Integrating Farming and Wastewater Management

A System Perspective

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Abstract

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Source separating wastewater systems are often motivated by their integration with farming. It is thus important to scrutinise the critical factors associated with such integration. This was achieved in this work using a life cycle perspective and qualitative interviews. The objective of the thesis was to examine the environmental performance of systems integrating crop production and wastewater management in urban and rural areas. The focus was on comparisons between source separating and conventional systems using a life cycle approach. Critical factors for beneficial recycling and use of source separated sewage products as fertilisers were also investigated.

Source separation of urine or blackwater with subsequent use in crop production proved most beneficial in locations where the eutrophying emissions were critical. For Swedish conditions, such separation techniques were particularly interesting as complement for on-site systems.

The life cycle studies highlighted the importance of a high substitution rate when sewage products replaced mineral fertiliser in crop production. Plant nutrient availability and the fertilisation strategy employed at farm level are important factors to consider in this regard. A carefully constructed system for separation and collection and choice of sanitisation method also proved important. It was demonstrated that resource aspects associated with recycling systems are far more than a matter of plant nutrient recovery rate, since different activities required for the recycling might be associated with considerable use of energy and other resources. Methods for weighting abiotic resources are therefore needed.

There are varying motives, roles and responsibility for actors involved in existing recycling schemes. The results stress the importance of local authorities and farmers devising strategies for better long-term utilisation of the nutrients in source separated sewage products. Providing arenas for participation, exchange and learning is recommended for continued development of recycling systems.

Keywords: environmental systems analysis, grain production, LCA, recycling, source separation, wastewater systems

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"What is done with this golden manure? It is swept into the abyss. Fleets of vessels are despatched, at great expense, to collect the dung of petrels and penguins at the South Pole, and the incalculable element of opulence which we have on hand, we send to sea. All the human and animal manure which the world wastes, restored to the land instead of being cast into the water, would suffice to nourish the world."

Les Miserables by Victor Hugo, 1862

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Appendix

Papers I-IV

The present thesis is based on the following papers, which are referred to by their Roman numerals. Published papers are reproduced by permission of the journals concerned.

I. Tidåker, P., Mattsson, B. & Jönsson, H. 2007. Environmental impact of wheat production using human urine and mineral fertilisers – a scenario study. *Journal of Cleaner Production 15*, 52-62.

II. Tidåker, P., Kärrman, E., Baky, A. & Jönsson, H. 2006. Wastewater management integrated with farming – an environmental systems analysis of a Swedish country town. *Resources, Conservation and Recycling 47*, 295-315.

III. Tidåker, P., Sjöberg, C. & Jönsson, H. 2006. Local recycling of plant nutrients from small-scale wastewater systems to farmland – A Swedish scenario study. *Resources, Conservation and Recycling 49*, 388-405.

IV. Tidåker, P. & Jönsson, H. 2007. Organising recycling of source separated sewage products to farmland. (Manuscript).

Notes on the authorship of the papers:

In Paper I, the planning was made by Tidåker and Jönsson. Tidåker carried out the inventory, the assessment and wrote the paper with revisions from the co-authors.

In Paper II, Tidåker carried out the data collection and assessment of the agricultural system, gave input to the inventory of the wastewater systems, compiled the results and wrote the paper with input from the co-authors.

Paper III was planned by Tidåker and Jönsson. Tidåker carried out the inventory with contributions from Sjöberg. The paper was written by Tidåker with comments from the co-authors.

In Paper IV, the planning was primarily carried out by Tidåker. Tidåker performed the investigation and wrote the paper with comments from Jönsson.

Introduction

Traditionally, human excreta have been used as a crop fertiliser in many countries. Unprocessed latrine waste was commonly used in the Swedish countryside, whereas latrine waste arising in some cities was further processed by adding slaked lime or peat (Wetterberg & Axelsson, 1995). The powdery *poudrette*, transformed from urine and excreta collected in cesspools, was a popular fertiliser marketed to growers in *e.g.* Paris and industrialists developed human fertilisers based on source separated urine (Barles, 2007).

The introduction of the waterborne sewage system resulted in highly diluted wastewater, which in general was discharged into the nearest watercourse. Many rivers running through cities thus became gigantic sewers. However, in several European cities, *e.g.* London, Paris and Berlin, irrigation of arable land was introduced (Mårald, 1999). At the beginning of the 20th century, sewage from Parisians irrigated more than 5000 ha (Barles, 2007). Through irrigation, the water is purified at the same time as the crops are fertilised. The costs associated with the maintenance of these systems and their long-term failure in fulfilling the sanitary requirements led to a gradual decrease in this practice (Mårald, 1999). In the 1970s, chemical precipitation was introduced in large-scale treatment plants in Sweden (Isgård, 1998), which opened up new options for recycling of the phosphorus found in sewage.

Use of sewage products as fertilisers is often motivated by saving the finite reserves of high-grade phosphate. However, the Swedish EPA (2002) concluded that recycling of other plant nutrients and humic substances should also be considered in future wastewater systems in order to avoid sub-optimisation. The prospects for recycling plant nutrients other than phosphorus by sludge utilisation are restricted, since only minor fractions of the nitrogen and potassium in the influent are transferred to the sewage sludge. The Swedish food sector has also periodically refused to buy agricultural products grown on fields fertilised with sewage sludge (Berglund, 2001), making use of sludge in agriculture an uncertain option.

Source separation of urine or blackwater are two alternatives to a conventional wastewater system, proposed both for urban and rural areas (Otterpohl, 2001; Wilsenach *et al.*, 2003; Larsen *et al.*, 2007; Peter-Fröhlich *et al.*, 2007). Urine and faecal matter contain the majority of the plant nutrients in household wastewater, while most heavy metals and organic pollutants from the households are found in the greywater (Palmquist & Hanæus, 2005; Vinnerås *et al.*, 2006). Urine and blackwater are thus fertilisers with a high ratio of plant nutrients to undesirable pollutants, while at the same time the separate handling of these fractions implies that the eutrophying emissions to recipient waters from the wastewater system are reduced. Production of nitrogen fertiliser products contributes both to energy use and global warming (Jenssen & Kongshaug, 2003). Thus, by replacing mineral fertilisers with sewage products in crop production, the environmental burdens associated with the production of nitrogen mineral fertilisers could be avoided.

However, there might also be disadvantages with the use of sewage fertilisers compared with mineral fertilisers. Sewage fertilisers are often bulky and require spreaders, which increases the risk of soil compaction. Ammonia losses through volatilisation may occur during handling, which also affects the fertiliser value, and thus the extent to which mineral fertiliser products can be replaced. Implementing source separating systems also requires additional infrastructural investments in pipes and storage tanks and involves new strategies for organising the handling. Source separating systems are often justified by their integration with farming and their resource-efficiency. However, there is a need to scrutinise the critical factors associated with this integration.

There is an increasing emphasis on assessing different aspects of sustainability for wastewater systems. High investment costs and long life-time characterise the infrastructure in the wastewater sector (Malmqvist *et al.*, 2006). The choices made today will thus affect the performance of the wastewater system for a long time. It is therefore vital to use different strategic planning tools to find sustainable approaches for wastewater management. The Swedish trans-disciplinary research programme Sustainable Urban Water Management, of which this PhD project formed part, has developed and applied different tools to assist in planning and management in the pursuit of sustainable water and wastewater systems (Malmqvist *et al.*, 2006). One important question addressed by the programme was whether sustainable future wastewater systems could best be developed through improvements of the present systems or whether radical changes are required (Malmqvist, 2004).

Life cycle assessment (LCA) is one tool used for evaluating different options for wastewater management, *e.g.* the introduction of source separation systems applied in different settings. Many studies on nutrient recycling focus on the wastewater system, while paying little attention to the agricultural system and the interaction between the systems. Including an agricultural perspective when evaluating systems aimed at nutrient recovery is essential for getting information about the conditions in which implementation of such systems is a desirable approach.

Objectives

The overall objective of this project was to contribute to a more comprehensive understanding of aspects affecting the environmental performance of systems integrating farming and wastewater management in a Swedish context, with the emphasis on comparisons between source separating and conventional systems.

A specific objective was to investigate the environmental impacts and resource use in a life cycle perspective for systems integrating crop production and wastewater management in both urban (Papers I-II) and rural (Paper III) settings. A further objective was to identify motives and strategies for actors co-operating in existing nutrient recycling systems (Paper IV) and to discuss critical factors for beneficial recycling and use of source separated sewage products as fertilisers (Papers I-IV).

Structure of the project work

This thesis is based on four papers, briefly described below. The relationship between the four papers is illustrated in Figure 1.

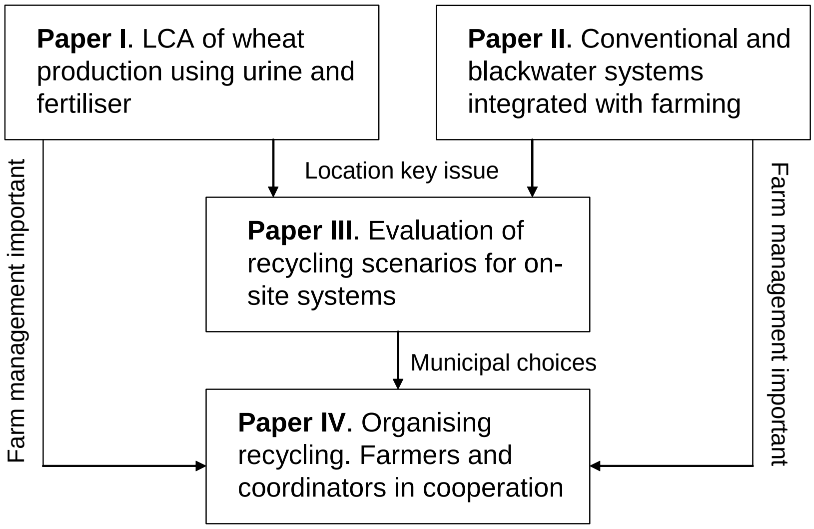

Figure 1. The four papers included in this thesis and their relationship to each other.

Papers I-III all used a life cycle perspective. Thorough descriptions of the scenarios studied in the first two papers are given in underlying reports (Tidåker, 2003; Tidåker *et al.*, 2005). In Paper I, the focus was on environmental impacts when urine replaced mineral fertilisers in wheat production. An urban setting with a urine separating system complementing an already existing conventional wastewater system was assumed. The study presented in Paper II was a sub-project within the model city Surahammar, one of five different model cities in Urban Water which were evaluated from the perspective of sustainability. In Paper II, three systems for recovering plant nutrients in sewage were assessed and compared with each other. The first system represented a system with waste disposers and where the sewage sludge was used for production of a soil conditioner. The second system was a conventional wastewater system with agricultural use of the sewage sludge. In the third system, blackwater was source separated, sanitised and used as a fertiliser in crop production. All three systems included both wastewater management and agricultural production on the same defined area generating the same crop yield.

Papers I and II highlighted the importance of a suitable fertilisation strategy for good environmental performance and concluded that the reduction of eutrophying emissions to recipient waters was one of the most important advantages with source separating systems. In Swedish municipalities, upgrading on-site systems is an urgent task since many of these do not fulfil the legal requirements on treatment. Actions are therefore proposed for enforcing the law and reducing their eutrophying emissions. However, comparisons using a life cycle approach of the environmental performance for different options were to a high extent lacking.

Paper III compared three alternatives for upgrading on-site systems, *i.e.* urine separation, blackwater separation and chemical precipitation in the septic tank. The focus was on the wastewater systems, while the agricultural use of the sewage products was considered only as avoided use of mineral fertiliser. The results showed that all three alternatives had both benefits and drawbacks when comparing energy use, recycling rate and expected reduction in nitrogen and phosphorus. Thus, the alternative that might turn out to be the most favourable depends on how different aspects are weighted to each other in the specific situation.

Since the handling at farm level has a decisive influence on the environmental performance of source separating systems, the farmer is one of the most important actors in the recycling scheme. The municipal authorities also play an important role, identified by Swedish EPA guidelines for small-scale wastewater systems as being responsible for providing conditions for beneficial use of sewage fractions. Paper IV used a qualitative approach and focused on cooperation between farmers and the coordinators assigned to establish recycling systems. Although substitution of mineral fertilisers is often considered an important feature of source separation systems, several recycling schemes failed in this respect. The results stressed the importance for the municipalities and farmers involved of devising strategies for long-term improvement in the utilisation of the nutrients in sewage products.

Methodological aspects and conclusions from Papers I-III are found in the chapter ‘A life cycle perspective on systems for recycling sewage nutrients’, while the Paper IV is discussed in the chapter ‘Organisational aspects of systems for recycling sewage nutrients’.

Background

Plant nutrients and agriculture

Inputs of plant nutrients are crucial for maintaining productivity in modern agriculture. To preserve soil fertility status, the plant nutrients removed or lost from the fields must be replaced. The increasing access to mineral fertiliser products after World War II enabled a spatial separation between feed production and animal breeding, which strongly influences the features of today’s agriculture.

The application of plant nutrients is unevenly distributed throughout the world. Some regions of the world are suffering from increased depletion of plant nutrients. In 1998, nutrient depletion in Africa was estimated at 17 kg nitrogen, 3 kg phosphorus and 20 kg potassium per ha and year (Sheldrick & Lingard, 2004). Nutrient balance calculations in Sweden show a surplus of nitrogen and phosphorus, although not of the same magnitude as reported from the other European countries (Statistics Sweden, 2005). In Europe, the mean nitrogen application rate exceeds the global mean by a factor of more than three (van Egmond *et al.*, 2002). The high imports of feedstuffs to some European countries for conversion to meat and milk result in areas with high application of manure.

Sweden also shows a high regional variability as regards livestock density. Cereal production is mainly concentrated to the plains areas, while livestock production is largely found in woodland districts in the south (Statistics Sweden, 2007a). This means that large amounts of plant nutrients are exported from the plains areas with the cereals sold for food and feed. The animal farms, on the other hand, buy nutrients with the feed, which largely end up on their fields as manure. Mineral fertilisers are thus needed as compensation for uneven distribution within agriculture, for replacing the plant nutrients found in foodstuffs ending up in the sewage system and also as compensation for nutrient losses from the fields and the manure handling.

There are many aspects associated with the use of different fertiliser products. Some of these are dealt with briefly below, primarily resource aspects relating to the production and current use in agriculture.

Nitrogen

Nitrogen can be captured from air biologically by fixation by bacteria or chemically by combining atmospheric nitrogen with hydrogen in *e.g.* natural gas under high pressure and temperature to ammonia (the Haber Bosch process). Natural gas is the principal source of hydrogen in most commercial fertiliser plants in the U.S. (Kramer, 2004), and the European fertiliser industry is the single biggest user of natural gas in Europe (EFMA, 2007a). Ammonia is later converted to other nitrogen compounds. Urea is the world's most commonly used nitrogen fertiliser product, due to its high concentration and relatively low price per unit of nitrogen (Kramer, 2004). However, application of urea is associated with a risk of evaporation if the urea is not incorporated into the soil. Within the EU, calcium ammonium nitrate and ammonium nitrate are the most commonly used nitrogen fertiliser products (EFMA, 2007b).

According to Jenssen & Kongshaug (2003), the global production of mineral fertilisers accounts for approximately 1.2% of the energy consumed in the world and is responsible for approximately 1.2% of the greenhouse gas emissions. They claim that it is theoretically possible to reduce the energy consumption by almost 40% and the greenhouse gas emissions even more by implementing new technology.

Nitrogen is abundant in organic matter in the topsoil and may be in the magnitude of 10 tonnes per ha (Karlsson *et al.*, 2003). The median value for Swedish arable soils is about 7 tonnes nitrogen per ha (Eriksson *et al.*, 1997). Predicting the mineralisation of this fraction is in general very difficult due to its high variability, and fertiliser planning therefore often fails to estimate the long-term delivery of nitrogen from organic matter.

Nitrogen fertilisers should preferably be applied only when needed by the crop, *i.e.* in spring and early summer for cereals under Swedish conditions, otherwise considerable nitrogen losses may occur (Weidow, 1999).

In 2004/05, the application of plant-available nitrogen to arable land in Sweden was on average 107 kg per ha (Statistics Sweden, 2006). Most of this (83%) came from mineral fertilisers. Animal farms use more total nitrogen per ha when nitrogen in organic matter is also included, but less directly plant-available nitrogen compared with farms specialising in cereal crop production.

A considerable amount of the nitrogen applied is lost through different processes. Denitrification, volatilisation of ammonia, leaching and soil erosion are the main routes of nitrogen removal, accounting for losses corresponding to about half the total input (Smil, 1999). For Swedish agriculture, the average losses to air and water are estimated to be in the magnitude of 50 kg of nitrogen per ha (Statistics Sweden, 2005).

Phosphorus

Fertiliser production plays a dominant role in the global demand for phosphorus and accounts for approximately 80% of the phosphate used world-wide (Steen, 1998). The reserves, defined as the fraction of the total resource that could be economically extracted or produced at current prices, are calculated to last for 124 years at the current extraction rate (USGS, 2007a). The reserve base is defined by the same source as the fraction assumed to become economically viable within planning horizons and is estimated to last 345 years at the current extraction rate. In addition, there are huge resources of phosphorus on *e.g.* the continental shelves, containing enough phosphorus for millennia (Smil, 2000). Higher prices will provide opportunities for extracting phosphate rock not currently defined as economically viable. Running out of phosphorus is thus not the main problem associated with current use. Instead, it is the associated contamination that causes concern (Smil, 2000). Waste giving rise to environmental degradation in mining areas is another problem that needs to be addressed. As phosphate ores with lower cadmium content are depleted, the average cadmium content will increase. Removal of cadmium is possible, but requires 18-32 MJ energy per kg phosphorus (Smil, 2000).

The phosphorus status of soils in different parts of the world is very variable. Many soils in sub-Saharan Africa are characterised by phosphorus deficiency and high phosphorus fixation (Ayaga *et al.*, 2006). The relatively high cost of mineral fertilisers compared with the value of the crop is one factor restricting the use of phosphorus fertilisers. In other regions, phosphorus is abundant in arable soils. Between the 1950s and 1990s, the phosphorus reserves in Swedish arable land have increased by some 700 kg per ha (Andersson *et al.*, 1998). The phosphorus level in the topsoil is generally high, with about 40% of all soils being in the highest class, P-HCl class V (Eriksson *et al.*, 1997). In 2004/05, 61% of Swedish arable land was fertilised with phosphorus (Statistics Sweden, 2006). On average, this area received 24 kg of phosphorus per ha, most as manure.

According to guidelines from the Swedish Board of Agriculture (2002), phosphorus should, if possible, be applied to each crop in the crop rotation. One factor behind this recommendation is that water-soluble phosphorus becomes less available with time (Hahlin & Johansson, 1977). For only a few crops, *e.g.* sugarbeet and potatoes, the recommendation is to apply additional phosphorus, to also cover the needs of one or two succeeding crops. Large application rates of phosphorus may result in high incidental phosphorus losses, which could be a more important source of phosphorus losses than diffuse losses through the soil (Withers *et al.*, 2003). This aspect is also an argument to adapt the application in accordance with crop requirements in the time perspective of one year (Ulén & Mattsson, 2003). The average losses of phosphorus from observation fields in a Swedish network have been estimated at 0.3 kg per ha and year (Ulén *et al.*, 2001). However, the variation has been considerable both in time and space, with high losses reported from a few critical fields.

Potassium and sulphur

Both potassium and sulphur are frequently used as fertilisers. The use of potassium on fertilised areas in Swedish agriculture was on average 94 kg potassium per ha in 2004/2005, thus almost of the same magnitude as the nitrogen use (Statistics Sweden, 2006). The major part of this potassium was applied in the form of manure. Clay soils often have a high capacity to deliver potassium through weathering (Öborn *et al.*, 2001). Omitted or reduced potassium fertilisation rates are therefore often used on clay soils. According to the U.S. Geological Survey (USGS, 2007b), the reserve life-time of potassium is estimated to be 277 years at current mine production rates.

The sales of sulphur fertilisers in Sweden are considerably higher than for phosphorus but lower than for potassium according to the Swedish Board of Agriculture (2007). The reserves of sulphur in crude oil, natural gas and sulphide ores are large, and sulphur in gypsum and anhydrite is almost limitless (USGS, 2007c). The vast majority of all sulphur produced is recovered at *e.g.* petroleum refineries and plants for processing natural gas and coking in order to meet compulsory environmental regulations. Using the concept of reserves and reserve base is thus inadequate since sulphur is primarily a by-product, thereby lowering the market price, and making the actual supply of sulphur dependent on the production of petroleum, natural gas, *etc*.

Cereal production

Cereals are grown on approximately on 1 million ha in Sweden (Statistics Sweden, 2007a). Winter wheat and spring barley are the two single most cultivated cereals, with oats in third place.

Winter wheat is preferably cultivated on well-drained clay soils with a good liming status. The average expected yield is 6.1 tonnes (Statistics Sweden, 2007b). Recommended sowing time in most places is September (Fogelfors, 2001). Phosphorus and potassium should preferably be provided in the autumn before sowing, while nitrogen should be provided in the spring at the time of stem elongation. Applying two or three smaller doses of nitrogen instead of one larger dose is desirable for optimising the application rate and thus minimising the risk of leaching.

Spring barley and oats are both grown all over Sweden. The fertiliser is normally applied in one dose in close connection with sowing in the spring. The expected yields are 4.2 and 3.9 tonnes per ha, respectively (Statistics Sweden, 2007b).

Manure handling

The majority of Swedish dairy and fattening pig farms have liquid manure systems (Statistics Sweden, 2006). Broadcast spreading is still the most commonly used technique in Sweden, although band spreading is constantly increasing. In 2004/2005, 35% of the slurry was applied using band spreading.

Broadcast spreading involves the manure making a trajectory in the air from the splash plate to the crop, while band spreading puts the slurry in parallel bands on the soil surface using trailing hoses (Rodhe, 2004). Shallow injection, a method involving incorporation of the slurry into the soil, is so far unusual.

Band spreading and shallow injection are favourable for reducing ammonia volatilisation, but their costs are higher than for broadcasting (Huijsmans *et al.*, 2004). Band spreading is also preferable to broadcasting for application in a growing crop due to lower risk of contamination of the crop (Steineck *et al.*, 2000), thus extending the time when manure can be spread. Slurry spreading requires in many cases intensive field traffic with heavy vehicles. This leads to soil compaction, which affects plant growth and production costs. Spreading slurry on soils with high moisture contents, *e.g.* wet clay soils during spring, causes considerable compaction, and thus yield losses (Arvidsson & Håkansson, 1991). Deep subsoil compaction is virtually permanent in clay soils, and should if possible be avoided (Håkansson & Reeder, 1994).

Wastewater management

Conventional wastewater systems

In Sweden, the vast majority (85%) of households are served by large-scale wastewater treatment plants (Statistics Sweden, 2004). The most commonly used treatment method includes mechanical, chemical and biological BOD and nitrogen treatment. Statistics Sweden (2004) reported an average removal of nitrogen and phosphorus to 56% and 95%, respectively, in 2002. For the largest treatment plants (>100 000 pe), the removal of nitrogen was 65%. The discharge of nitrogen from wastewater treatment plants to water is calculated to be 18% of the total anthropogenic discharge, and the corresponding figure for phosphorus is 16% (Brandt & Ejhed, 2002).

Continual improvements to reduce discharge from municipal wastewater treatment plants have not been matched by similar improvements in on-site systems. According to a questionnaire survey addressed to municipalities, only about 60% of on-site facilities in Sweden were reported to fulfil legal requirements (Ejhed *et al.*, 2004), *i.e.* further treatment of the effluent from the septic tank. Rough estimates of the anthropogenic phosphorus load to water indicate that approximately 20% originates from on-site systems, while the nitrogen load is considerably lower (Brandt & Ejhed, 2002). In the new guidelines from Swedish EPA (2006) for small-scale wastewater treatment, functional requirements have replaced the earlier technical requirements, which primarily promoted subsoil infiltration and sand filter beds. The new guidelines also identify the municipality as being responsible for providing conditions for beneficial use of sewage products, *e.g.* by establishing systems for collection, treatment, storage and transfer to farmland.

Source separating wastewater systems

Urine separation

Urine separation is based on a toilet with two outlets; one for urine and one for faeces and toilet paper. The urine is often conducted to a collection tank, while faeces may be handled separately or together with the other wastewater fractions.

There is growing interest in source separating system in different parts of the world. So far, many systems are in their infancy and experiences from large-scale are rare. Sweden is the country in Europe with by far the largest number of urine separating systems currently installed. Urine separation is, however, not a new phenomenon in Sweden. By the end of the 19th century, 20,000 urine separating closets were installed in Stockholm as a means to reduce the smell and increase the emptying intervals of the latrine buckets (Kvarnström *et al.*, 2006). Recycling systems based on source separation of urine have been implemented in several Swedish municipalities in the past decade as a result of promotion and requirements from the local authorities, *e.g.* in the municipalities of Norrköping, Linköping, Västervik and Tanum. Furthermore, many systems with urine separation have been initiated by individuals or groups of individuals striving towards a more sustainable way of living in ‘eco villages’. At least 15,000 urine separating porcelain toilets are estimated to be installed in Sweden (Kvarnström *et al.*, 2006). Together with all installations in plastic, presumably mainly in holiday homes, the total number of urine separation units installed is estimated at 135,000.

Urine separation has also been implemented in other parts of the world. In the eThekwini (Durban) municipality, South Africa, waterborne sanitation is not considered a viable option for sparsely populated areas. By 2005, over 20,000 urine separation toilets had been installed (Macleod, 2005), and the number is continually increasing. Research on environmental aspects relating to the sanitation concept is being performed in partnership with the University of KwaZulu Natal. Although on a more moderate scale, urine separation has also been introduced in other African countries (Klutze & Ahlgren, 2005; Morgan, 2005). In China, the number of urine separating systems is growing faster than anywhere else. In 2005, it was reported that approximately 700,000 urine separating toilets had been installed since the late 1990s (Werner, 2005).

There are different strategies for handling of the separated urine. Existing large-scale recycling systems rely on a use of stored but otherwise unprocessed urine, sometimes mixed with flushwater. There have also been several research projects on methods for concentration of urine (Lind *et al.*, 2000; Maurer *et al.*, 2006). Although existing systems rely on separate pipe systems for urine, solutions using the existing wastewater system have also been proposed. The transdisciplinary research project Novaquatis investigated future scenarios where the urine was stored in a tank integrated in each toilet and released through the conventional sewer system *e.g.* at night when the risk of pollution and dilution normally is lower (Larsen & Gujer, 1996). The urine could then be collected and treated separately.

The hygiene risks related to handling of source separated urine mainly depend on faecal cross-contamination as a result of misplaced faeces. Storage time, concentration and temperature affect the microbial reduction. Experimental studies, measurements on existing systems and hygiene risk assessments have concluded that recycling of urine to arable land is associated with only a low risk of gastro-intestinal infections (Höglund, 2001). Based on those findings, guidelines for the reuse of human urine have been adopted.

Discharge of pharmaceuticals to recipient waters is a growing concern since pharmaceuticals are only partly eliminated in wastewater treatment plants (Larsen *et al.*, 2004). A literature review of 212 pharmaceuticals revealed that on average, nearly two-thirds of each active ingredient was excreted via urine and one-third via faeces (Lienert *et al.*, 2007). However, there was an extreme variability not only between different therapeutic groups, but also within some groups and even some products. The high concentration of medical residues in urine poses both possibilities and obstacles. The small volume of urine makes treatment more feasible compared with treatment of the entire wastewater fraction. The use on arable land also provides a possibility for degradation of the pharmaceuticals in biologically active soil. However, more research on the fate and degradation of pharmaceuticals and hormones in soil is needed, as well as risk assessment of this practice compared with release into a water body.

Blackwater separation

Separation of blackwater implies that the closet water, *i.e.* urine, faeces, toilet paper and flushwater, is collected separately from the other wastewater fractions. Low-flush toilets are in general preferred in order to decrease the amount of flushwater to be handled by the system. Occasionally, blackwater systems use vacuum for transporting the material to the collection tank (Otterpohl, 2001). In Sweden, the blackwater fraction generated is normally transported by truck for further treatment in a municipal wastewater treatment plant, but there are also possibilities for reuse of treated blackwater as fertiliser. According to a national survey, 13% of the on-site facilities included in the study were reported to handle the blackwater separately (Ejhed *et al.*, 2004).

Microbial inactivation can be achieved through storage, thermal treatment or chemical treatment (Vinnerås, 2002). Storage is not considered a reliable method for material low in free ammonia. The inactivation is higher at higher temperatures, negatively affecting the performance in countries with a cold climate. Thermal treatment can be achieved through an external heat source, *e.g.* pasteurisation as an additional step in a biogas process, or through liquid composting. Liquid composting is a thermophilic aerobic process for organic liquid waste. Aeration is the most energy-consuming part of the process. Skjelhaugen (1999) reported the energy consumption for two commercial reactors to be 17 kWh per m^3 treated substrate for a 32 m^3 reactor, and 24 kWh per m^3 for a 17.5 m^3 reactor. Foam cutting and mixing, especially needed when processing livestock slurry, also required additional electricity (5-6 kWh per m^3 treated substrate). Through the temperature achieved in the process, the content of thermotolerant coliform bacteria, used as an indicator for pathogens, was reduced to below the limit value set by Norwegian authorities (Skjelhaugen, 1999).

Chemical treatment can be achieved mainly by adding acids, bases and/or oxidising agents (Vinnerås, 2002). For a sewage product intended for agricultural use, a chemical increasing the fertiliser value is of particular interest. By adding ammonia or an ammonia-based product, *e.g.* urea, to a sewage substrate, pathogenic bacteria can be efficiently inactivated, while at the same time the fertiliser value of the treated material is increased as the nitrogen is not consumed during the treatment. The antimicrobial effect of urea treatment has been evaluated for cattle manure (Park & Diez-Gonzalez, 2003; Ottoson *et al.*, in press) and also for reduction of bacterial and parasitic pathogens in faeces (Nordin, 2006; Vinnerås, 2007).

Sewage products as fertiliser

The recommended fertiliser application rate to a crop depends on its expected yield and is also related to the capacity of the soil to deliver plant nutrients. Different sewage products have different characteristics as regards *e.g.* plant nutrient content, fertilising effects and physical properties, thus requiring different application strategies. Combining mineral fertilisers with sewage products makes it possible to optimise the fertilisation strategy. An advantage with this combination is that mineral fertilisers can be used when the risk of soil compaction is high, since the equipment used for spreading sewage products is normally heavier than the equipment used for spreading mineral fertilisers.

In Table 1, the current use of mineral fertilisers in Swedish agriculture is compared with the content of plant nutrients in different sewage fractions.

Table 1. *Use of nitrogen, phosphorus and potassium in mineral fertiliser products in Sweden, compared with the amounts in different sewage fractions from the Swedish population (tonnes per year)*

	Nitrogen	Phosphorus	Potassium	Source of data
Mineral fertiliser use	158 000	14 000	28 000	Statistics Sweden, 2006
Found in sewage sludge	9 200	6 700	*Not measured* [a]	Statistics Sweden, 2004
Expected in urine[b]	37 000	3 300	9 100	Vinnerås *et al.*, 2006
Expected in faeces[b]	5 000	1 700	3 300	Vinnerås *et al.*, 2006

[a] Roughly 500 tonnes based on figures from Andersson & Nilsson (1999) and dry matter production from Statistics Sweden (2004)

[b] Calculated from proposed design values and 9.1 million inhabitants

Sewage sludge

Sewage sludge is mainly considered a phosphorus fertiliser, but its content of organic matter and nitrogen is also valuable in agriculture. Numerous studies have investigated the extent to which phosphorus in sewage sludge is plant-available. According to literature reviews, the results are ambiguous (Johansson, 2000; Kvarnström, 2001). The discrepancies between different studies in phosphorus availability from sludge can be attributed to several factors, *e.g.* differences in origins and treatment of the sludge, analysis and experimental techniques and length of the experiments. Johansson (2000) concluded from a literature review that fresh sludge appeared to have higher availability than dried sludge. However, this was not confirmed in an experimental study by Kvarnström *et al.* (2000), in which the dewatering process did not change the availability of the phosphorus significantly.

An evaluation of the long-term effects of sludge use showed that the relative phosphorus availability of sludge compared to water-soluble phosphorus fertilisers was 60%, but this value was not statistically different from 100% (Kvarnström, 2001). A conclusion based on Swedish pot experiments was that twice as much phosphorus was required in sludge compared with mineral fertiliser to achieve the same effect (Ottabong, 2003).

Nitrogen in sewage sludge is primarily present in organically bound form, and must thus be mineralised before it becomes available for uptake by the crop. The rate at which organically bound nitrogen is mineralised is not easy to predict and can differ considerably between years. If the content of NH_4-N is low, the application time becomes less critical for spreading *e.g.* in the autumn.

The sewage sludge composition reflects the surrounding society. Measures for better control of the flows of hazardous substances are therefore required. Levlin *et al.* (2001) reported that five of the regulated metals in sewage sludge have decreased by more than 80% during the period 1969-1998. Their data also showed that approximately half of all sludge samples had either heavy metals or organic pollutants exceeding the permitted limits.

Human urine

The concentration of plant nutrients in stored human urine reflects the amount of flushwater used. Urine contains most of the nitrogen excreted by humans (Table 1). Most of the nitrogen and phosphorus in urine is directly available for plants. According to field trials and pot experiments, the fertiliser effect of phosphorus and ammonium after volatilisation is comparable to that of mineral fertilisers (Kirchmann & Pettersson, 1995; Richert Stintzing *et al.*, 2001). With a high pH in urine and most of the nitrogen occurring as NH_4^+, the risk of ammonia volatilisation is high. Covered storage and cautious spreading, which minimises the ammonia losses, are therefore recommended. Spreading the urine directly before sowing or in the growing crop with trailing hoses are strategies that allow for efficient use of the plant nutrients.

Blackwater

As with separated urine, the concentration of blackwater is inversely related to the amount of flushwater used. By also collecting faecal matter, most of the phosphorus in household wastewater can be recovered and recycled to arable land (Vinnerås *et al.*, 2006). Hygiene aspects need to be thoroughly considered when handling blackwater. The sanitising strategy can also affect the nutrient content, *e.g*. liquid composting can increase the content of NH_4-N during the process (Norin, 1996). Using urea or ammonia enhances the fertiliser value as regards nitrogen. However, a higher nitrogen content in sewage products also poses a higher risk of ammonia volatilisation during storage and spreading. A strategy for the agricultural handling must thus be worked out.

Sustainability indicators for wastewater management

Sustainability aspects of wastewater systems have been increasingly in focus since the 1990s. In the Urban Water programme, funded by MISTRA, a comprehensive analysis of sustainability embraced users, organisation and technology and covered five perspectives: health, environment, economy, socioculture and technical function (Figure 2). Each perspective could be linked to sustainability criteria with indicators for a further assessment (Malmqvist *et al.*, 2006).

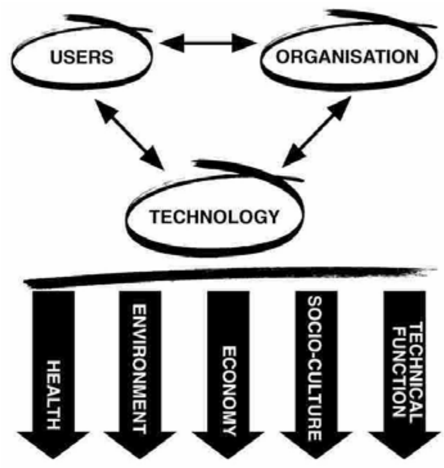

Figure 2. The conceptual framework guiding the projects in the Urban Water programme.

Sets of indicators have been developed in several countries (Balkema *et al.*, 2002). A life cycle perspective is frequently used for capturing sustainability indicators of the environmental performance. Typically, these indicators reflect the performance of the wastewater treatment and the effluent quality, but in several cases resource utilisation and recycling are also included. Hellström *et al.* (2000) considered among other indicators the potential recycling of phosphorus. Indicators of recycling of both nitrogen and phosphorus are addressed by several authors (Lundin *et al.*, 1999; Mels *et al.*, 1999; Lundin & Morrisson, 2002). Palme *et al.* (2005) expressed the indicator for recycling as 'P and N that is recycled and thereby forms a potential substitute for artificial fertilizers'.

LCA methodology

Life cycle assessment (LCA) is a method used for analysing complex systems in an organised way. LCA aims to evaluate the environmental burdens associated with a certain product or service in a cradle-to-grave perspective, from raw material extraction to waste management and final disposal. Standardisation is made through the framework of ISO, the International Organization for Standardization (ISO, 2006a,b). According to those standards, the LCA technique with appropriate justification can also be applied in studies that are not defined as LCA, *e.g.* cradle-to-gate studies or studies on specific parts of the life cycle such as waste management. There are also a number of guidelines available for different audiences and with a different focus (*e.g.* Lindfors *et al.*, 1995; Guinée, 2001; Udo de Haes *et al.*, 2002; Baumann & Tillman, 2004).

An LCA includes different phases: goal and scope definition, inventory analysis, impact assessment and interpretation (Figure 3). The interpreted results may then be input in a decision-making process.

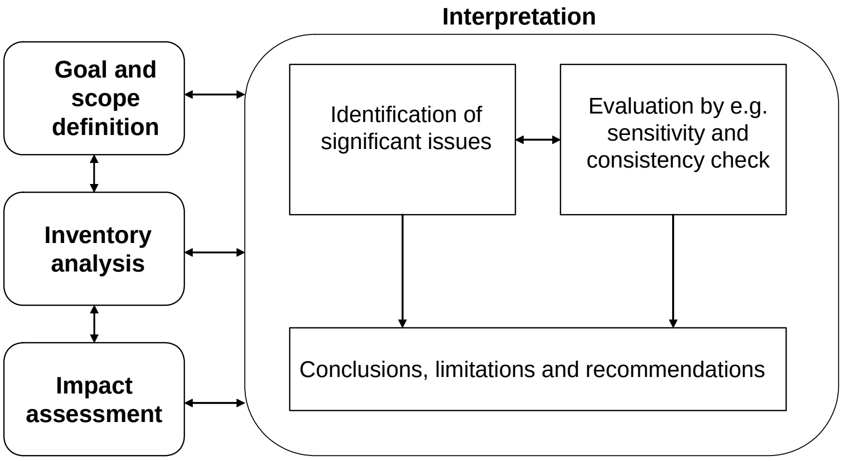

Figure 3. A life cycle assessment framework (modified from ISO, 2006b).

The goal and scope definition is a critical part of LCA, as the results will depend on how the system, the functional unit and system boundaries are defined (Lindfors *et al.*, 1995). The functional unit is a well-defined measure of the main function of the system or what the system delivers, to which all environmental inputs are related. The inventory analysis includes a detailed description of the functions and boundaries of the system, data collection and calculation. When more than one product is produced in a process or when several products undergo the same process, allocation procedures are needed. According to ISO (2006a), allocation is defined as 'partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems'. In the impact assessment, the results from the inventory are aggregated. Different environmental impacts are classified into impact categories and quantitatively characterised. A valuation weights different environmental impacts against each other using a mixture of scientific, political, ethical and administrative considerations (Lindfors *et al.*, 1995). If weighting is used, it is desirable to use several different types, since the outcome might differ substantially between different methods (ISO, 2006b).

Two different types of LCA have been distinguished; accounting and change-oriented LCA (Baumann & Tillman, 2004). However, there are different terms in use for describing those two types of LCA. Accounting LCA is also called retrospective or attributional LCA, while change-oriented LCA is sometimes called prospective or consequential LCA (Ekvall & Weidema, 2004; Ekvall *et al.*, 2005). The type of LCA used depends on the aim of the study. Different approaches are thus relevant to different situations and have implications on methodological issues considering *e.g.* allocation procedure and data choices (Tillman, 2000; Ekvall & Weidema, 2004). An accounting LCA aims to describe relevant environmental flows from raw material extraction to waste management and is characterised by partitioning of environmental burdens and use of average data. A change-oriented LCA focuses on those parts that differ between alternatives, reflecting the effects of a change by using system expansion. Marginal data are more often used. However, there are also limitations with a change-oriented approach. Modelling the full consequences of a choice is difficult, and effects on the market due to a change in demand might influence other systems (Ekvall *et al.*, 2005).

Previous life cycle studies on source separating systems

Both national and international studies have evaluated different wastewater systems using a life cycle approach (see *e.g.* Lundin, 2003 for a compilation). Below, some previous studies on source separating systems are presented.

The simulation model ORWARE has been used for comparing the environmental performance of different operational options of waste and wastewater handling (Sonesson *et al.*, 1997). The results showed that the eutrophying emissions and the electricity use decreased when urine separation was implemented. Increased transport led to an increased use of oil and increased the air emissions related to such transport. A later ORWARE study, but with measured and thus better data on the urine separating system, arrived at similar conclusions (Jönsson, 2002). The ORWARE model has also been used in the research programme Urban Water, and has gradually been transformed into URWARE. The model has been used for comparing the operation of a high-tech combined system with a source separating system including the transport to arable land, but not the avoided use of fertiliser production (Jeppsson & Hellström, 2002). It was concluded that the systems scored differently on different environmental criteria, thus giving decision-makers an important role in defining which system is superior. When comparing the operation of a well-functioning conventional system with urine separation with dry faecal handling, the latter has been found to have advantages concerning eutrophication, energy use and recycled nutrients (Jönsson *et al.*, 2005).

Kärrman (2001) also came to the conclusion that the operation of a urine separating system was potentially more energy-efficient compared with a conventional system when the avoided use of mineral fertilisers was included. On the other hand, a blackwater system relying on liquid composting for sanitation proved to have high energy consumption due to the use of electricity for the vacuum system and the reactor.

Tillman *et al.* (1998) and Lundin *et al.* (2000) evaluated the construction and operation of separating systems including the avoided use of fertiliser production in comparison with conventional systems. The urine separation system was considered preferable in many environmental aspects. The energy use related to the investment phase was shown to be sensitive to changes in lifetime and choice of different technical components. Lundin *et al.* (2000) considered the differences in the plant-availability in different sewage products and concluded that mineral fertiliser production and agricultural practices should be included within the system boundaries when separating systems are assessed.

Maurer *et al*. (2003) examined different removal and recovery techniques for nutrients in urine using a life cycle perspective. Their analysis showed that recovery has the potential to be more energy-efficient than removal in a WWTP combined with new production of mineral fertilisers. The field application and the collection system at household level were emphasised as crucial parts of the system deserving particular attention.

A comprehensive Danish assessment of different waste and wastewater options for a medium-sized town revealed that strategies based on source separation followed by agricultural use of the products could be beneficial both as regards energy consumption and recycling (Magid *et al.*, 2006). It was also concluded that source separating systems could be operated close to the cost level of conventional wastewater systems.

In an EU demonstration project performed in Berlin, source separating concepts were evaluated using *e.g.* LCA, cost calculations and field trials (Peter-Fröhlich *et al.*, 2007). It was concluded that the source separating alternatives had smaller environmental impacts than conventional systems if substitution of mineral fertilisers and energy production via faeces digestion were achieved.

A life cycle perspective on systems for recycling sewage nutrients

A life cycle approach to wastewater systems aimed at recycling plant nutrients emphasises aspects considering both the wastewater sector and agriculture and this involves methodological considerations that might influence the results. In this chapter, methodological aspects of Papers I-III are described and environmental benefits and drawbacks with the different recycling systems considered in those three papers are discussed.

System boundaries

The system boundaries differentiate the system under analysis from its environment. Several dimensions must be considered, such as boundaries relating to natural systems, time, geography and production capital and boundaries relating to life cycles of other products (Tillman *et al.*, 1994).

One example of a methodological difficulty when LCA is applied to agricultural production is whether agricultural soil should be considered part of the production system or part of the environment. In Papers I and II, emissions of nitrogen and phosphorus from arable land were defined as emissions when leaving the root-zone. One could thus say that the upper part of the soil was considered part of the production system. In order to draw conclusions on *e.g.* heavy metals and nutrients accumulating in the soil, balance accounting was performed.

The chosen time horizon for the three studies was one year. It was assumed that the separating system would be gradually implemented in the near future and currently available techniques were used. However, during the lifetime (approximately 30 years) of the separating systems, other changes that were not considered in the study may occur. One example is whether mineral fertilisers will be produced in the future using significantly more energy-efficient processes. The energy figure for ammonia production was a weighted value, including 20% of the production being carried out in less efficient plants in Central and Western Europe (Davis & Haglund, 1999). There is obvious room for improvement as regards both nitrogen and phosphorus fertiliser products (Jenssen & Kongshaug, 2003), but still uncertainty about the rate at which those production plants might be replaced by more efficient systems.

The studies performed were geographically placed in a Swedish context both as regards the agricultural conditions and the technology used for wastewater handling. The wheat and oat production in Papers I and II, respectively, was assumed to be performed according to current practices in the eastern part of Sweden surrounding lake Mälaren, an area dominated by sedimentary clay soils. The wastewater systems assumed were essentially based on the performance of existing Swedish systems. Although originating from rather site-specific data, the conclusions drawn from the studies are also applicable for other countries using similar production/technology. Many of the identified hot-spots, *i.e.* activities in the life cycle causing a significant environmental impact, are aspects that also need to be emphasised in recycling systems based on other technologies or applied in other production systems, *e.g.* the importance of a high substitution rate of mineral fertilisers and a carefully constructed system for collection.

The investment phase is considered to be of minor importance when evaluating large-scale wastewater treatment in a life cycle perspective, while investments associated with small-scale wastewater systems may make considerable contributions (Tillman *et al.*, 1998). As regards life cycle assessments of agricultural systems, investments are often omitted. In Papers I-III, only the infrastructural investments that differed between the alternatives evaluated were considered. Therefore, infrastructural investments in pipes and storage tanks intended for separating the urine were included in Paper I, while the infrastructure relating to wastewater handling for the remaining fractions was omitted, since it was assumed that separation of urine would not affect the wastewater treatment of the remaining fractions. The results from the three studies clearly demonstrate that the investment phase can have a considerable impact on the energy use of the systems.

When the life cycles of different products are interconnected, an allocation problem might occur. By expanding the system, allocation can be avoided. System expansion can be implemented by either adding subsystems providing missing functions, or by subtracting subsystems with excessive functions (Lindfors *et al.*, 1995). The principal approach in all three papers was to use system expansion, either by adding or subtracting subsystems.

Papers I-III all had different aims and scopes and thus the system boundaries were defined differently. Paper I focused on the agricultural system, including activities relevant to wheat production, while the scenario using human urine also accounted for the changes associated with the introduction of the urine separating system. In both Papers I and II, all relevant activities relating to grain production were included. Field operations considered were stubble cultivation or ploughing, harrowing, sowing, fertilising, pesticide application and harvesting. Paper II considered both the agricultural and the wastewater systems, with special emphasis on changes occurring when a blackwater system was assumed to complement the conventional system. Figure 4 illustrates the blackwater system evaluated in Paper II. Paper III emphasised the wastewater system, while the agricultural use was included only as avoided use of mineral fertilisers. The main focus in Paper III was on the differences between the three alternatives for upgrading existing on-site systems.

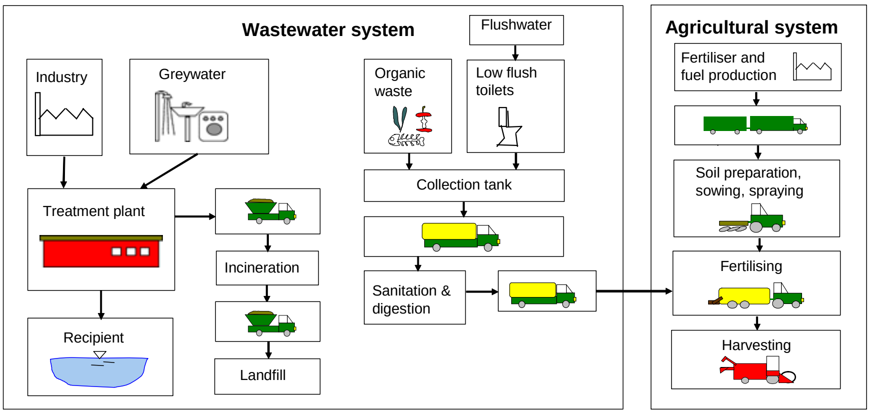

Figure 4. Flow chart for the blackwater system evaluated in Paper II.

The three studies performed considered both environmental impacts related to current production systems and technology, and changes associated with new strategies for wastewater management. There was thus no pure adoption of either an accounting or a change-oriented approach, since both approaches were represented in the studies. One reason for this was that the aim was both to illustrate the chain of activities the current handling is accounted for by highlighting hotspots in the different systems and to examine the consequences of changes. The impact from *e.g.* an altered fertilisation strategy could thus be compared with other steps involved in the grain production system. Although the distinction between the accounting and change-oriented approaches can be a support when structuring a study, a combination of the two types was found to be most relevant in the present studies in revealing the relative importance of changes.

Land use

Land use is associated with different types of environmental impacts. Human land use both occupies land that could be used for other purposes, and has an impact on biodiversity and life-support functions (Guinée, 2001). Land use issues initially received limited attention in LCA methodology, although the environmental impact on land is considered very important for *e.g.* agriculture (Finnveden & Lindfors, 1996). However, the issue has received increasing attention since the late 1990s. Land use has traditionally been used only to describe the land area occupied by an activity. These data are generally combined with the time required to produce a certain output and also with qualifications of the land under use or change (Lindeijer, 2000). A special task for agricultural LCA has thus been to focus on and expand this impact category, as land use has a direct impact on physical, chemical and biological properties of the arable land.

Indicators for land use

Different authors have suggested sets of indicators and criteria for land use, but no harmonisation has been reached so far. Mattsson *et al.* (2000) suggested indicators for soil properties according to a goal aimed at ensuring biological production and quality of the land. Proposed indicators are *e.g.* erosion, soil organic matter, soil structure, plant nutrient status, pH and impact on biodiversity.

Cowell & Clift (2000) also suggested a methodology for assessing soil quantity and quality in LCA. Relevant factors affecting soil properties that should be considered are *e.g.* losses due to erosion, weeds, pathogens, nutrients, heavy metals, pesticide residues, salts, pH, organic matter, and soil water, texture and structure. They suggested that when *e.g.* nutrients and pH are not maintained at the same productive level, an approach can be to model the activities required to restore the soil to the level prior to the time period under analysis. Similar factors have also been presented by Audsley *et al.* (1997), who discussed whether it is possible to aggregate quantitative data for soil quality into a value related to its impact on the potential crop yields. The conclusion was that further research is needed regarding whether aggregation of these values into one single parameter is possible, or even appropriate. Others do not recommend aggregation of the data collected, at least when considering agriculture. Mattsson et al. (2000) proposed that the land use category should also include non-aggregated and descriptive parts.

Application of sewage products may have an impact on land use. In Papers I and II, the main approach was to include differences in land use within other impact categories as far as possible. The direct yield losses due to soil compaction in the upper layer and wheel traffic were thus reflected in the functional unit. The long-term effects of yield losses due to soil compaction in the subsoil and the reduced need for future application of mineral fertilisers due to mineralisation of organically bound nitrogen in the sewage products were both transferred to the year under study. Hypothetically, it would also be possible to include the yield-increasing effect due to improved soil structure resulting from the application of organic matter. However, this was not done in these studies, since the basis for quantifying this effect was rather weak. Element balances seem to be a suitable method for assessing accumulation of heavy metals and nutrients and this approach was used for nitrogen, phosphorus and cadmium in Paper I.

Land requirement

Land use is an inevitable consequence of agricultural production. An interesting issue is how to evaluate systems producing similar products, quantitatively and qualitatively, but with different land requirements. Gärtner & Reinhardt (2001) argued that the fate of the area not occupied for the production concerned should be considered by modelling its alternative land use. In their study, they considered a reference system with the same land use and the same main function as the system in question.

Different use of land occurs when comparing *e.g.* conventional and organic production of agricultural products, since organic farming often involves a lower yield per hectare, thus requiring a larger area for producing an equal quantity. Another example when the yield might differ between different alternatives is when the use of heavy vehicles gives rise to soil compaction, thus affecting the yield in both the short-term and the long-term.

The above methodological issue was addressed in Paper II. The functional unit included a specified quantity of grain on a specified area. There are certain options reflecting an alternative use for the released area, *i.e.* the area not needed for the production of grain in the functional unit. In Paper II, this released area was assumed to be set-aside area. Set-aside subsidies have been an established part of the common agricultural policy within the EU, aimed at decreasing agricultural over-production. For this reason, we considered that the most relevant alternative use for abandoned area in the short-term for Swedish conditions was expected to be fallow set-aside. However, the area required for *e.g.* energy crop production could be expected to increase greatly in the future, thus giving priority to other more probable alternative use for the released area.

A change-oriented LCA focuses on changes between alternatives and reflects these changes by expanding the system boundaries (Tillman, 2000). If agricultural systems under comparison require different areas of land for producing the same quantity, the fate of the released area not needed in alternative(s) with the highest yield per ha is an interesting aspect to consider. Even an abandoned area, *e.g.* set-aside area, is associated with (minor) leaching, as well as air emissions. Areas of land not required for food or feed production could also be used for energy crop production. A suggestion based on the results arrived at in this thesis is that the difference in area requirement should be considered if a change-oriented perspective is applied. In such cases, our results suggest that the most probable fate for the released areas not needed for the actual production should be taken into account by expanding the system boundaries to include the same area for all alternatives under study, as the area-specific emissions are otherwise very difficult to handle. The use of this area is decided by time and site-specific conditions.

Abiotic resource use

Use of abiotic resources has long been in focus in LCA. The term 'resource depletion' is often used for metals, but is misleading according to Steward & Weidema (2005), since metals only can be dissipated, not depleted. They also argue that the extraction is not the main problem, rather the dissipative use and disposal. There are various proposals on how to evaluate the use of abiotic resources. Some methods are based on measures of current extraction rate and deposits, referring to the total quantity of resource deposits (Udo de Haes *et al.*, 2002). However, this approach can be very uncertain, since the availability of the total resource stock is highly variable and often includes low grade resources not immediately available for human needs. Referring to the reserves extractable at current prices is another approach, which was used in Paper II. A disadvantage with this approach is that short-term price fluctuations can affect the size of the reserves. An additional option is to model the future consequences of resource extraction (Müller-Wenk, 1998; Steen, 2006), whereby impacts related to future extraction from resources with lower concentrations can be taken into account. A conclusion from Steen (2006) is that the perceived importance of resource use is related to the time perspective applied. In only a short time horizon, other environmental impacts are considered more important. If a historical perspective is applied, however, the current use might become a massive problem.

Resource use is of interest for all kinds of human activities. However, this aspect might play an exceptional role in systems recycling sewage nutrients, since arguments for recovering phosphorus for agricultural use are used to motivate implementation of such systems in Sweden. Efficient resource use requires careful attention to all resources associated with wastewater management in order to avoid sub-optimisation. Hence, there is a need to evaluate not only phosphorus, but also use of other resources required for recovering plant nutrients. The intricate side of this issue is highlighted in Paper II. The blackwater system modelled in the study replaced virgin phosphorus at the expense of higher use of energy. Recycling phosphorus in sewage may also involve a decrease in the overall sulphur requirement, since large amounts of sulphur are required when producing phosphoric acid – an intermediate product in the production of phosphorus fertiliser products. In Paper II, use of different abiotic resources was compared using a weighting method based on the equation $w = 1/(RU)$, where w is the non-renewable resource index, R is the size of the reserve and U is the static reserve life. Including sulphur in the weighting method had a decisive influence on the results and outweighed the other resources. However, including sulphur could be questioned since almost all sulphur on the world market is a by-product originating from compulsory processing of crude oil and natural gas in order to meet environmental regulations, something that lowers the price of sulphur and thus also the reserve and the static reserve life.

Eutrophication

Eutrophication includes impacts on both terrestrial and aquatic systems. Terrestrial eutrophication covers the negative effects of excess nutrients (ammonia and nitrogen oxide) on plants and species composition. Terrestrial eutrophication is recommended to be included in LCA, but is not usually characterised in LCA at present (Udo de Haes *et al.*, 2002).

Aquatic eutrophication is defined as nutrient enrichment of the aquatic environment leading to a chain of ecological effects (Udo de Haes *et al.*, 2002). During the decomposition of the biomass, oxygen is required. In Europe, most freshwater is phosphorus-limited, whereas in marine waters, nitrogen is considered to limit production (Udo de Haes *et al.*, 2002). However, it is an oversimplification that only one single nutrient limits phytoplankton growth according to most researchers (Rabalais, 2002). An international evaluation of the nutrient enrichment in the Baltic Sea stressed that the efforts on decreasing the phosphorus load to the Baltic sea need to be intensified (Swedish Environmental Objectives Council, 2007).

Weighting factors for eutrophication have been presented by *e.g.* Lindfors *et al.* (1995). These weighting factors do not account for the primary oxygen demand due to nitrification, *i.e.* when ammonia to air and ammonium to water are oxidised to nitrate in the recipient waters. Therefore, higher weighting factors for oxidation of ammonia and ammonium were proposed by Kärrman & Jönsson (2001a). In Papers I-II, those latter, updated weighting factors were used.

As algal growth in different aquatic systems is limited by different nutrients, a general characterisation raises questions. How to account for nitrogen emissions to air is also problematic, as only a fraction of the emissions reaches the aquatic system (Lindfors *et al.*, 1995). In order to overcome this dilemma, region-specific fate factors have been presented for direct deposition of ammonia and nitrogen oxide emitted to air in the European marine environment (Huijbregts & Seppälä, 2000). In addition to direct deposition, air emissions could also reach the aquatic environment through run-off and leaching after deposition on the soil. Fate factors for this have been presented for the Netherlands, Europe and the world (Huijbregts & Seppälä, 2001). No regional factors for Sweden are currently at hand. In the papers included in this thesis, water emissions totally dominated the emissions of eutrophying substances. For this reason, no region-specific emission factors for nitrogen emitted to air were used. Instead, the results were presented using emission factors from a maximum scenario, *i.e.* both nitrogen and phosphorus were assumed to contribute to eutrophication.

Environmental benefits and drawbacks with nutrient recycling systems

Nutrient recycling from wastewater systems to farmland may be associated with both environmental benefits and drawback. Below, some aspects relating to the substitution of mineral fertilisers are considered and results from Papers I-III as regards energy use and emissions to water and air are presented and discussed in relation to other relevant studies.

Impacts relating to fertilisation

The evaluation of the conventional wheat production system in Paper I and the oat production system in Paper II revealed that the production of nitrogen mineral fertiliser makes a substantial impact on both energy use and contribution to global warming compared with other activities in the grain production system. An LCA study on wheat production addressed this issue by modelling the response when different nitrogen fertiliser rates were used (Brentrup *et al.*, 2004). The substitution rate of mineral fertilisers is thus an important aspect to consider when evaluating the environmental performance of systems recycling sewage fertilisers. Generally, blackwater separation systems provided high potential substitution rate both of nitrogen and phosphorus, while chemical precipitation systems meant that primarily phosphorus was recycled (Paper III). Urine separation proved more interesting for substituting nitrogen than phosphorus.

The sensitivity analyses performed in Papers I and II examined different spreading strategies for urine and blackwater. Some of the strategies involved a low utilisation of nitrogen and thus a low substitution rate of mineral fertiliser, which influenced the environmental performance of the whole system. As described in those papers, there might be conflicts between high utilisation of the nitrogen in urine and blackwater and minimisation of soil compaction on clay soils. This was obvious if *e.g.* a heavy spreader with high capacity was assumed to be used on a clay soil early in spring, when nitrogen utilisation is high, but the soil also is wet and sensitive to compaction (Paper I). The yield reduction was calculated to be 14% when both immediate and future effects from the soil compaction were allocated to the year under study. Spreading in the early autumn or in the late spring involved considerably lower yield losses as the soil was then dryer, but only the spreading in the late spring meant that the nitrogen in the urine was well utilised. Thus, the results clearly demonstrate the importance of optimal spreading, *i.e.* when the risk of soil compaction is low and plant nutrients are required by the crop. In this respect, equipment for spreading liquid fertilisers in the growing crop, *e.g.* band spreading, is suitable.

Spreading of phosphorus is less vulnerable to direct losses. An essential factor influencing the substitution rate of phosphorus is the extent to which phosphorus in *e.g.* sewage sludge is plant-available. The plant-availability reported from different studies shows a high variability (Johansson, 2000; Kvarnström, 2001), making this aspect extremely difficult to evaluate. In Papers II and III, 50% of the phosphorus in sewage sludge was assumed to replace mineral phosphorus fertiliser. This assumption reflects a lower initial availability of the chemically precipitated phosphorus in sludge compared with phosphorus in mineral fertilisers. The high doses normally applied in Sweden when sewage sludge is used, sufficient to cover a whole seven-year crop rotation with phosphorus, also raise questions as to whether the plant-availability of the phosphorus applied with the sewage sludge will decrease over time. The uncertainty regarding the extent to which phosphorus in sewage sludge could replace mineral fertilisers in the long-term emphasises the need for cautious interpretation of this aspect.

The conclusion from these studies is that the fertilisation strategy used could have a larger impact on the substitution rate, especially of nitrogen, than estimation of the plant-availability of nutrients in the original sewage product. When evaluating different systems aimed at recycling, different potential fertilisation strategies and their impact on the substitution rate should therefore be examined.

Energy use

Whether a source separating system is more energy-efficient than a conventional system depends on several factors. The use of fossil fuels and electricity for wastewater management is related to both the construction and operation of wastewater systems. Papers I-III demonstrated that the construction phase of different goods required for separating wastewater fractions could make a significant contribution to the total energy use. The design, material and life-time is thus important to consider for an energy-efficient system (Tillman *et al.*, 1998; Lundin *et al.*, 2000; Papers I-II). Concrete tanks are preferable to plastic tanks in this respect, and long plastic pipes should, if possible, be avoided. The energy use associated with investments is larger for small-scale systems than for large scale (Tillman *et al.*, 1998). It is thus important to pay attention to the investments required for decentralised source separating systems when comparing those with conventional systems. Figure 5 illustrates the energy use associated with the grain production system evaluated in Paper I using different fertilisation strategies and with the urine assumed to be collected either in tanks in concrete or in plastics.

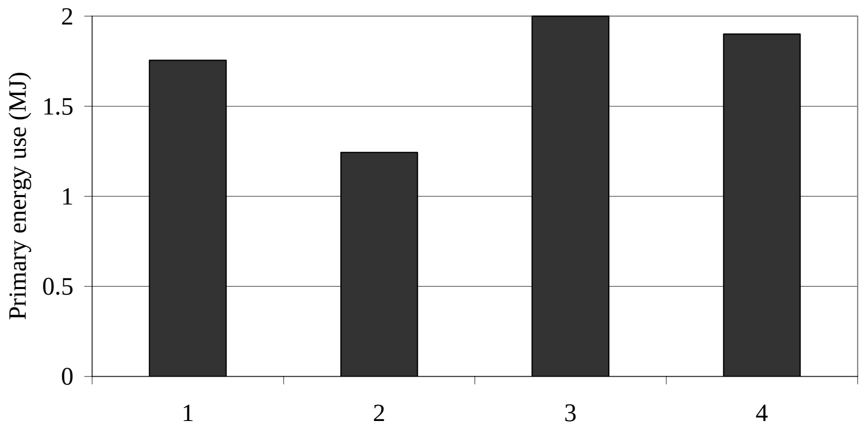

Figure 5. Energy use (MJ/kg) associated with a wheat production system using different collection and fertilisation strategies. 1) Only use of mineral fertilisers, 2) use of urine collected in concrete tanks, 3) use of urine collected in plastic tanks and 4) urine collected in concrete tanks but no substitution of mineral fertiliser assumed.

The extent to which nitrogen mineral fertiliser products were replaced by the source separating sewage products was one important aspect for the energy use, as exemplified by Figure 5. A separating system might also save electricity due to a reduced need for producing water for flushing, as well as a decreased need for pumping and treating wastewater. With moderate transport distances (10 km was assumed in Paper I), the energy use for transport could be kept rather low compared with other activities associated with the separating system.

Urine separation provides a sanitation option with only storage required for microbial inactivation. The volume of the separated urine fraction could also be kept relatively low, with no or only small amounts of flushwater added. Those two factors contributed to a lower energy use for urine separation than for blackwater separation in Paper III. The importance of the sanitation strategy for energy use in the blackwater separation system was clearly shown when liquid composting was replaced by chemical sanitation using urea. While ammonia-based sanitation is a promising option from an energy point of view, it also means that spreading must be performed carefully to minimise nitrogen losses through volatilisation and to avoid overdosing.

Chemical precipitation of phosphorus in septic tanks is an interesting and rather simple alternative for upgrading existing on-site systems. One major disadvantage identified in Paper III was the high use of precipitation chemicals. The energy use associated with this handling would be lower if the dose could be lowered, *e.g.* by being regulated by the actual flow instead of the anticipated flow.

Emissions to water

Source separation provides a measure for reducing the eutrophying load to recipient waters. This environmental effect was demonstrated as one of the most important aspects according to the normalisations made in Papers I and II, a result also in accordance with *e.g.* Kärrman & Jönsson (2001b). However, the change might be relatively small compared with the total nutrient discharge if urine separation was complementing an existing conventional wastewater treatment plant with efficient removal of nitrogen and phosphorus, as in Paper I. In this respect, source separation is therefore far more interesting for unsewered areas, *e.g.* for on-site systems. According to Paper III, blackwater separation proved an interesting alternative if both nitrogen and phosphorus have to be significantly reduced. Chemical precipitation in the septic tank was interesting for phosphorus reduction, while urine separation was highly efficient primarily for reducing nitrogen.

In Paper I, the efficiency of the treatment plant was assumed not to be affected by the urine separation. However, the effect of urine separation on the treatment efficiency is scale-dependent. A Dutch study modelling the effects on wastewater treatment plants when urine is separated illustrated that up to 60% separation would be beneficial for the nutrient removal processes (Wilsenach & van Loosdrecht, 2003). A study made with the URWARE model showed that connecting the greywater from a small community with urine and faecal separation to a large wastewater treatment plant actually decreased its total eutrophying emissions (Jönsson *et al.*, 2005).

Air emissions

In Papers I and II, greenhouse gas emissions and acidifying emissions were calculated. In both papers, the contribution to global warming potential (GWP) was of the same magnitude for all the systems studied. Although a reduced use of mineral fertilisers in the source separating scenarios resulted in lower emissions of GWP-gases, production of capital goods, transport and increased field emissions more or less outweighed those savings.

Emissions of CO_2 from electricity production are highly linked to the electricity mix used. The Swedish average is mainly produced from hydropower and nuclear power. However, a change in the electricity use might involve a change in the marginal production, produced from *e.g.* coal-based power plants. A sensitivity analysis in Paper I illustrated the effect of a change in electricity production if this instead was assumed to be produced by coal power plants. Although the GWP-emissions were dominated by emissions from sources other than the energy sector, the choice of data for electricity production had an effect on the GWP-emissions. The total GWP-emissions decreased by 17% and the emissions of CO_2 decreased by 44% in the urine-spreading scenario.

When emissions of NO_X and NH_3 were considered acidifying according to a maximum scenario, the potential acidification was higher for source separating systems compared with conventional systems (Papers I and II). Emissions of NO_X originated primarily from transportation and production of goods for separation and collection, while NH_3 was emitted during storage and spreading. Measures taken to reduce ammonia volatilisation are thus beneficial both for reducing the need for mineral fertiliser products and for reducing potential acidifying emissions. However, if NO_X and NH_3 act as fertilisers and are not leached from the system, no acidification from these substances will occur, and the acidifying effect will be overestimated (Lindfors *et al.*, 1995).

Trade off

Recycling and energy conservation are integral objectives in the Swedish Environmental Code. However, there are obviously systems that exhibit a trade-off between recycling rate and energy use. The results from Paper III demonstrated that a high reduction and recycling of phosphorus was achieved with the more energy-requiring systems (Table 2). A conclusion from Paper III is that the most environmentally favourable alternative depends on how important *e.g.* reduced emissions to water are considered in a specific situation.

Evaluating a recycling system for sewage nutrients only on the basis of amounts of plant nutrients recycled to farmland is inadequate, since other resources, *e.g.* use of fossil fuels and uranium, also need to be considered to avoid sub-optimisation. How to rank a system with increased recycling at the expense of increased energy use thus needs to be addressed.

Table 2. *Primary energy use, possible substitution rate and expected reduction to recipient water for systems evaluated in paper III. The substitution and reduction rates are compared to the amount originally found in the influent. Detergents without P assumed* [a]

Sanitation strategy	Energy use (MJ)	Substitution of P (%)	Reduction in P (%)	Reduction in N (%)
Urine separating + storage	172	41	74	80
Blackwater + liquid composting	1966	86	94	94
Blackwater + urea treatment	509	86	94	94
Chem. precipitation + urea treatment	1024	43	93	53

[a] Content of P in greywater taken from Swedish EPA (1995).

However, as described earlier, methods for weighting abiotic resource use are problematic and in need for further development. In the absence of suitable weighting methods, it seems reasonable that a recycling system claiming to be more resource-efficient than a conventional system also must emphasise energy-efficiency. Thus energy use throughout the whole recycling chain must be considered. Although afflicted with shortcomings, weighting methods could be used as a basis for discussion of sustainable resource use.

Organisational aspects of systems for recycling sewage nutrients

Nutrient recycling systems are far more than just a technical issue. The systems involve stakeholders with different roles and responsibilities, and may give rise to a multitude of conceivable collaboration forms. A comprehensive assessment of wastewater systems requires a trans-disciplinary approach, involving both organisational and institutional aspects (Malmqvist *et al.*, 2006). Organisational capacity is related to the capability of the organisation responsible to plan and operate *e.g.* a wastewater management system, while institutional capacity embraces a wider perspective, including laws, rules, procedures, routines, norms, shared perspectives etc. (Storbjörk & Söderberg, 2003).

The Urban Water programme used seven criteria as a checklist for successful implementation of sustainable urban water systems (Malmqvist *et al.*, 2006):

- Presence of policy entrepreneurs, such as initiators and implementers
- The sphere of action through legislative and political support
- A value coalition of shared views, problems and goals
- Access to financial resources and knowledge

- Explicit division of responsibility and risk
- An arena for participation and conflict management
- Communication with users

The first four criteria were considered to be most critical in the beginning of the implementation process, while the remainder became more critical later in the process.

The interaction of a wastewater system with farmers and their organisations is one important aspect to consider when environmental issues relating to recycling are evaluated (Magid *et al.*, 2006). Paper IV examined how recycling was organised in seven existing Swedish cases. The study identified motives for farmers' participation and discussed critical factors for beneficial use of the sewage fertilisers. Five of the above criteria were used for structuring the analysis. The sphere of action through legislative support and communication with users were considered more relevant for other parts of the recycling chain and were thus mentioned only briefly.

When a contemporary phenomenon is examined in a real-life setting, a case study approach is suitable (Yin, 2003). If the case study includes inter-organisational relationships, information should be gathered from different perspectives. Qualitative interviews are frequently used in case studies (Kvale, 1996). A qualitative interview is semi-structured, *i.e.* an interview guide is used to ensure that the selected themes are covered. Through the interviews, diverse perspectives are captured and new and unexpected aspects can be illuminated.

In Paper IV, the cooperation in each case was examined through semi-structured interviews with both the farmers involved and official representatives assigned to coordinate the recycling scheme. The cases included four urine separating systems, two blackwater separation systems and one handling system for recycling of quality-assured sewage sludge. Below, stakeholders in the national and local arena are briefly presented and critical factors for beneficial use of source separating products as identified in Paper IV are discussed.

Stakeholders in the national arena

The use of sewage products on farmland is promoted to varying degrees by the authorities in Sweden. A goal for 60% of the phosphorus in wastewater to be recycled to productive soils by 2015 has been decided upon by the Swedish parliament, assuming a future scarcity of phosphate ores (Prop, 2005). At least half this proportion should be recycled to arable land. This goal is primarily directed towards the recycling of sewage sludge, due to the vast majority of urban citizens being connected to municipal wastewater treatment plants with a high level of phosphorus removal. However, the use of sewage sludge on farmland is a controversial issue, reflecting different values among stakeholders (Bengtsson & Tillman, 2004). On-site systems based on recycling of plant nutrients are promoted by national guidelines issued by the Swedish EPA (2006) and a national action plan for strategies on recovery of phosphorus in sewage proposed that 10% of all on-site systems should be able to recover plant nutrients by the year 2015 (Swedish EPA, 2002).

The Federation of Swedish Farmers (LRF) and the food industry, which is closely linked to LRF through co-operative ownership, are important stakeholders when recycling systems are to be implemented. Both LRF and representatives of the food industry have periodically been opposed to the use of sludge on arable land (Berglund, 2001). In 1999, LRF recommended that their member farmers stop using sewage sludge, a decision based on reports on *e.g.* brominated flame-retardants and silver found in sludge intended for agricultural use. In order to restart a dialogue among the stakeholders, a project named ReVAQ was initiated, aimed at clarifying whether the use of sewage sludge from waterborne systems was consistent with Swedish environmental quality objectives (Kärrman *et al.*, 2007). In 2006, seven wastewater treatment plants participated in the ReVAQ project, which also involved the food industry and LRF. The quality-assured process opened the way for acceptance from the agricultural and food sectors of sludge from the ReVAQ-certified treatment plants.

In 2007, there is still uncertainty among the food industry concerning the extent to which sewage fertiliser products should be approved (Giers, 2007; Kärrman *et al.*, 2007). A feasibility study on the conditions required by the food industry for accepting source separating sewage fertilisers indicated that the handling rather than the product ought to be quality-assured (Giers, 2007). For source separated products, the number of analyses could be kept low provided that the handling was well documented.

Generally, there is no legal restriction on using source separated sewage products in Sweden, except in organic production. According to the EU Council Regulation No 2092/91, fertilisers derived from human excreta are not generally allowed in organic production. However, KRAV (2007), the Swedish incorporated association for development of organic standards, permits a restricted use of human urine if certain conditions are fulfilled, *e.g.* if a close cooperation exists between the farm and the households. If the system includes households outside the farm, the collection system must be approved by the certification body.

The Swedish Water and Wastewater Association (Svenskt Vatten) represents essentially all municipal units for water and wastewater. Svenskt Vatten has long been involved in discussions on sewage sludge use on arable land and participated in the ReVAQ project. It is currently responsible for drawing up a proposal for a certification system for nutrient recovery from sewage. In 2002, a good half of the sewage sludge produced from the wastewater treatment plants was surveyed by Statistics Sweden (2004). They reported that approximately 12% of this sludge was used on agricultural land. The majority of the sludge was used for other green areas, for production of construction soils and for final covering of landfills.

Consumer and environmental NGOs are also engaged in issues regarding recycling to varying extents. The Swedish Society for Nature Conservation (SNF) participated in ReVAQ, together with the above-mentioned actors in the steering group.

Stakeholders in the local arena

Although the national stakeholders are important for developing long-term goals and general frameworks, they are rarely directly involved in existing recycling systems. In several cases, the local authority (*e.g.* in Norrköping and Tanum) initiated existing recycling schemes by requirements on urine separation for new on-site systems (Paper IV). In other cases, urine separating systems have been implemented as a result of *e.g.* private initiatives, with a municipal authority possibly being involved at a latter stage, when the installations have already been made (Degaardt, 2004; Paper IV). As illustrated in Paper IV, the responsibility for coordinating activities along the recycling chain may either be divided between several actors, including different units within the municipality, or concentrated to merely one person.

Recycling of sewage sludge is also organised in different ways. In some cases, the officials in the local authority have all contact with the farmers (Kärrman *et al.*, 2007), while in other cases this task is handed over to a contractor (Paper IV).

Policy entrepreneurs are individuals driven by belief rather than support from established institutions. They often play an important role when innovative changes occur in the wastewater sector (Söderberg, 1999). Policy entrepreneurs were important both for implementation of the source separated wastewater systems in Paper IV and for launching the quality-assured ReVAQ system for recycling of sewage sludge at municipal level (Kärrman *et al.*, 2007).

Farmers are important actors since the farm handling of sewage products affects many environmental aspects of the recycling system. Many farmers also have experience from entrepreneurship and manure handling, both of which could be valuable when organising recycling. The role of farmers in existing recycling systems is highly variable, as demonstrated in Paper IV. Some only make available their land, while others are sub-contractors involved in collection, storage and spreading, with continuous contacts with the municipal representatives.

Organising for beneficial use of sewage fertilisers

Papers I-III highlight the importance for environmental performance of a high substitution rate when sewage products replace mineral fertilisers in crop production. This aspect was also emphasised by the coordinators interviewed in Paper IV as one of the strongest motives for introducing source separating systems. However, the motives of farmers participating in these systems differed significantly. Provided economic conditions were beneficial, they emphasised the business aspect and the entrepreneurship. They considered the environmental advantage as being related to the reduced eutrophication when treatment in wastewater treatment plants was avoided. In fact, many cases included in Paper IV displayed a low substitution rate, including also some cases where the source separated sewage products did not replace any mineral fertiliser at all. The utilisation of the source separated products was high only for the two R&D cases. There were several reasons for this low substitution rate. The small and/or dilute volumes led to inefficient handling, with low incentives for substituting mineral fertilisers. Not all farmers had access to suitable equipment for spreading in a growing crop. Their awareness of the risk of soil compaction in spring when using heavy spreaders on their clay soils resulted in a strategy where the source separated sewage products were spread in the autumn, despite the fact that the need for plant nutrients at that time was negligible. Some of the farmers were also uncertain about the plant nutrient content of the sewage products they applied.

The establishment of recycling schemes for source separated products is often associated with costs for several actors and at several levels. With low political support in the municipality, additional costs relating to the recycling system were often considered a critical factor, especially if no plans for further expansion were at hand (Paper IV). In contrast, in the municipalities with a long-term goal to increase the number of source separating installations, initial costs seemed to be a minor problem.

Livestock farms can be expected to have access to both storage and spreading facilities, thus decreasing the costs of handling. However, although it might be tempting to engage a livestock farm, it might also reduce the environmental benefits of the recycling system, since many farms with access to manure already have a surplus of plant nutrients on the farm. Engaging a farm specialising in cash crop production might, on the other hand, require investment in storage facilities and hiring of spreading equipment. Although there is no easy solution for handling this situation in a real setting, it is important to highlight these sometimes conflicting aspects. Scrutinising different alternatives for storing and spreading the sewage products before the recycling scheme is launched is thus crucial. By providing the farmers with results from nutrient analyses of the sewage products, the chances of more efficient use of the plant nutrients are increased. For municipalities with an expressed aim to substitute mineral fertilisers with sewage products, the extent to which this is fulfilled should be evaluated together with other aspects of the recycling scheme, in a system for continual improvement.

The coordinators interviewed who were interested in agricultural practices and farm conditions were in general more interested in further development of the agricultural aspects of the recycling scheme (Paper IV). The farmers also expressed their appreciation with a cooperation based on mutual exchange of experiences and information. Providing arenas for participation where actors involved can exchange experiences and participate in a further development of the recycling scheme is hence strongly recommended.

The seven cases presented in Paper IV clearly demonstrate that the organisation of a recycling system can vary considerably, as regards *e.g.* the division of responsibility among the different actors. By dividing the responsibility among different actors, it is probably easier to motivate their participation. However, without a regular exchange of experiences, there might be a risk of the system becoming fragmented. In several municipalities, there are a growing number of source separating systems being installed, which might call for future reconsideration of the recycling scheme. A coordinator assigned to promote collaboration between the participating actors would thus be beneficial for future development of the recycling scheme.

General discussion

Source separating systems are often justified on the basis of their integration of wastewater management and agriculture. Different aspects of this integration were investigated in this thesis using a life cycle perspective and through a qualitative approach. By including the agricultural production within the system boundaries in Papers I-II, factors affecting *e.g.* the yield level could be evaluated. Thus both soil compaction related to the spreading and crop damage by wheel traffic in the growing crop were considered. Including crop production when evaluating nutrient recycling systems illustrates the arrangements required for nutrient handling and highlights different fertilisation strategies and how they fit into current crop production. However, this also implies that the number of aspects to be considered will increase. There is thus a need for simplification. Under conditions where the yield is affected only marginally or not at all, including crop production when evaluating different wastewater systems is not necessarily required. If the spreading strategy results in an altered yield, only the environmental impact associated with this change could be included. It is thus important to investigate whether the yield will change.

The environmental benefits associated with the source separation systems investigated here were rather modest when such systems were assumed to be applied as a complement to an existing well-functioning conventional sewerage system. However, many on-site systems in Sweden are not fulfilling the legal requirements and actions are therefore needed to improve their effluent quality. Source separating systems are thus particularly interesting alternatives for on-site systems situated in regions where the recipient waters are sensitive to eutrophying emissions. They could also be alternatives in areas where connection to a centralised sewerage system is considered too expensive or when an existing wastewater treatment plant has limited capacity for extended connection or needs to be upgraded with removal of nitrogen and/or pharmaceuticals.

The results from Papers I-IV clearly illustrate that resource aspects associated with recycling systems are far more than just a matter of recovery rate of plant nutrients. The resources required for recycling must be considered in order to avoid sub-optimisation. Although the energy required for producing mineral fertiliser products is often considered a motive for introducing recycling systems, awareness of the aspects influencing the substitution at farm level seem to be largely lacking. Indicators are often used for measuring the sustainability of wastewater systems. The results from this thesis stress the importance of including not only the potential recycling, but also the actual substitution rate of mineral fertilisers when evaluating different systems. Thus both the plant nutrient availability and the fertilisation strategy employed at farm level need to be scrutinised. In addition, the energy and other resources required for recovering plant nutrients should be considered using a life cycle approach. It could be questioned whether recycling of nitrogen should be included as a sustainability indicator, as this aspect primarily considers energy use. A better indicator would be to compare the potential energy savings when substituting nitrogen mineral fertilisers with the energy required for this recycling. The recycling rate of nitrogen is therefore best evaluated embedded in an overall analysis of the energy flows. Recycling should thus be considered one possible instrument for increasing sustainability, not as a goal in itself. The goal should rather be an efficient use of resources.

There is an array of technical options available for recovering nutrients from sewage. The choice of technical structure is one important aspect, since both the construction and operation of systems are associated with different environmental impacts. However, the performance of the same technical system can also be highly variable, depending on how the system is organised. Paper IV shows that there seem to be few incentives for optimised performance all through the recycling chain. The use of urine and blackwater at farm level is but one factor illustrating that the actual handling in many existing systems is far from optimal (Fernholm, 1999; Paper IV). There might thus be a discrepancy between a hypothetical system evaluated in a LCA and the real performance. On the other hand, there is also room for technical improvement in source separating systems, since they are still only in their infancy. Blackwater systems are interesting alternatives if both nitrogen and phosphorus are to be reduced to a high degree. As illustrated by Papers II and III, the extent to which the blackwater is diluted with flushwater affects the environmental performance of the system, since all those volumes require collection, transport, treatment, storage and spreading. The cost of these activities is also increased with increased volumes. Another challenge for blackwater systems is to provide a reliable, cheap and robust method for sanitising the material. Ammonia-based sanitation is a promising option (Paper III), but needs to be tested and evaluated on a large scale. There is thus room for improvement as regards blackwater systems. Urine separation with dry handling of the faeces provides an alternative for energy-efficient handling of the household excreta, at the same time as most of the nutrients in the household wastewater could be recycled (Jönsson *et al.*, 2005). However, further development is needed before those systems would be broadly accepted by users.

Hoffmann *et al.* (2000) criticised the idea of considering some specified techniques to be sustainable in themselves, since this approach overlooks the importance of the local context. Farm conditions are one such local factor considered in this thesis. The storage and spreading facilities available, plant nutrient situation on the farm, requirements from purchaser and interest of the farmer in participating in a recycling scheme are some factors influencing the performance of a recycling system. According to Ljung (2001), there are several arguments for broader participation by farmers in environmental management systems. One argument is that farmer participation emphasises local knowledge. Farmers' experiences of systems for plant nutrient management could provide valuable insights when developing management strategies for recycling of sewage products. Thus, the potential to increase their participation is interesting in many existing systems.

According to a mail survey of 127 respondent Swiss farmers, 42% were willing to purchase a urine-based fertiliser product (Lienert *et al.*, 2003). However, the price would have to be moderate, since 62% would pay at most 80% of the price of the fertilisers currently used. In Paper IV, most farmers found the monetary value of the source separating products limited, while the entrepreneurship related to the handling was highlighted as an interesting business aspect. This has also been stressed by Sjöberg (2003). There is hence an interest from farmers not primarily in purchasing a sewage fertiliser, but in being involved as contractors in different parts of the recycling chain. Czemiel Berndtsson & Hyvönen (2002) emphasised the need for a future market for nutrients found in human excreta. However, the costs associated with urine spreading and associated soil compaction might be of the same magnitude as the financial value of the fertiliser (Degaardt, 2004). In addition, there are also costs related to the collection, transport and occasionally also the storage at farm level. It seems therefore unrealistic that the financial value of the source separated fertiliser would act as a driver for an increased implementation in the foreseeable future.

An interesting approach would be to use source separated sewage fertilisers in organic farming. Low-cost sewage products would be particularly interesting as a fertiliser on organic farms specialising in cash crop production, since those farms need to be supplied with external plant nutrients. With often low access to cheap fertiliser products approved for use on organic farms, there would probably be a stronger motive for them to economise with the plant nutrients found in the sewage. Theoretically, one could therefore expect a higher willingness in organic farming compared with conventional farming to spread, store and eventually also transport sewage-based fertilisers in exchange for the plant nutrients. Current EU regulations on organic farming do not generally permit the use of human excreta on organic farms, but the exceptions currently being made by KRAV (2007) might perhaps open the way for new forms of collaboration between organic farmers and *e.g.* eco-villages.

Although there are many options for organising recycling of source separated products to farmland, there are also examples where the entire recycling system has been jeopardised due to difficulties in finding alternatives to agricultural reuse (Degaardt, 2004; Regionplane- och trafikkontoret, 2006). With restrictions in the use in organic farming and with an occasionally low interest among conventional farmers, there is thus a need to devise alternative strategies for recycling the plant nutrients in source separated sewage fertilisers, *e.g.* by using urine in domestic gardens or in public green areas, and by using blackwater in biomass production for energy purposes.

Conclusions

- Evaluating nutrient recycling systems from a life cycle perspective is important for a comprehensive assessment of their benefits and drawbacks.
- By including crop production when assessing nutrient recycling systems, factors affecting the yield, *e.g.* soil compaction, wheel traffic and fertilisation strategy, were included, and agricultural arrangements required for recycling were visualised. This approach also highlighted potential conflicts as regards nutrient utilisation at farm level.
- If a change-oriented approach to LCA is applied to agricultural production systems, it seems appropriate to include an alternative use for the released area. The use of this area is decided by time and site-specific conditions.
- Source separation of urine or blackwater with subsequent use in crop production proved most beneficial in situations where the eutrophying emissions are critical. For Swedish conditions, both those separating strategies could be of interest for upgrading on-site systems. Urine separation is an energy-efficient alternative for reducing nitrogen, while blackwater separation proved interesting if both nitrogen and phosphorus are to be significantly reduced. Ammonia-based sanitation is a promising option for sewage products from on-site systems.
- Whether a source separating system applied in an urban setting is more energy-efficient than a conventional system depends on several factors, *e.g.* the construction of infrastructure for separation and collection, the sanitisation method used and the substitution rate of mineral fertilisers. Therefore, those factors should be optimised and scrutinised when planning, implementing and evaluating source separating systems aiming at efficient use of resources.
- Resource aspects relating to recycling are far more than just a matter of recovery rate of plant nutrients, since different activities required for the recycling might be associated with considerable use of energy and other resources. Strategies and methods for weighting use of different abiotic resources are therefore needed.
- Systems relying on the same separation techniques might vary considerably as regards design of different components and actual performance. Therefore, a technical system in itself might not be inherently sustainable. There is thus a need to assess both the technical system in detail and organisational aspects and drivers affecting the performance of the recycling system.
- The fertilisation strategy used could have a larger impact on the substitution rate, especially of nitrogen, than the estimation of the plant-

availability in the original sewage product. When evaluating different systems aiming at recycling, different possible fertilisation strategies and their impact on the substitution rate should therefore be examined. It is also important to provide information on the plant available nutrient content of the product to the farmer.

- There seem to be few incentives for optimised performance all through the recycling chain. There is thus room for improvement at different levels. A future task for local authorities and farmers involved in existing recycling schemes is therefore to devise strategies for improved long-term utilisation of the nutrients in sewage products. Providing arenas for enhanced collaboration is proposed for a further development of recycling systems.

Recommendations for future work

This research emphasised different aspects affecting the environmental performance of systems integrating farming and wastewater management in a Swedish context. Some suggestions for future studies are summarised below:

- This work focused on sewage products suitable for application with spreaders. For more dilute sewage products, irrigation on *e.g.* energy crops is a more feasible spreading strategy. An interesting task would be to evaluate the environmental performance of irrigation of sewage products with different origins, *e.g.* effluents from wastewater treatment plants or on-site systems in a life cycle perspective.
- There is no consensus on how to handle differences in land requirement in LCA. There is thus a need for further development and evaluation of different approaches and how they relate to different types of LCA performed.
- Recycling systems are often motivated by conservation of abiotic resources. However, there is a need for further development of methods comparing and weighting the use of different resources.
- The limited feasibility of conventional farmers using mineral fertiliser products means that the incentives to use source separated products as fertiliser are low. The access to easily available plant nutrient sources is more restricted in organic farming. Evaluating future strategies for utilisation of sewage products in organic farming would emphasise environmental and resource aspects related to this handling.
- Many poor regions suffer from plant nutrient depletion. An important task would be to further scrutinise and evaluate the options for a large-scale introduction of source separating systems and the use of sewage fertilisers in agriculture in those regions.

- From the farmers' perspective, the entrepreneurship arising from the handling of source separated sewage products might be an interesting business activity. It is necessary to develop different strategies for farmer participation and to scrutinise the economic incentives for such systems.
- This work highlighted several critical factors for a beneficial use of sewage products as fertilisers. A further step would be to use this information as decision support when future recycling systems are being planned, and to develop strategies for improved management at farm level.

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