of phosphorus (P) added to soil per functional unit, mainly due to the large losses of nitrogen (N) in the composting process resulting in a compost with a N:P ratio of about 1:0.9 (**Paper IV**). Thus, the use of non-renewable phosphate rock was largest for the MCR scenario to meet the amount of phosphorus added to soil in the MC scenario (*Table 3*). Meat meal also contained relatively large amounts of phosphorus per kg available nitrogen and thus the reference MMR scenario (**Paper I**), used relatively large amounts of non-renewable phosphate fertiliser per functional unit.

Of all fertilisers studied, mussels added the largest amount of cadmium per kg plant-available nitrogen spread on arable land (*Table 3*). The mussels contained 89 mg cadmium per kg phosphorus. However, the lime added in the MCR, MA and MAR scenarios also contained significant amounts of cadmium, 0.6 mg per kg liming effect, compared with 0.4 mg per kg liming effect for the mussels. In total, including added phosphate rock, the MCR added more cadmium to arable soil per functional unit than the MC scenario and the MA and MAR scenarios added the same amounts (**Paper IV**). Digested food waste contained 39 mg cadmium per kg phosphorus (**Paper III**), meat meal 3 mg per kg phosphorus (**Paper I**), blackwater 11 mg and urine 0.6 mg (**Paper II**). A cadmium content of 3 mg per kg phosphorus was assumed for phosphate rock in all scenarios, as this is the content of phosphate rock originating from the Kola Peninsula, which is the main source of chemical fertilisers used in Sweden. This is considered a very clean phosphate rock. The average cadmium content of phosphorus fertilisers used in Sweden during the agricultural season 2011/2012 was 4.9 mg per kg phosphorus (SCB, 2013b) while the European median value is around 87 mg cadmium per kg phosphorus (Nziguheba and Smolders, 2008).

Based on a literature review (Bernstad and la Cour Jansen, 2012) and data used in the EASEWASTE model (Hansen *et al.*, 2006), sequestered carbon from addition of organic material was assumed to be 7% of carbon added from initially degraded products such as compost and digestate over 100 years. For the meat meal, the initial rapid degradation was set to 50% and thus, in total, 3.5% of additional carbon added to the arable soil with meat meal was assumed to be potentially sequestered after 100 years. The scenarios adding most organic material to soil, *i.e.* the MC and the DF scenarios, had the largest potential for carbon sequestration (*Table 3*). The potential carbon sequestration was added as avoided carbon dioxide emissions to the results in **Paper III**, but not in the other studies.

Table 3. *Use of non-renewable phosphate rock (kg P), cadmium flow to arable soil (mg) and potential carbon sequestration (kg) in the different scenarios studied in* ***Papers I-IV****, all presented per kg plant-available nitrogen to arable soil after spreading*

Scenario	MM	MMR	TB	TU	TR	DF	DR	MC	MCR	MA	MAR
Phosphate P	-	0.38	-	0.05	0.15	-	0.24	-	0.88	0.76	0.76
Cadmium	1.1	1.1	1.7	0.2	0.5	9.5	0.7	78	137.8	130.7	130.7
Pot. carbon seq.	0.13	-	0.19	-	-	1.50	0.73	1.38	-	0.27	-

5.7.6 Environmental impact in short

The organic fertilisers studied each had their own environmental profile. Regarding GWP, the meat meal fertiliser (**Paper I**) and the toilet waste fraction fertilisers (**Paper II**) reduced the emissions compared with the reference scenario (*Table 4*). Regarding primary energy use, all fertilisers investigated except meat meal (**Paper I**) and digested food waste (**Paper III**), reduced the energy use compared with the reference scenario. However, the Swedish infrastructure and energy system chosen had a great impact on the results for GWP and primary energy use. All fertilisers included in this thesis increased the potentially acidifying and eutrophying emissions compared with their reference scenarios except meat meal, which gave similar results for acidification, and anaerobically stored mussels, which gave similar results for eutrophication. Urine fertiliser (**Paper II**) and composted mussels (**Paper IV**) were the only fertilisers that added less cadmium to soil compared with the reference scenario in this Swedish context, while meat meal and anaerobically stored mussels added about the same amount. However, the amount added with mussel fertilisers and their reference scenarios (**Paper IV**) greatly exceeded the recommended levels of KemI (2011).

Table 4. *Organic fertilisers studied in **Papers I-IV** compared with the reference scenario, with use of chemical fertiliser. + = ≥20% better, - = ≥20% worse, 0 = <20% difference*

Scenario	MM	TB	TU	DF	MC	MA
Primary energy use	-	+	+	-	+	+
GWP	+	+	+	-	-	0
Potential eutrophication	-	-	-	-	-	0
Potential acidification	0	-	-	-	-	-
Cadmium	0	-	+	-	+	0

In general, of all the fertilisers and impact categories considered in this thesis, the urine fertiliser reduced the impact in the greatest number of categories. Blackwater and composted mussels each reduced the impact in two categories. Anaerobically stored mussels and meat meal also had some environmental advantage compared with the use of chemical fertilisers, but none of the fertilisers investigated in this thesis was more advantageous for all impact categories compared to their reference scenario.

6 Discussion

6.1 Methodology

This thesis shows the advantages of using a life cycle perspective when assessing complex systems such as those studied in **Papers I-IV**. A life cycle perspective is needed both for finding hot-spots in the system under study and the level of impact of other systems included, *e.g.* energy and wastewater treatment systems. One example is the mussel study (**Paper IV**), where the results on cadmium showed that even though the mussels themselves contributed greatly to the flow of cadmium to arable soil, when lime and chemical fertiliser were added in the other scenarios, they contributed significantly more. The handling of organic fertilisers was found to have a major environmental impact in all studies. Other LCA studies on food production systems also show that the main environmental impacts are related to on-field activities, *e.g.* fertiliser application (Andersson, 2000; Brentrup *et al.*, 2004; Williams *et al.*, 2010), although few of these studies include the storage of organic fertilisers, which was found to be important in this thesis.

In general, it is difficult to estimate the emissions from fertiliser management with high accuracy as the activities involved consist of many complex biological processes that depend on many factors such as characteristics of the fertiliser, soil type, climate and weather, technique used *etc.* (Brentrup *et al.*, 2000; Nemecek and Gaillard, 2010). As exact measurements of emissions under specific conditions and specific fertilisers for *e.g.* soil application are often lacking or time-consuming to measure, use of a model is recommended (Brentrup *et al.*, 2000; EC, 2010). However, there are limitations with the use of models too, as comprehensive data are often needed and sometimes lacking. The model can also be too limited or omit important aspects (Nemecek and Gaillard, 2010). Due to time limits, models for estimating emissions from storage and spreading were not used in **Papers I-IV**.

There are a number of sources of uncertainty in LCA work, such as uncertainty in the LCA model, lack of inventory data, inaccuracy of data collected, regional and temporal variability in data *etc.* (Björklund, 2002). A source of uncertainty, due to the complexity of agricultural systems is the variability and uncertainty of the data, which hence should be dealt with (Nemecek and Gaillard, 2010). In the included papers, this was handled by varying relevant uncertain data in the sensitivity analyses. A more comprehensive way to deal with the issue would be to carry out statistical analysis on the variations in the input data and use the results for performing *e.g.* Monte Carlo simulations (Björklund, 2002; Payraudeau *et al.*, 2007).

Due to the uncertainties in LCA, the results should not be considered an exact guiding value on the environmental impact, but rather an indication of benefits and drawbacks of a certain system. Hot-spots in a system, *e.g.* activities and processes that have major impacts in a system, can be identified using LCA. From knowledge of these hot-spots in the agricultural system, further research or system changes and development can be carried out to reduce the impacts from these processes and activities, and thus potentially lead to further improvement of the farming system (Bentrup *et al.*, 2004; Nemecek and Gaillard, 2010).

Regarding choice of attributional or consequential LCA, in the ongoing debate on the most appropriate LCA approach for different types of studies (see section 4.1), the most important aspect on which a majority of the LCA community agrees, is the need for transparency about the data and system boundaries used (Tillman, 2000; Brander and Wylie, 2011). The main impact on the results from **Papers I-IV** if a consequential approach had been used instead would be the impact of the energy sources chosen in the systems. In **Papers I-III**, this aspect was included in the sensitivity analysis, where a more consequential approach was applied to the choice of energy source.

The cadmium flow to arable land was included in all studies. This was due to cadmium being the heavy metal that is of the highest concern regarding fertiliser use in Swedish agriculture (Andersson, 2000; KemI, 2011). It was included in **Papers I-IV** as the amount of cadmium added to arable soil, but could also have been assessed with a characterisation method adopted for LCA. A number of models have been developed over the last 20 years and due to differences in scope, modelling principles, classification criteria *etc.* these produce significantly different results (Finnveden *et al.*, 2009; Pizzol *et al.*, 2011). The most recently developed model, USEtox, is based on preceding models by means of constructing a consensus model for LCA use (Rosenbaum *et al.*, 2008). Since only one toxic substance was included in **Papers I-IV**, and only midpoint values assessed, this kind of model was not necessary.

6.2 The issue of multi-functionality

One can define waste as a by-product that has no economic value, and thus a common assumption for waste entering a process is that no environmental burden is allocated to it. This is sometimes called the "zero burden assumption" (Ekvall *et al.*, 2007). However, as the waste hierarchy in the Waste Directive of the European Union (EC, 2006b) gives re-use and recycling higher priority than disposal, the view on waste is currently that it is a resource. When waste is partially recovered as energy or matter, it can acquire an indirect positive or negative economic value, so the question of whether the waste is a "zero burden" product or a co-product arises. The ILCD Handbook (EC, 2010) recommends that any co-product with an economic value over zero should be solved as a multifunctional system with either allocation or system expansion applied between the waste and the first system, depending on the goal and scope of the study. In comparative studies, a way to include a waste product without including the system by which it is produced is if an identical amount of waste is treated in all the different scenarios of the LCA. However, this is only valid when the waste-producing system is identical and produces the same amount of waste for all scenarios (Finnveden, 1999). This was how the issue was handled in **Papers I-IV**, where *e.g.* the waste product in the meat meal studies (**Paper I**) was a certain amount of slaughterhouse waste coming from the same slaughterhouse.

A question arising from the multi-functionality of the included studies is if they also could be considered as assessments of different waste treatments. However, if the focus of **Papers I-IV** had been to assess waste treatment, then other waste treatment techniques would perhaps have been chosen, which might not have included nutrient recovery. Thus it is of the utmost importance to state the 'main function' for these kinds of studies and therefore waste treatment was here considered as an 'additional function'.

It can be argued that a system using system expansion rather than allocation with partitioning is more complex and thus contain more uncertainties, as more assumptions on substituted and added processes are included. It can also be argued that also partitioning is based on assumptions and further, that the system becomes more relevant, better representing the reality when broadening the boundaries of a system, provided that the included functions are valued also in reality. Although, it is important that all processes included are given within the system boundaries. For example, the infrastructure, *e.g.* included systems fulfilling additional functions other than nutrient content of the fertiliser, proved to have a major influence in all studies, stressing the impact of the multi-functionality of the systems studied. An example on the impact of a chosen alternative system fulfilling an additional function is a study by Tidåker *et al.* (2007) where the use of urine as fertiliser, compared to chemical fertiliser use, reduced potential eutrophication, in contrast to increasing it as in the study of **Paper II**. This was due to that Tidåker *et al.* (2007) included the removed nitrogen and phosphorus from the wastewater flow, *i.e.* due to use of source-separated urine as fertiliser, as reduced eutrophying effluent from the WWTP, instead of stricter removal at the WWTP, as in **Paper II**.

If *e.g.* economic allocation had been used in **Papers I-IV**, in general it could have lowered the impact of fertiliser production, as the waste most often would have a lower economic value than the energy produced in the system (see *e.g.* **Papers I** and **III**). However, this could change if the cost of alternative plant nutrient sources became higher, *e.g.* if the price of chemical fertilisers increased.

6.3 Emissions from storage and spreading

The environmental impact from the use of various organic substrates as fertiliser is greatly influenced by the emissions from storage and after spreading. The effect on the environmental impact occurs both as the emissions themselves have an impact on global warming potential (methane and nitrous gas emissions), eutrophication and acidification (ammonia and nitrogen oxides emissions) and as losses of nitrogen also result in a larger amount of substrate to handle per kg of plant-available nitrogen after spreading, thus indirectly affecting all impact categories included here. The processes causing emissions at storage are complex and depend on a number of different factors, such as characteristics of the substrate, available oxygen, temperature, crust formation *etc*. The emissions are therefore difficult to estimate and predict. For example, several studies report reduced methane emissions from storage of digested slurry compared with undigested slurry (Amon *et al.*, 2006; Clemens *et al.*, 2006), whereas a recent Swedish study showed larger emissions from digested slurry than from undigested slurry (Rodhe *et al.*, 2013). In general, greenhouse gas emissions from storage of slurry are dominated by methane emissions (Clemens *et al.*, 2006; Petersen *et al.*, 2013) and methane formation is highly dependent on temperature (Sommer *et al.*, 2000; Sommer *et al.*, 2007). Thus, the study representing Swedish climate by Rodhe *et al.* (2013) was used on methane emissions from storage of digested food waste.

Emissions from composting of mussels (**Paper IV**) had the greatest impact on the results of emissions included in this thesis, as composting is an aerated process with high microbial activity where about 50% of the nitrogen is lost, often largely as ammonia (Sonesson, 1996). Methane and nitrous oxide are also produced in the composting process. Emissions of methane and nitrous oxide were also included for the storage of digestate (**Paper III**) and contributed significantly to the results on GWP, especially for the storage of solid digestate. For the studies on toilet waste fractions (**Paper II**) and the anaerobically stored mussels (**Paper IV**), nitrous oxide and methane emissions from storage were not included, but in retrospective they should have been. However, the contribution would probably have been low, as the emissions of methane for untreated slurry in Swedish climate have been shown to be significantly lower than for digested slurry and nitrous oxide emissions negligible (Rodhe *et al.*, 2013).

Emissions after spreading are also highly dependent on conditions such as weather, spreading technique, soil properties *etc.* Ammonia emissions after spreading are well documented, although also complex, with dominant factors influencing the emissions being time between spreading and incorporation, viscosity, pH and ammonium nitrogen content of the substrate. The importance of incorporation could be seen for the spreading of urine, where studies show that the ammonia emissions are close to 1% of the applied N when the urine is incorporated directly into the soil (Rodhe *et al.*, 2004). The digestate and mussel scenarios (**Papers III** and **IV**) had the greatest impact on ammonia emissions after spreading, as the mussel compost and the solid digestate were considered to have the same ammonia emissions as solid manure, about 30% of ammonium nitrogen added (Karlsson and Rodhe, 2002). In the studies, the fertilisers were assumed to be incorporated after 4 hours. If they had been incorporated directly instead, the emissions would have been almost halved (Karlsson and Rodhe, 2002). Direct nitrous emissions from spreading contribute to global warming impact. These emissions were quite similar for all scenarios studied in **Papers I-IV**, as they depend on total nitrogen and the scenarios represented 1 kg of plant-available nitrogen.

Nutrient losses from soil were only included in the meat meal study (**Paper I**, but not in the results presented in section 4.5). They were estimated based on analysed flows on nitrogen and phosphorus in basins of agricultural-type areas. Use of a model considering fertiliser characteristics, soil type, precipitation *etc.* (Brentrup *et al.*, 2000; EC, 2010) was excluded, as it would have been time-consuming and the differences small between the scenarios, since the amounts of nitrogen and phosphorus added to the same soil were similar in all scenarios. However, use of an organic fertiliser containing more organic nitrogen would have had a higher impact on potential eutrophication than fertilisers with less organic nitrogen, which ought to be taken into consideration.

6.4 Primary energy use and GWP

The results on primary energy use also depended a lot on nitrogen emissions from the handling processes of the organic fertilisers, as more emissions meant that a larger fraction of substrate was needed per functional unit. The impact of this was especially large in the mussel study (**Paper IV**), where the composted mussels required a significant amount of the energy used. Another factor with a large influence on the primary energy use was the energy content of the substrate and whether energy was recovered from it. In **Papers I** and **III**, energy was recovered in all scenarios and there was a general trend that when energy was recovered from the whole organic waste fraction, more energy was saved than when parts of the organic fraction were used as fertiliser without or with less energy recovered. Removal of nitrogen and phosphorus at a WWTP (**Papers II** and **IV**) also had a large impact on the primary energy results.

Regarding GWP, the alternative technological systems included, *i.e.* the energy system and wastewater treatment system, also had a great influence on the results. Thus, for both primary energy and GWP, appropriate choice of system boundary was highly important in order to correctly and fairly compare recycling of plant nutrients from wastes and by-products with existing systems. Nitrous oxide emissions from the wastewater treatment at a WWTP contributed significantly to the GWP. Previous measurements of these emissions vary widely, depending on treatment method used at the WWTP and time of study (Westling, 2011; Arnell, 2013), and should thus be considered an important source of uncertainty. Storage of digestate and composted mussels both contributed greatly to GWP, as discussed above. In the digestate scenario, methane losses from biogas production also contributed greatly to GWP. A previous LCA study on Biofer 10-3-1 gave a net GWP of 0.07 kg CO_2-eq. per kg fertiliser produced (Cederberg *et al.*, 2011) compared with 0.04 kg CO_2-eq. in **Paper I**. This difference was mainly due to Cederberg *et al.* (2011) allocating the burden from slaughterhouse waste treatment to the meat produced and thus not including the avoided emissions from animal fat replacing fossil fuel.

6.5 Non-renewable phosphate fertiliser use

Phosphorus, just as the other macro-nutrients, is essential for plant production and cannot be replaced. Phosphate rock, from which phosphate fertilisers are derived, is a non-renewable resource with limited reserves (Cordell and White, 2011; USGS, 2013). Current phosphate use is highly inefficient, which in many cases causes water pollution and it is estimated that about 25% of the 1 billion tonnes of phosphorus mined since 1950 has ended up in water bodies or is buried in landfills (Rosmarin, 2004). With a growing global population (UN, 2013), a higher level of phosphorus recovery and reuse is required to meet future demands (Cordell and White, 2011). Agriculture contributes 90% to the global phosphorus demand (Cordell *et al.*, 2009) and use of non-renewable phosphate rock is thus an obvious process to include in LCAs of agricultural systems (Brentrup *et al*., 2001).

In the results on use of non-renewable phosphate, the amount of nitrogen lost in the handling of the organic substrate had a great impact, as a larger nitrogen reduction decreases the N:P ratio, *i.e.* with higher nitrogen losses the organic fertiliser will contain more phosphorus per kg nitrogen. This can be seen in the mussel scenario (**Paper IV**) where the composted mussels, which had the largest nitrogen losses of all scenarios included, also had the largest ratio of phosphorus to nitrogen added to soil. Thus in the reference scenario of the composted mussels, the largest amount of non-renewable phosphate fertiliser was required. Naturally, the N:P ratio of the waste and by-product before treatment also influenced the addition of phosphorus to soil. Meat meal had the lowest N:P ratio (3:1) of all investigated fertilisers before further treatment, which was the main reason to the meat meal scenarios adding the second largest amounts of phosphorus to arable soil, per kg plant-available nitrogen. However, looking at the total potential of avoiding non-renewable phosphate rock in Sweden, the toilet waste fractions has the largest potential (Table *1*).

In **Papers I-IV**, use of phosphate rock was estimated as the use of phosphate rock fertiliser consumed in the different scenarios, *i.e.* the amount of phosphorus added to arable soil with the organic fertiliser. This did not directly correspond to the total amount of non-renewable phosphate rock used, as losses in the mining process and fertiliser production were not considered. The phosphorus losses from mining of phosphate rock depend on mining technique, ranging between 10-30% (Althaus *et al.*, 2007; Van Kauwenbergh, 2010). The mining losses are the major losses. For example, when producing the chemical fertiliser compound triple super phosphate from the phosphate rock, only about 1% of the phosphorus is lost (Althaus *et al.*, 2007; Nemecek and Kägi, 2007).

6.6 Cadmium

In **Papers I-IV**, the phosphate fertiliser used was assumed to have a cadmium content of 3 mg per kg phosphorus, which is the level in low cadmium phosphate rock coming from the Kola Peninsula (Hyltén-Cavallius, 2010). If the median European value, *i.e.* 87 mg cadmium per kg phosphorus, had been used in **Papers I-IV**, the reference scenarios would have given larger cadmium flows than the organic fertiliser scenarios in all cases (and a plus sign for all investigated fertilisers in *Table 4*). Thus, in most of Europe, all organic fertilisers in this study would have decreased the flow of cadmium to arable soil compared to the chemical fertilisers presently used. The low cadmium concentration of the phosphate fertilisers used in this thesis should thus be considered, especially since the reserves of low cadmium phosphate rock, which is favourable in phosphate fertiliser production, are declining (SEPA, 2010a). There are no commercial technologies implemented today for removing the cadmium from the phosphate without increasing the price of phosphate fertiliser to uncompetitive levels (SEPA, 2010a).

Lime is used in agriculture to maintain a suitable pH of the soil. The average cadmium concentration of lime products used in Sweden is 0.6 mg per kg CaO liming effect (KemI, 2011). This had a great impact in the mussel scenarios, as the mussel shells contributed a liming effect to the soil, which required crushed limestone to be added in the other scenarios to fulfil this additional function. In total about 40 kg of cadmium are sold annually with liming products for agricultural and horticultural purposes in Sweden (KemI, 2011; SCB, 2013c) and 50 kg with phosphate fertilisers (SCB, 2013b). This corresponds to about 17 mg per hectare of utilised arable land with lime (Kemi, 2011; SCB, 2013c) and about 21 mg cadmium per hectare with phosphate fertilisers (SCB, 2013b). This can be compared with the annual deposition of cadmium in Sweden, which is estimated to be 30-400 mg per hectare, depending on location (KemI, 2011). Thus for cadmium flow to arable soil liming products also have a great influence, though the largest contribution currently comes from deposition.

However, the cadmium added with many organic fertilisers mainly originates from the arable soil and thus does not cause much further accumulation in the soil. On the other hand, it is important to consider the possibilities of decreasing the cadmium content of arable soils, with KemI (2011) recommending a maximum level of 12 mg Cd per kg phosphorus for fertilisers.

6.7 Carbon sequestration

Addition of carbon in the form of organic matter to soil increases microbial activity, increasing soil fertility and improving soil structure (Hansen *et al.*, 2001). Several studies have shown a long-term increase in soil organic matter, and thus also in soil carbon, following addition of large amounts of organic matter (Lal, 2008; Tuomisto *et al.*, 2012). Thus, it was relevant to include in the assessment on organic fertilisers. In **Papers I-IV**, carbon sequestration was included in the form of potential carbon sequestration of each fertiliser investigated, based on estimates of carbon sequestered after 100 years. To estimate this value more accurately a mathematical model would need to be used. Also other parameters, such as higher crop yields *etc.* affect the amount of organic matter left in the soil (Gosling and Shepherd, 2005), while to make the comparison fair, the effect of differences in yield, tillage and future climate *etc.* would need to be considered. Potential carbon sequestration was only included in the GWP results in the DF and DR scenarios (**Paper III**), where it had an impact of about 3% and 5%, respectively, related to all other contributing emissions, negative and positive. The largest decrease of the GWP would be in the TB scenario (**Paper II**), where it would decrease the result with 7%. In the other papers, it would have contributed 0-2%, *i.e.* not significantly, to GWP.

6.8 Suitability in arable farming

Fertiliser recommendations in Sweden for a soil with the most common phosphorus content (P-AL Class III) (SEPA, 2010b) specify a ratio between nitrogen and phosphorus (N:P ratio) of about 6-9:1 for cereals and oil-seeds and about 2-4:1 for potato and sugar beet (Jordbruksverket, 2013b). The corresponding ratio for the fertilisers investigated in this thesis was 10:1 for urine, 8:1 for anaerobically stored mussels, 7:1 for blackwater, 4:1 for digested food waste, 3:1 for meat meal and around 1:1 for composted mussels. This indicates that from a N:P ratio perspective, urine, anaerobically stored mussels and blackwater are suitable fertilisers for cereals and oil-seed crops, while meat meal and digested food waste are suitable fertilisers for crops with higher phosphorus demands, such as sugar beet and potato. Fertilisers with higher phosphorus content, *e.g.* composted mussels, can also be applied to a crop with a lower phosphorus requirement if the amount applied is considered also for the following crop season. Most excess phosphorus is stored in the soil, although there is a certain risk of leakage (Börling, 2003).

Using existing infrastructure, *e.g.* machinery and other equipment, for the spreading of the fertiliser is also an important component when using less common fertilisers. Suitable, but perhaps not optimal, equipment existed for all fertilisers investigated in this thesis. Proper hygiene treatment ensuring safe use of organic fertilisers is another important aspect. This was fulfilled in all studies in the thesis. The economics of use of the organic fertilisers was not included, however. For example, for digestate there are ongoing discussions on how to increase the economic value and monitor the nutrient balance of the fertiliser (Avfall Sverige, 2010; Aarsrud *et al.*, 2010).

A drawback with using most of the fertilisers investigated in this thesis, and organic fertilisers in general, compared with chemical fertilisers is that they are bulky, *i.e.* have a low nutrient concentration. This requires more transport, more turns during spreading and more soil compaction, as a larger volume needs to be handled.

All the fertilisers investigated in this thesis are permitted in organic and conventional farming except urine and faeces, which are currently banned in organic farming (EC, 2008). However, all fertilisers fit well with the basic principles of organic production, which *e.g.* should be based on recycling of nutrients to arable land and minimise the use of non-renewable resources (EC, 2007; IFOAM, 2013b). Although, it is of greater importance for organic farming to find new plant nutrient sources, the fertilisers studied here could also be of interest in conventional farming. Reduced climate impact is a high priority globally and within the EU, and thus fertilisers resulting in less greenhouse gases than chemical fertilisers, such as meat meal, urine, blackwater and to certain extent anaerobically stored mussels (see *Table 4*), ought to be of interest for agriculture in general. Recycling is also a higher priority than energy recovery according to the European Waste Directive (EC, 2006b). In addition, using organic waste and by-products as fertiliser could help improve soil quality, which is a challenge for conventional or organic farms without access to manure (Tuomisto *et al.*, 2012).

6.9 Future research

As the fertilisers investigated, except the meat meal fertiliser, caused large emissions during storage and after spreading, future research should focus on developing techniques and systems for reducing emissions from the management of various organic fertilisers. This could include implementation of gas-proof covers with methane collection, storage at lower temperature and pH to inhibit microbial activity, improved spreading techniques *etc.* Techniques to keep the nitrogen in forms that avoid emissions of ammonia and nitrous oxide, as *e.g.* in the meat meal study, should also be further studied.

Solid digestate is a special challenge, as large amounts of greenhouse gases are emitted during storage.

In addition to reducing emissions, increasing the dry matter content of organic fertilisers is of great concern as it would reduce the volume of material needing to be handled. Thus, techniques for increasing the dry matter content of organic fertilisers also need to be developed.

The anaerobic storage of mussels in **Paper IV** was a theoretical scenario, as storage of mussels in this way is not yet implemented. As the results on the environmental performance of this kind of fertiliser were promising, the technique deserves development efforts. Furthermore, mussel cultivation has only been carried out on the east coast of Sweden for a couple of years and further research is needed to optimise the cultivation system with regard to the local conditions.

Large-scale anaerobic digestion is a much younger technology than large-scale incineration, so further research on optimising biogas production is needed. For example, techniques for reducing methane losses in biogas production and upgrading based on solid substrates needs to be developed. For source-separated food waste collection, studies should be performed on optimising the collection route and also on the type of collection system that should be implemented, *e.g.* type of bags, coloured plastic bags, type of waste bins *etc.* When assessing the collection system, the environmental aspects should of course be considered, but also social aspects, degree of source separation performed *etc.*

The data used in this thesis on the consequences of stricter removal of nitrogen and phosphorus at a WWTP were rough estimates and these should be verified by simulations. The nitrous oxide emissions from different type of wastewater treatment techniques should also be further studied.

Furthermore, standardised methods for estimation of certain LCA inventory data need to be developed and formalised. Within agricultural systems, there is a need for applicable models for estimating various emissions, such as emissions from storage and spreading and losses of nitrogen and phosphorus from soil. The models should be practically feasible for LCA practitioner to use, *e.g.* not too time demanding or complex to use but still accurate enough. These emissions can have a great impact on the results and development of models to estimate such emissions more accurately would also increase the comparability between different studies and lower the uncertainty in LCA of agricultural systems.

7 Conclusions

- It is essential to assess complex recycling systems based on waste and by-products using a stringent and comprehensive approach such as LCA methodology, in order to carry out a fair comparison between systems studied.
- Choice of alternative technological systems in the system expansion, *e.g.* the energy and wastewater treatment systems included, had a great impact on the final results in this thesis.
- The renewable fertilisers investigated here all gave rise to goal conflicts, as none of them was able to reduce the impact for all impact categories studied.
- Of the organic fertilisers investigated, urine fertiliser reduced the impact in the largest number of categories compared with the reference scenario (use of chemical fertiliser).
- Meat meal, blackwater and urine fertilisers reduced greenhouse gas emissions compared with the reference scenarios (use of chemical fertiliser).
- Blackwater, urine, composted mussels and anaerobically stored mussels as fertilisers reduced primary energy use compared with the reference scenarios (use of chemical fertiliser).
- Urine and composted mussels as fertilisers reduced cadmium flow to arable soil compared with the reference scenarios (use of chemical fertiliser). Urine fertiliser added the lowest amount of cadmium to arable soil of all fertilisers investigated. Digested food waste and mussels both exceeded recommended cadmium levels (per kg phosphorus) on safe fertiliser use.
- The mussel scenarios added similar or lower amounts of cadmium to arable soil than to the reference scenarios (use of chemical fertiliser),

provided that the liming effect was exploited. Mussel-based fertilisers should thus be used as a combined liming and fertilising treatment.

- If the median levels of cadmium for chemical fertilisers in Europe had been used, all organic fertilisers investigated would have added less cadmium to the soil than chemical fertiliser.
- All organic fertilisers investigated avoided use of non-renewable phosphate rock, with the toilet waste fractions having the largest potential for avoiding non-renewable phosphate rock.
- Regarding carbon sequestration, digested food waste had the largest potential in relation to the recycled amount of nutrients.
- Better technologies and systems for reducing global warming potential and potentially acidifying and eutrophying emissions from the storage and spreading of organic fertilisers should be developed, as these emissions had a major impact in most comparisons.
- A system and technology for anaerobic storage of mussels should be developed, as this is currently only a theoretical option which had environmental advantages.
- Many aspects in the digestion of food waste need to be improved and further developed, such as methane losses from the biogas plant, reduced greenhouse gas emissions from storage of digestate (mainly nitrous oxide emissions from storage of solid digestate), improved food waste collection *etc.* Sensitivity analysis showed that if these improvements were applied similar results could be obtained for GWP and primary energy use for digested food waste and chemical fertiliser use.

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