Life Cycle Assessment of Grain Production Using Source-Separated Human Urine and Mineral Fertiliser

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ABSTRACT

Source-separation of human urine is one promising technique for closing the nutrient cycle, reducing nutrient discharge and increasing energy efficiency. Separated urine can be used as a valuable fertiliser in agriculture, replacing mineral fertiliser. However, a proper handling of the urine at farm level is crucial for the environmental performance of the whole system. This study started from an agricultural point of view, demonstrating how grain production systems using human urine might be designed. The main objective was to evaluate the consequences on environmental impact and resource management when human urine replaced mineral fertiliser in arable farming. Production of winter wheat and spring barley when only mineral fertilisers were used was compared to a scenario where a combination of human urine and mineral fertilisers was used. The method for assessing the two different scenarios was Life Cycle Assessment (LCA), and the functional unit was 1 kg of grain.

In the systems analysis, a change-orientated perspective was used whereby all major changes in the agricultural system using urine (the urine-separating scenario) were taken into account, compared to the conventional scenario. When urine is separated from the remaining wastewater, the production of drinking water as well as the wastewater handling is affected. These changes were taken into account through subtraction of the burdens avoided when separating urine. Production of capital goods, e.g. storage tanks, was also included in the urine-separating scenario in those cases where differences between the scenarios appeared.

The results obtained were quite similar as regards the two grain production systems. Differences appeared instead when comparing the conventional scenario to the urine-separating scenario. For both scenarios, most of the energy required was fossil fuel. The use of fossil fuel was slightly higher in the scenario using human urine as fertiliser, but electricity consumption was higher in the conventional scenario. Whether a urine-separating scenario will decrease the energy usage depends on many factors, and is not self-evident. The construction phase might make a considerable contribution and the sense in which the existing water and wastewater system is affected will also be important. With the assumptions made in this study, the urine can be transported more than 40 km one way without exceeding the total energy used in the conventional scenario. However, minimising transports is just one of several key issues from an energy point of view.

The contribution of greenhouse gases, expressed as GWP, from the two scenarios was of the same magnitude, although slightly less from the urine-separating scenario. For both scenarios, nitrous oxide originating from soil emissions gave the highest contribution. The difference in contribution to eutrophication was considerable between the two scenarios, due to the avoided emissions of eutrophying substances in the urine-separating scenario. Which scenario contributed most to acidification depended on in what sense nitrogen compounds contribute to acidification.

A considerable part of the phosphorus required as mineral fertiliser can be replaced by phosphorus in human urine. When half of the nitrogen required in winter wheat was applied as human urine, approximately 40% of the phosphorus required came from the urine.

Guaranteed quality is of major importance when discussing the use of human urine on arable land. The composition as regards heavy metals, organic pollutants, pathogens and plant nutrients must therefore be guaranteed. The level of heavy metals in human urine is very low. The contribution of e.g. cadmium is even lower than in some “cadmium-free fertilisers”. The hygienic risks can be almost eliminated with adequate storage. However, the risks related to pharmaceuticals in urine must be further investigated.

SAMMANFATTNING

Källsortering av humanurin är en lovande teknik för att sluta kretsloppet av växtnäring, reducera utsläpp av närsalter och öka energieffektiviteten. Den separerande urinen kan användas som ett värdefullt gödselmedel inom jordbruket och därvid ersätta mineralgödsel. För att kunna dra de miljömässiga fördelarna av urinsortering krävs dock att hanteringen av urinen på gårdsnivå utförs på rätt sätt. Denna studie tar sin utgångspunkt i jordbruket och visar hur odlingssystem där man använder humanurin kan utformas. Rapportens huvudsyfte var att utvärdera konsekvenserna på miljö och användning av naturresurser när humanurin ersätter mineralgödsel i odlingen. Konventionell produktion av höstvete och vårkorn jämfördes med ett scenario där mineralgödsel delvis ersattes av humanurin. Den metod som användes till att utvärdera scenarierna var livscykelanalys (LCA) och den funktionella enheten var 1 kg spannmål producerad på gården.

I systemanalysen användes ett ”förändringsorienterat” perspektiv. Härigenom beaktades alla större förändringar i det odlingssystem där urin användes (det urinsorterande scenariot) jämfört med det konventionella scenariot. När urin utsorteras från det resterande avloppsvattnet kommer detta att påverka produktionen av dricksvatten liksom behandlingen av avloppsvatten. Dessa förändringar togs hänsyn till genom subtraktion av de sluppna emissionerna och övriga miljöbördor som erhölls när urinen sorterades ut. Även produktionen av kapitalvaror, t ex lagringstankar, inkluderades i det urinsorterande scenariot i de fall dessa inte var identiska mellan de båda scenarierna.

Resultaten var relativt lika oavsett om produktionen var höstvete eller vårkorn. Skillnader uppstod istället när man jämförde det konventionella scenariot med det urinsorterande. För båda gällde att energiförbrukningen främst utgjordes av fossila bränslen. Användningen av fossila bränslen var något högre i det urinsorterande scenariot, medan däremot elförbrukningen var högre i det konventionella scenariot. Huruvida ett urinsorterande system kommer att öka energieffektiviteten är inte helt självklart och beror på många faktorer. Produktionen av kapitalvaror är betydelsefull, likaså påverkan på det existerande VA-systemet. Med de antaganden som gjordes i denna rapport, kunde urinen transporteras 40 km enkel väg utan att den totala primära energianvändningen i det konventionella systemet överskreds.

Bidraget till växthuseffekten var av samma storleksordning i de båda scenarierna, om än något lägre från det urinsorterande. I båda scenarierna gav lustgasemissioner från mark de största bidragen. Skillnaden i eutrofiering var betydande mellan de båda scenarierna eftersom utsläppen av övergödande ämnen minskas vid urinsortering. Vilket scenario som bidrar mest till försurning styrs av i vilken utsträckning kvävenedfallet verkar försurande.

En avsevärd del av grödans fosforbehov kan täckas av urinens fosforinnehåll. När höstvetets halva kvävebehovet tillfördes i form av urin, täcktes även fosforbehovet till drygt 40% av urinen.

Kvalitetssäkring är av största vikt när man diskuterar jordbruksanvändning av humanurin. Kvalitén med avseende på tungmetaller, organiska föroreningar, patogener och växtnäringsinnehåll måste därför garanteras. Tungmetallinnehållet är generellt lågt i humanurin. Till exempel är innehållet av kadmium i humanurin till och med lägre än i vissa så kallade kadmiumfria mineralgödselmedel. Hygieniska risker kan i det närmaste uteslutas vid en tillräcklig lagring. Påverkan på människors hälsa och miljön av urinens innehåll av läkemedelsrester är dock viktig att utreda!

FOREWORD

After more than a century of development of the Swedish urban water systems, it could be said to fulfil many requirements concerning safety, hygiene and protection of the environment. However, the systems have been questioned for not being sustainable. The research programme "Sustainable Urban Water Management", abbreviated as Urban Water, has the following main goals for sustainable water management.

- Towards a non-toxic environment
- Improved health and hygiene
- Saving human resources
- Conserving natural resources
- Saving financial resources
- Increasing Sweden's competitiveness

In the Urban Water programme, both improvements of existing systems as well as introduction of alternative wastewater systems are considered and assessed. My doctoral project is part of the Urban Water programme, focusing on the agricultural use of different sewage products and financed by Urban Water and PROWARR (the Research Programme Organic Waste – Resource or Risk in Sustainable Agriculture). This study considers the use of source-separated human urine as fertiliser in grain production. It is my hope that this study will contribute to some answers about future sustainable wastewater systems in Sweden. And maybe pose some new questions that need to be considered.

I am especially grateful to my supervisor Håkan Jönsson for his involvements and comments all through the work, and to my co-supervisor Berit Mattsson for comments on an earlier version of the report. I also want to thank Janne Linder for initiated information about agricultural practice in the region studied and Erik Kärrman for comments on part of the manuscript.

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BACKGROUND

Human excreta have traditionally been used in agriculture in many countries. Using unprocessed latrine was for example common on the Swedish countryside, whereas latrine in the cities was further processed to “pudrett”, i.e. powdered latrine mixed with slaked lime or peat (Wetterberg & Axelsson, 1995). With the introduction of the waterborne sewage system, the possibilities to use the plant nutrients in the sewage decreased considerable. In most places, the sewage with its content of plant nutrients was discharged into nearest watercourse and many rivers floating through the cities became gigantic sewer ditches. In some European cities, e.g. London, Paris and Berlin, sewage water was partly used for irrigation of arable land (Mårald, 1999). Through irrigation, the water was purified at the same time as the plants were fertilised. This practice has ceased essentially as the irrigation systems became too expensive and failed in fulfilling the sanitary requirements. According to Liebig’s mineral theory, enunciated in the 1840’s, minerals are crucial for plant growth. When the minerals were flushed into watercourses, the soil from where they originated, was slowly impoverished. Instead of recycling plant nutrient in sewage, fertilisers in the form of guano and bones were used on farmland (Mårald, 1999). The farming system became dependent on exploitation of finite resources.

Today, the idea of recycling of nutrients from urban areas to arable land is seen as an imperative in a future sustainable society and recycling of phosphorus is a prioritised political goal in Sweden (SOU 2000:52). However, the recycling of sewage products to arable land does have many implications and is therefore a controversial and lively discussed issue. Some debaters consider the agricultural use of sludge as only one way to dispose an undesirable waste product. They fear that sludge used as fertiliser in the long run will give negative effects on human health, soil quality and future food production. Others look upon sludge as a most valuable fertiliser, which should be utilised in order to decrease the risk of a future lack of phosphorus ore worth mining. At present, only little sewage sludge is spread on arable land in Sweden, depending on deteriorated confidence in sludge from the agricultural organisations and the food industries. Several large Swedish food companies, including Arla and Cerealia, have long time denied buying products grown on fields fertilised with sludge. In November 1999, the Federation of Swedish Farmers, LRF, recommended all their members to stop using sewage sludge, due to new alarming reports about brominated flame retardants and silver in the sludge. Since this latest boycott of sewage sludge begun, hardly any food companies seem willing to accept sewage sludge in the near future. Regarding the use of other sewage fertiliser products than sludge, most of the food industries still have not worked out any policies (Berglund, 2001).

New alternatives for wastewater handling, e.g. source-separating systems, seem to a larger extent to fulfil the requirements from agriculture of a fertiliser product with a high nutrient value, but without the many of the hazardous compounds found in sewage sludge. Urine separation is one promising technique for closing the nutrient cycle, reducing the nutrient discharge and increasing the energy efficiency. Several systems analyses of wastewater systems point out urine separation as a more favourable alternative in most environmental aspects than a conventional system (e.g. Bengtsson et al., 1997; Kärrman et al., 1999; Jönsson et al., 2000). Also an exergy analysis comparing human urine with commercial fertilisers came to the result that utilising human urine might increase the energy efficiency through a lower net exergy consumption (Hedström et al., 2000). The distance required for transporting the urine to the field might however have a great impact.

With new alternatives for wastewater handling in progress, emphasis on the requirements from agriculture can be put more in focus. A future demand from agriculture of sewage products might have a major impact on the design of the wastewater system and the sustainability of farming. This study starts from an agricultural point of view aiming at answering the questions: What are the environmental consequences when human urine replaces mineral fertiliser in arable farming? How will the use of human urine influence the sustainability of farming?

SCOPE OF THE REPORT

This report consists of two parts, a literature review and an LCA-study of grain production using human urine.

The literature review starts with an overview over different aspects on systems for source-separation of human urine. The main focus is however on methodological aspects on life cycle assessments and especially on the implementation of LCA to agricultural land use. A state-of-the-art is given, and possibilities and shortcomings with the implementation of LCA in agriculture are presented and discussed.

After the literature review follows the main part of this report - an LCA-study on grain production where conventional production according to current practice in eastern part of Sweden (Mälardalen) is compared to a scenario where source-separated human urine partly replaces mineral fertiliser. Winter wheat and spring barley are the two crops under study.

SOURCE-SEPARATED HUMAN URINE – A LITERATURE REVIEW

Human urine is the largest contributor of nutrients to the household wastewater. In Sweden approximately 36 000 tons of nitrogen is found in the urine fraction (calculated from Naturvårdsverket, 1995). The sales of nitrogen as mineral fertiliser for agricultural and horticultural purposes are approximately 200 000 tons in Sweden yearly (Statistics Sweden, 2002a). Human urine has been proposed as a valuable fertiliser in future farming systems, especially in ecological farming, due to its content of easily available nitrogen. Ecological farming in Sweden was initially allowed to use human urine according to the regulation of KRAV (Hansson, 1995). Since the Swedish entrance into the European Union, use of human urine in ecological agriculture is not allowed any longer, according to current regulation (EEG 2092/91).

Design of the urine separating system

Source-separation of urine is usually based on a toilet equipped with two bowls, one for collection of urine and one for faecal material. The two bowls have separate flushing mechanisms, with a minor amount of flushing water used in the urine bowl. The bowl for urine is connected through pipes to a collection tank. The tank is often buried in the ground and therefore normally keeps a low temperature between 0°C and 10°C. After storage, which should be sufficiently long for reducing potential pathogens, the urine is spread on arable land.

Plant nutrients in source-separating urine

Most of the macronutrients present in the wastewater from the households are found in the urine fraction as illustrated in Table 1. This is especially valid for nitrogen, as 80% of the total amount of the nitrogen present in the household wastewater occurs in the urine. The figures in Table 1 have been proposed as new Swedish design values, based on both earlier norms from the Swedish EPA (Naturvårdsverket, 1995) and a revision based on thorough investigations of the different wastewater flows (Vinnerås et al.).

Table 1. Average figures on plant nutrients expected in household wastewater fractions per person and year (Vinnerås et al.).)

Parameter	Urine	Faeces	Grey water
Mass [kg]	550	51	36 500
N [g]	4 000	550	500
P [g]	365	183	190
K [g]	1 000	365	365

The nutrient concentration in the collected urine will depend on different factors and may therefore show large variations, as demonstrated by the mean values in Table 2. Within each mean value reported exists further large variation, which is not shown here. The amount of flush water used will dilute the urine, thus increasing the volume and cost for storage, transportation and spreading. Food customs and the users' habits and willingness to handle the toilet in a correct way will have a further impact on the composition and quality of the urine mixture.

Table 2. Concentration of plant nutrients (g/l) as mean values in urine mixture (urine+flush water) as reported by different authors

References	Tot-N	NH_4-N +NH_3-N	P	K	S
Carlsson (1995)	2.6	1.7	0.19	0.45	
Carlsson (1995)	1.8	1.6	0.11	0.36	
Jönsson et al. (1998)	3.5	3.4	0.31	0.94	0.33
Kvarmo (1998)	3.7	3.3	0.27	1.22	0.33
Lundström & Lindén (2001)	2.5	2.1	0.25	0.70	
Olsson (1995)	2.4	2.2	0.24	0.65	0.2
Pettersson (1994)	2.2	2.1	0.21	1	0.2
Vinnerås (1998)	2.3	2.1	0.14	0.48	0.17

Nitrogen efficiency

Several field experiments with human urine as fertiliser have been carried out in Sweden. In nine experiments with winter wheat and oats on organic farms, the effect on grain yields, crude protein in the kernel, nitrogen efficiency and risk of nitrogen leaching were investigated (Lindén, 1997). No treatments with mineral fertiliser were included in the experimental plan, therefore no correct comparison with the yield-increasing effect from commercial mineral fertilisers can be made from these experiments. Comparisons with other trial series with winter wheat indicate however that the total N in human urine had about 60-80% of the yield-increasing capacity compared to mineral fertilisers. For oats, the corresponding figures were 50-60%. No increasing nitrogen leaching risk seemed to occur due to fertilisation with human urine instead of mineral fertilisers.

In field trials performed by JTI (Swedish Institute of Agricultural and Environmental Engineering) during three years, yield, nitrogen efficiency, ammonia volatilisation and risk for nitrogen leaching were examined (Richert Stintzing et al., 2001). The yield when using human urine was 70-115% of the yield from plots fertilised with the same amount of nitrogen in the form of mineral fertilisers. Nitrogen from organic matter in human urine was also included in the comparison. Nitrogen efficiency during two years, expressed as nitrogen uptake compared to the total amount added, was 44 and 70% when urine was used. Use of mineral fertiliser resulted in an efficiency of 61 and 83% respectively. Ammonia volatilisation never exceeded 10% and was in average 5% when urine was spread in springtime. Spreading of human urine in a growing crop resulted in no detectable volatilisation. However, only one year of experiments of spreading in a growing crop was performed. No increasing risk for nitrogen leaching seemed to occur due to the use of human urine instead of mineral fertilisers.

In a ^{15}N-labelled pot experiment with spring barley, fertilisation with human urine resulted in 7% lower nitrogen uptake compared to ammonium nitrate, mostly due to ammonia volatilisation (Kirchmann & Pettersson, 1996). The crop uptake efficiency for *urine-N* was 42% compared with 53% for ammonium nitrate. The efficiency of phosphorus in urine was found to be some what higher than that of soluble phosphorus in mineral fertiliser.

In another pot experiment, the nitrogen efficiency, according to the difference method, did not show any differences between fertilisation with urine and mineral fertilisers (Kvarmo, 1998). According to Kvarmo, no toxic effects are likely to occur when normal levels of human urine are used. The seed germination may be delayed approximately one day, but no inhibition is to be expected (Kvarmo, 1998).

Heavy metals

The urine fraction stands for a minor proportion of the heavy metals found in the household wastewater (Vinnerås, 2001). Many of the heavy metals analysed in earlier studies, have been below the detection limit, while other metals may occur in detectable amounts (Table 3). If for instance copper pipes are used within the urine pipe system, concentration of copper may be significant higher than otherwise.

Table 3. Concentrations of heavy metals in urine solution (mg/kg) from four source-separating systems

Parameter	Understenshöjden [a)]	Palsternackan [a)]	Hushagen [b)]	Ekoporten [c)]
Hg	0.00044	<0.0004	<0.001	0.00043
Cd	<0.001	<0.0013	<0.001	0.00058
Pb	<0.01	<0.027	<0.02	0.019
Cr	0.019	0.02	<0.006	0.013
Co	<0.005	<0.0025	<0.003	
Ni	0.061	<0.022	<0.010	0.040
Mn	0.037	<0.0045	<0.005	
Cu	2.5	3.00	0.25	1.82
Zn	0.2	0.52	0.16	0.18
Mo	0.036	0.02	0.01	
Fe	0.39	0.40	0.05	
B	0.61	0.53	0.24	

[a)] Jönsson et al., 1998
[b)] Vinnerås, 1998
[c)] Vinnerås, 2001

The concentration of cadmium in urine mixture is often below the detection limit (Table 3). The ratio between cadmium and phosphorus in the different studies was lower than 2 to 7 mg cadmium per kg phosphorus, depending on the detection limit. This is a low level of contamination, and in comparison with “cadmium-free” phosphorus fertiliser products (Växtpressen, 1998). The “cadmium-guarantee”, set up by the manufacturer Hydro Agri, guarantees less than 5 mg cadmium per kg phosphorus (Hydro Agri, www).

Medical residues

The occurrence of medical residues in wastewater flows has been identified as an important issue when discussing the sustainability of wastewater systems. Absorbed substances are mainly excreted via the urine, and substances not absorbed are excreted with the faeces. So far, the knowledge of the environmental behaviour of those substances in nature is limited and more studies are called for (Naturvårdsverket, 1996). Antibiotics are of special interest. Besides the risk of microbial resistance, they might have an impact on a range of organisms, both in aquatic and terrestrial environment. When organic fertilisers containing antibiotics and antibiotic residues are applied to the soil, the medical substances might have an impact on soil fertility and productivity through a change in the composition of soil microbes. The use of antibiotics for humans was about 80 tons during 1994 in Sweden, mainly (70%) consisting of different penicillin. Between 1994 and 1997, the consumption decreased with 22%, a decrease that might be explained by an increasing awareness of the problems related to microbial resistance (Socialstyrelsen, 2000). Additionally, antibiotics are used as therapeutics in livestock production. Since the prohibition of antibiotics as growth promoters 1986, the use has decreased and during 1997 the total amount of antibiotics used in Swedish livestock production was 20 tons of active substances (Odensvik & Greko, 1998). The consumption has continued to decrease and was 17 tons the year 2000 (SVA, 2001). The most frequently used antibiotics are of natural origin, and they could therefore be expected to be degraded in nature (Naturvårdsverket, 1996).

Hormones are biological active in low concentrations. In wastewater, the concentration of hormones excreted normally by humans may be 100 times higher than hormones originating from drugs (Naturvårdsverket, 1996). The amount of hormones excreted by animals and subsequently found in manure, is considered as much higher than hormones excreted by humans, and the additional risk when using e.g. human urine as fertiliser should therefore not be of major importance. In a Danish study on urine-separating systems, the content of oestrogen was constantly on a level of 4 mikro-gram per liter (Kolby & la Cour Jansen, 2001). No distinction was made between the amount excreted naturally by humans, and hormones from contraceptives. The amount of hormones was constant during the storage, indicating that no degradation occurred in the storage tank. Paracetamol, an analgesic for home medication, was detectable in all samples. Facts about its degradation and potential impact on nature have not been found.

However, when assessing the risk for effects on health and environment from medical residues in source-separating flows, a comparison with the conventional handling of wastewater in a wastewater treatment plant must be included. Therefore, an interesting question is whether an additional risk is introduced with the handling of e.g. source-separated urine. No chemical risk assessments have been found in the area, but it is likely to assume that the soil will act as an extra filter, probably providing less risk for contamination of the water than in the conventional system. More studies about the risks of plant uptake and impact on soil microbes when sewage products are applied to arable land are however deeply needed.

Hygienic aspects

The hygienic risks related to handling of source-separated urine is mainly dependent on the faecal cross-contamination as a result of misplaced faeces. Storage time as well as temperature will then affect the microbial reduction. Experimental survival studies performed indicate that gram-negative bacteria, such as *Salmonella* and *E.coli*, are rapidly inactivated, whereas gram-positive faecal streptococci are more resistant (Höglund, 2001). The same author reports that bacteriophages and rotavirus are not inactivated at the low temperature of 5°C, while oocysts of Cryptosporidium (causing diarrhoeal diseases) might be less persistent. Spores from clostridia (used as an indicator organism) were not reduced at all during 80 days, neither at 20°C nor at 4°C.

Results from a Danish study reveal that common bacterial pathogens are reduced below the detection limit within 20 days (Dalsgaard & Tarnow, 2001). In contrast to earlier studies, Cryptosporidium oocysts were not found to be inactivated within five months.

Guidelines for use of human urine as fertiliser

Minimising the risk for transmission of infectious diseases is of vital importance when implementing a system for recycling of human urine. Experimental studies and measurements on existing systems reported above as well as a hygienic risk assessment performed (Höglund, 2001) conclude that recycling of urine to arable land is associated with a low risk for gastro-intestinal infections.

In Sweden more detailed guidelines or recommendations for the use of human urine as fertiliser have been suggested (Jönsson et al., 2000). The relationship between storage, possible remaining pathogens and recommended use on crops are shown in Table 4.

Table 4. Guidelines for the safe reuse of human urine

Storage temperature	Storage time	Possible pathogens occurring	Recommended crops
4°C	≥1 month	Viruses, protozoa	food and fodder crops that are to be processed
4°C	≥6 months	Viruses	food crops that are to be processed, fodder crops
20°C	≥1 month	Viruses	food crops that are to be processed, fodder crops
20°C	≥6 months	probably none	all crops

Based on Danish experimental studies on microbial reduction, four months of storage is recommended in a report from Danish EPA (Dalsgaard & Tarnow, 2001).

LCA METHODOLOGY

Reducing emissions to air and water from point sources has been a predominant environmental strategy in many industrial countries. But despite an increasing awareness and actions taken, environmental problems are considered to increase in magnitude and complexity (Lindfors et al., 1995). For a better understanding of complex environmental problems, a systems analysis approach is fruitful. Life cycle assessment (LCA) is one method used in different areas for analysing complex systems in an organised way. LCA aims at evaluating the environmental burdens associated with a product system or activity by identifying and describing the energy and materials used, as well as the emissions and wastes released to the environment, and to assess the impacts of those on the environment (Lindfors et al., 1995). LCA normally takes into account all activities related to a certain product or service, i.e. a cradle-to-grave perspective.

Since LCA still is under development, intensive efforts for harmonising LCA methodology on an international level, is going on. Standardising is made through the framework of ISO (International Standard Organisation). SETAC (Society of Environmental Toxicology and Chemistry) has an LCA Advisory Group, which provides a forum for identification and communication of issues regarding LCAs, and co-ordinates and provides guidance for the development and implementation of LCAs (SETAC, www). Several detailed guidelines on how to perform an LCA have also been worked out, e.g. "Nordic Guidelines on Life-Cycle Assessment" from 1995, and more recently, a comprehensive report named "Life cycle assessment. An operational guide to the ISO standards" (Guinée, 2001).

LCA is intended as a tool for decision-support for authorities, companies and consumers and is often used for comparative studies aiming at comparing and evaluating different alternatives. LCA has however been used more as a tool for learning, than for making decisions (Baumann, 1998).

There are obvious limitations with LCAs, as well as with other systems analysis methods. As an LCA is a simplified model of a complex system, it cannot provide a complete picture of every environmental interaction. Most LCAs are site-specific case studies. Therefore, generalisations from such results, without considering the underlying assumptions, could be misleading. To avoid an incorrect application, it is necessary to state the assumptions and to give a detailed description of the system, data sources etc.

Different types of LCA

Life cycle assessment may be divided into two categories; retrospective (accounting) LCA and prospective (change-oriented) LCA (Tillman, 2000). An accounting LCA deals with the question of what environmental impact a product or service can be responsible for. A change-oriented LCA compares environmental consequences of different alternatives, modelling a change (Baumann & Tillman, 2000). The purpose, and thus the type of LCA used, will affect system boundaries, allocation procedures as well as choice of data. If a complete system is analysed without effects of any choice, average data might be used, and if the purpose is to model any change, marginal data might be used (Frischknecht, 1997). The choice whether to use marginal or average data on electricity data can have a substantial impact on the results, as the difference between marginal and average electricity production in the Nordic countries is large (Ekvall, 1999).

The structure of LCA

An LCA includes different phases; goal and scope definition, inventory analysis and impact assessment (Figure 1). The interpreted results may then be input in a decision-making process.

One proposed way of performing an LCA-study, is to start with an initial screening study, where key issues or hot spots shall be identified for further and more detailed investigations (Lindfors et al., 1995). Hot spots are parts of the life cycle, which are responsible for substantial parts of the environmental impacts, or where major differences between the alternatives will appear. Performing an LCA is often an iterative process. After a sensitivity analysis, additional data or redefinition of the goal and scope may be required (Lindfors et al., 1995).

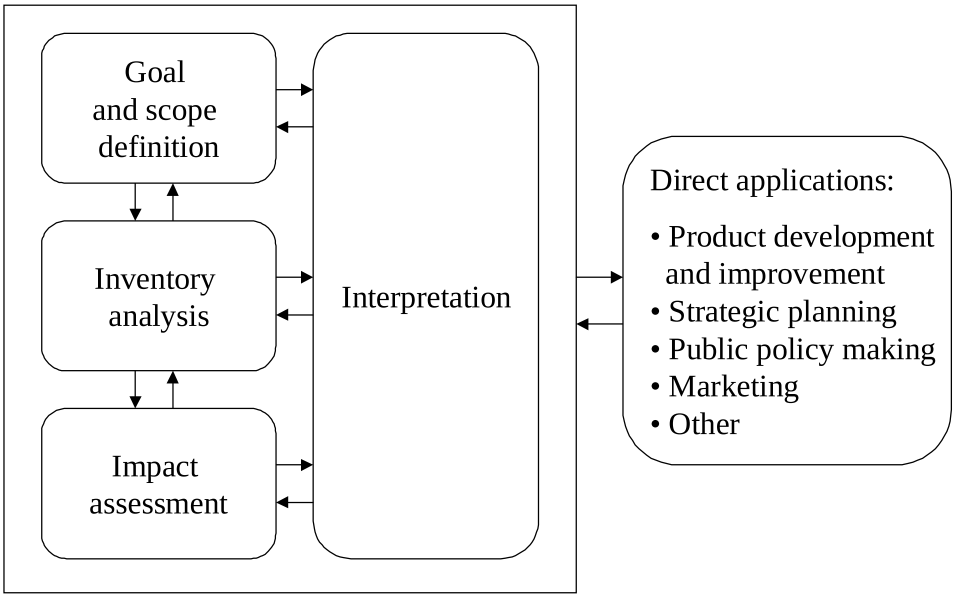

Figure 1. Phases of an LCA (ISO 14040)

Goal and scope definition

As a first step in an LCA, the purpose and scope of the LCA-study is stated. According to the ISO-standard (ISO 14040, 1997) the intended application of the study as well as to whom the results are intended should be defined. This phase 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).

Functional unit

An important concept in the LCA methodology is the *functional unit*, a clearly defined measure, based on the main function of the system or what the system delivers. As all data will be related to the functional unit, it is of crucial importance that the compared systems fulfil the same function. If the purpose is to compare different systems for the production of a product, e.g. wheat with a certain content of protein, the functional unit in an agricultural LCA can be one kg of the wheat produced. But if the main purpose is to compare different uses of arable land, one hectare could sometimes be a more appropriate functional unit (Audsley et al., 1997).

System boundaries

The system boundaries differentiate the analysed system from its environment. If one system fulfils more functions than another, i.e. more than the main function of interest, expanding the system can improve the comparability of the systems. This may avoid an allocation problem and will give a more complete model of the system. The main disadvantage is that the systems may be large and complicated (Lindfors et al., 1995). Expanding the boundaries can be done either through adding subsystems, that will provide missing functions, or subtracting hypothetical subsystems with the excessive functions. According to Nordic Guidelines (Lindfors et al., 1995), adding subsystems is preferable, as it will provide a higher transparency. From subtracted systems, environmental impacts may be negative, due to "avoided emissions".

How to handle subtraction can be exemplified by a study of the use of Thomas meal in agriculture. Here, the "Avoided burdens approach" was used, which means that the displaced burdens in a background system were taken into account (Figure 2). The burdens considered before usage in the system under study, were those arising from transportation and processing specifically for use in the system, minus burdens that will no longer appear in the background or extended system (Audsley et al., 1997). Burdens in the background system could be e.g. emissions from landfills.

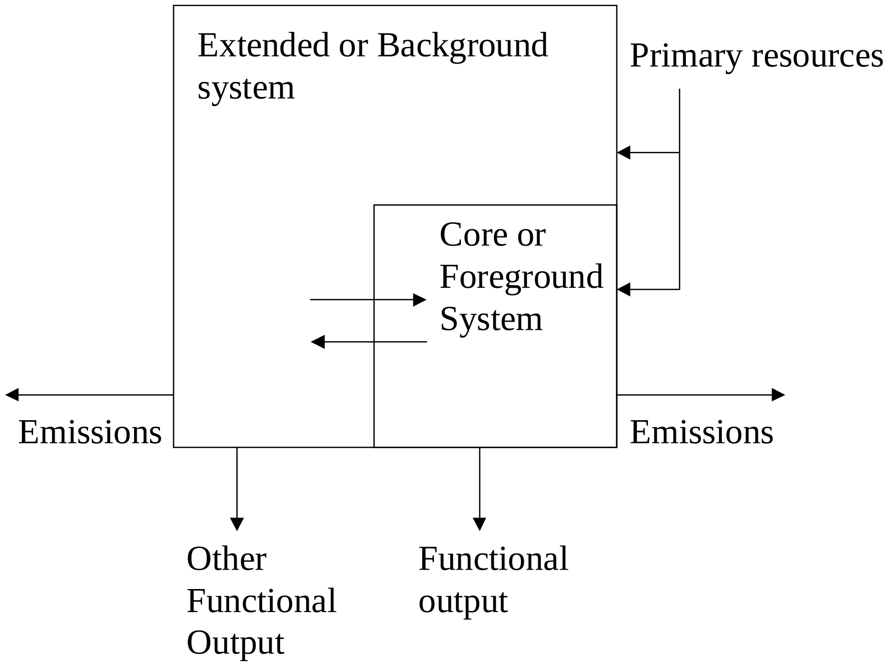

Figure 2. The system under study is the foreground system. The displaced burdens in the background system are the avoided burdens (Audsley et al., 1997).

Sometimes it may be convenient to allocate on the basis of economic value. This is seen as the least prioritised approach according to ISO 14041, because the price may fluctuate significantly within a short time period. In practice however, this approach has been used in many studies, exemplified by allocation between wheat-flour and wheat fodder meal, and cheese and whey (Mattsson & Stadig, 1999).

Inventory analysis

The inventory analysis includes a detailed description of the functions and boundaries of the system, data collection, calculation and assessment of sensitivities and uncertainties.

An LCA-study can either be based on typical, or average conditions, representing a relevant process or area, or be based on case-specific conditions. The choice depends mainly on the definition of the goal of the study. In most cases a mix of the two will be used for practical reasons.

Data gaps may affect the results from an LCA seriously. In order to discover if this gap is a hidden “hot-spot”, a worst-case scenario can be used. Data gaps should never be excluded, unless justified by other references.

Impact assessment

Occasionally, an evaluation can be drawn already from the results from the inventory, without using further stages in LCA. The result from such an inventory is called *Life Cycle Inventory* (LCI). More often however, an aggregation of the results is necessary to facilitate an overview of the impacts. This impact assessment includes classification, characterisation and valuation.

Classification and characterisation

The classification is made through addressing different environmental impacts to impact categories. The following characterisation is mainly a quantitative step, where different contributions to the impact categories are assessed.

Several lists on which impact categories to be included have been suggested. The Nordic guidelines (Lindfors et al., 1995) recommend the following impact categories to be studied.

- Resources - Energy and materials
- Resources - Water
- Resources - Land
- Impacts on human health (toxicological and non-toxicological impacts, excluding and including work environment)
- Global warming
- Depletion of stratospheric ozone
- Acidification
- Eutrophication
- Photo-oxidant formation
- Eco-toxicological impacts
- Habitat alterations and impacts on biological diversity
- Inflows not traced back to the system boundary between the technical system and nature
- Outflows not followed to the system boundary between the technical system and nature

The two last are not impact categories, but should be included according to the Nordic Guidelines.

The SETAC-Europe Working group on LCIA has suggested a similar list (Udo de Haes, 1996 in Finnveden & Lindfors (1997).

Input related categories
1. Abiotic resources (deposits, funds, flows)
2. Biotic resources (funds)
3. Land

Output related categories
4. Global warming

5. Depletion of stratospheric ozone
6. Human toxicological impacts
7. Ecotoxicological impacts
8. Photo-oxidant formation
9. Acidification
10. Eutrophication (including BOD and heat)
11. Odour
12. Noise
13. Radiation
14. Casualities

As a *pro memoria*, flows not followed up to system boundary are considered.

If a specific emission contributes to more than one impact category, it should be included under all headings. Emissions of e.g. CFC should therefore be included both in “Global warming ” and “Depletion of stratospheric ozone”, as both categories are potential impacts of the emissions. Secondary effects of an emission should on the other hand not be counted (Lindfors el al., 1995). Secondary effects could for instance be the effects on biodiversity and human health from green house gases emitted.

In specific case studies, some of the impact categories may be omitted. If the aim of a study is to evaluate the total environmental burden, all impact categories should be considered. Sometimes it is not possible to assess the impact due to lack of knowledge. Instead red flag classification can be performed in order to point out hazardous compounds, e.g. carcinogenic, banned or regulated chemicals.

Valuation

A valuation weights the different environmental impacts against each other. This cannot be based on natural sciences only, but need also political, ethical and administrative considerations (Lindfors el al., 1995). Use of different valuation methods, may therefore result in different conclusions. If valuation methods are to be used, it is recommended to use several different methods (ISO 14042).

Some valuation methods also include a normalisation step, when data from the actual study are related to the total magnitude of an impact category. Normalisation may in some cases provide a better platform for valuation and discussion, but a problem is the lack of relevant data and how to define the reference area under study.

Three different types of valuation methods exist according to the Nordic Guidelines:

- Case-specific, expert-based qualitative methods
- Case-specific, expert-based quantitative methods
- Formalised, quantitative methods

Mostly formalised quantitative methods are used in LCA. Within this type, different methods have been formulated. One is monetarising, based on the willingness a society shows for avoiding an emission or an impact. The EPS-system (Environmental Priority Strategies in product design) is one economic valuation of the environmental impacts. In EPS the safe-guard subjects human health, abiotic resources, biodiversity, ecosystem production capacity and cultural and recreational values are compared to the willingness to pay to avoid negative environmental impacts (Bengtsson, 2000; Steen, 1999).

In the Eco-indicator 99 method, three types of environmental damages are weighted, human health, ecosystem quality and resources (Goedkoop & Spriensma, 2000). As the underlying values, e.g. cultural orientation and view of nature, among the valuators will influence how different damages are scored, the result are presented for three groups separately, as well as one combined (Bengtsson, 2000). Hereby, both long-term perspective with high scientific uncertainties and short time perspective where only proven effects are included could be taken into account. A balanced time perspective is chosen as a default (Goedkoop & Spriensma, 2000).

Another method of valuation, called the “eco-scarcity”-approach, is based on load limits set by national environmental laws and regulations. These limits are then compared to the actual amount of the emissions in a specific area during a certain time-period (Lindfors el al., 1995). The eco-scarcity method does not explicitly weight the different goals against each other, something that may influence the result considerable.

Agricultural LCAs

The purpose of agricultural LCAs is to determine the differences in resource use and environmental impact between different agricultural systems with equivalent functions (Audsley et al., 1997). The application of LCA to agricultural production systems is a rather new phenomenon, which first has to establish how LCA may be applied to agriculture, and to point out methodological difficulties, which require further research. LCA was originally developed for industrial products. Methodological difficulties may therefore arise when LCA is applied to complex agricultural production systems, where the technical system is integrated into nature. Today there are intense international efforts to adopt LCA methodology to agricultural production systems and proposed recommendations have been made on a number of important issues (e.g. Audsley et al., 1997). Solutions how to solve methodological issues should be both pragmatic and realistic; pragmatic in order to be applicable for a broad group of users and realistic in that sense that an LCA should be as close to reality as possible (van Zeijts et al., 1999).

In the following section, some methodological issues with special emphasise on agricultural LCAs will be further discussed.

Soil - technical system or environment?

One important example of a methodological difficulty when LCA is applied to agriculture is whether agricultural soil should be considered as part of the environment or part of the production system, i.e. the technical system. If it is part of the environment, substances like nutrients and heavy metals applied to the field should be regarded as emissions to nature. If, on the other hand, agricultural soil is considered a part of the production system, then the application of those substances should only be considered as resource usage. In the latter case, attention should focus on those substances that leak out of the field into surrounding nature and those that are incorporated into plants, producing a toxic effect. A mix of these approaches has been proposed by Audsley et al. (1997). Agricultural soil is considered a part of the production system during the time period studied. After this period it passes the time boundary and becomes a part of the environment. In this way all remaining relevant changes made to the soil (soil productivity, the build-up of nutrients and heavy metals, soil compaction, biodiversity etc.) during the studied period are taken into account.

Another proposal how to look upon agricultural soil is exemplified by a Dutch study, where agricultural soil and crop residues are part of the environmental system, and the crop to be yielded is part of the technical system (van Zeijts et al., 1999).

Fertiliser in LCA

Most LCA studies on fertiliser take their starting point in the agricultural production, but also studies of different wastewater systems have considered fertiliser usage in agriculture inside the system boundaries through the avoided use of mineral fertiliser when sewage products are used on arable land. Bengtsson et al. (1997) draw the conclusion from three case studies of wastewater systems in Sweden that avoided use of mineral fertiliser in agriculture has a determining influence on the results from the whole system analysis. Their case studies included transports of fertiliser products and spreading of sewage products, but no other agricultural field operations. The authors enlightened the issue of substitutability of nitrogen and phosphorus of sewage products compared to mineral fertilisers. Only the plant nutrients nitrogen and phosphorus were included in the studies.

The industrial production of mineral fertilisers, especially nitrogen, requires a great amount of fossil fuel. Calculations show that 530 MJ is used for producing 1 kg calcium ammonium nitrate (N28) in Western Europe (Davis and Haglund, 1999). Nitrogen fixation in green manure crops will require a much smaller amount of fossil fuel (3 MJ/kg N), but instead a considerable amount of farmland will be used (approximately 70 m^2/kg N) (Mattsson, 1999). A thorough life cycle inventory of emissions and use of resources in industrial production of commonly used fertilisers, revealed that approximately 90% of the total energy requirement when producing NPK-fertilisers is used during the production of ammonia (Davis & Haglund, 1999). The products were studied from extraction of raw materials until the final products left the factory.

A study by Välimaa and Stadig (1998) considered environmental effects from usage of mineral fertilisers and how they can be identified and estimated in an LCA. Their study included nitrate leaching, ammonia emissions, N_2O emissions, phosphorus losses, the cadmium level in the soil and crop, changes in humus content and lime status of the soil when winter wheat is grown in different Swedish regions. The results from their study pointed out that plant nutrient issues are of significant importance when considering the environmental impact of vegetable production. Especially the contribution from fertiliser production proved to be of vital importance in the production system.

Fertiliser in a crop rotation

Another problem concerning fertilising is how to allocate the environmental burdens in a crop rotation. Fertilisers, especially relevant for phosphorus, may be applied to one crop, but some of it will also be available for subsequent crops in a crop rotation. Audsley et al. (1997) recommend that the environmental burdens associated with fertilisation should be allocated to each crop in the rotation according to the nutrient requirements by each crop. The allocation on different crops can be based on the recommended quantity for the crop. When an organic fertiliser containing many different nutrients is used, allocation can be based on the most limiting nutrient or by using the economic value of different nutrients. In the latter case, environmental burdens are allocated to the system under study in proportion to the financial value of the nutrients used by the system compared with the total content of applied nutrients. A Dutch proposal how to look upon different plant nutrients is that nitrogen should be allocated to the actual crop of application, and phosphate and potassium should be allocated to the different crops according to uptake and uptake efficiency per crop (Zeijts et al., 1999). It should however be noticed that the allocation of potassium is based on the behaviour in clay soil. Organic matter could be allocated according to percentage of total land use in a crop rotation.

Phosphorus as a non-renewable resources

Use of phosphorus ore processed to a fertiliser product in agriculture is assessed in the category "Resources – Energy and materials". This category can be further divided into subcategories, e.g. renewable and non-renewable resources. The use of raw phosphate in agriculture is of special interest as it is a limited resource and – if dispersed and not recycled – a non-renewable resource. But how should plant nutrient depletion be assessed when the actual uptake and loss is less than the use of nutrients? One suggestion by Cowell and Clift (1997) is that phosphorus depletion should be the amount used in the agricultural system minus phosphorus in sewage that is subsequently spread on farm land, minus phosphorus remaining in the soil (including incorporated straw).

Resource depletion in LCA is currently related to the size of remaining reserves of each resource. In order to facilitate a comparison of the severity of different environmental impacts to resource depletion, the following approaches have been discussed (Audsley et al., 1997).

1. Downstream analysis. The degree of dispersion should be taken into account. If for instance nutrients are recovered in the sewage sludge, they should not automatically been looked upon as dispersed and depleted. But if they are not used on land but discharged into rivers, lakes or the sea, they could be looked upon as dispersed.
2. Hypothetical closure of the life cycle. The starting-point is that future generations may need to extract the same kind of resources as the generation of today, but with lower quality. The product system should therefore even include future recovery of the resources. First in the analysis the quality of the materials leaving the system should be established. Next step is to determine technologies and energy demand for retrieval. The hypothetical closure of the life cycle of phosphorus may act as an example. It is assumed that the present exploitation of sediment rock can continue for quite a long time. Thereafter it is assumed that organic sources and waste streams will be used. Therefore no additional resource value has been ascribed to the use of phosphorus.
3. Exergy analysis is quite similar to the idea of hypothetical closure of the life cycle, but the future energy requirements are based upon thermodynamic optimal limits.

Land use as an impact category

Land use is associated with many and severe environmental impacts. Human land use will both make use of land that could be used for other purposes, and have an impact on biodiversity and life-support functions, i.e. the quality of land (Guinée, 2001). In a global context, the area available for agricultural production is a limited resource. However, in Sweden, the area required for agricultural production has been less then the available area, with the consequence that one fourth of the former agricultural area from 1940 until now has been abandoned, and in most cases transformed to forest (Larsson, 1997). This has a severe impact on biodiversity in Sweden as two thirds of the threatened species among the vascular plants are hosted in habitats formed by historical agriculture (Naturvårdsverket, 1994a).

However, in other parts of the world it is mainly the natural habitats that are hosting biodiversity.

Land use issues initially received limited attention in LCA methodology, although the environmental impact on land is considered very important for agriculture and forestry (Finnveden & Lindfors 1996). However, land use has started to receive increasing attention since the late 90s. The impact category land use has traditionally been used only to describe how large area an activity is occupying. 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, 2000b). A special task for LCA concerning agriculture and forestry has therefore 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. Whether land use assessment deals with a net change of the land, or with the occupation in it self, will have an influence on how the impact category is handled.

Dimensions in time and space

Land use within the agricultural system has both a temporal and a spatial dimension. When considering the temporal dimension, care must be taken to include all uses of land, for example fallow periods, liming and green manure crops. Whether an assessment of one crop should be performed, or if instead one or several crop rotations should be considered, depends upon the purpose of the study. A time scale including at least one crop rotation could be appropriate when the purpose is information for agricultural policymaking, while in a comparison of different foodstuff, it may be appropriate to use a much shorter time scale (Cowell & Clift, 2000). A shorter time scale will be less time-consuming, but will at the same time introduce an increasing number of allocation issues. The spatial dimension includes usually the furrow slice on agricultural land. Two additional aspects should also be considered according to Cowell and Clift (2000) to get a more complete picture of the land use impacts; subsoil compaction and nutrients leaching into the subsoil

Another interesting aspect on land use is whether to take into account the land transformation or not. Land transformation in e.g. Brazil is of high relevance where natural vegetation is cleared for agricultural production with severe impacts on the natural biotopes as the result. In Sweden, the land transformation to agricultural production took place a long time ago. According to Cederberg (2002) it is therefore reasonable to omit the effects of historical land transformation, since it is not affected by today's decisions.

Choice of indicators

Authors have recently suggested different indicators and criteria, but no harmonisation has been reached so far. A starting point should be a simple list of indicators, and when required, more detailed and sophisticated indicators should be used. Today, most indicators suggested relate to biodiversity, and are measured as vascular plant diversity, due to lack of more extensive data (Lindeijer, 2000b).

To assess physical habitat depletion, Cowell (1998, in Cowell & Lindeijer, 2000) uses four biodiversity indicators, i.e. area, number of rare species, number of species and number of individuals. As indicators for productivity, organic matter and soil compaction can be used.

Lindeijer (2000a) has proposed two indicators for assessing land use impacts on a global scale; vascular plant species diversity and free net primary biomass production, as these are considered as the most important contributors to the ecological values of an area. Life support functions consider the maintenance of the processes in an ecosystem, e.g. closing the substance cycles, climate regulation and maintaining a well-functioning soil structure. It therefore stands for the productive, adaptive and renewing capacity of land, water and biosphere (Lindeijer, 2000a). As an indicator for life support, free net primary biomass production (fNPP), i.e. the amount of biomass the nature can apply for its own development, is chosen. The carbon level of the soil is also indicated by fNPP.

Mattsson et al. (1998) have suggested criteria and indicators for soil properties according to a goal aiming at assuring biological production and soil fertility. Criteria and indicators for direct impact on biological, physical and chemical properties suggested by them are as follows:

- *Criteria for biological properties*: maintaining a good level of organic matter in the soil, a diversified soil fauna and using cultivation methods, which are not leading to increased weed problems. Example of an indicator is content of organic matter of the soil.
- *Criteria for physical properties:* avoiding soil erosion and promoting a good soil structure and efficient water drainage. Example of an indicator is soil losses caused by erosion.
- *Criteria for chemical properties:* assuring favourable soil properties and avoiding accumulation of heavy metals. Examples of indicators are pH, P-AL (plant available phosphorus) and CEC (cation exchange capacity). Metal balance in order to determine the accumulation of heavy metals could also be calculated.

Also crop variety in the agricultural landscape is suggested as a criterion, as a mono-cultural dominance in an area, makes the crop under study more susceptible to attacks from insects and fungus. A second goal proposed by the authors, aims at assuring a diversified rural landscape of high aesthetic value, which also has a high ability to maintain resilience of the ecosystem (Mattsson et al., 1998).

The approach described above was further tested in three case studies of cultivated vegetable oil crops: Swedish rape seed, Brazilian soybean and Malaysian oil palm (Mattsson et al., 2000). The results point out the indicators erosion, soil organic matter, soil structure, soil pH, phosphorus and potassium status of the soil and the impact on biodiversity as possible to get information about. On the other hand, data on heavy metal accumulation and impact on aesthetic landscape values were more difficult to obtain (Mattsson et al., 2000).

Cowell and Clift (2000) have also suggested a methodology for assessing soil quantity and quality in LCA. Relevant factors affecting soil properties listed by the authors are found in Table 5.

Table 5. Different factors affecting soil properties (Cowell & Clift, 2000)

Factor group	Factor
Soil quantity	
Mass of soil	Loss from erosion
	Addition from incorporation
Soil quality	
Living organisms	Weeds and weed seeds
	Micro- and meso-organisms
	Pathogens
Trace substances	Nutrients
	Heavy metals
	Pesticide residues
	Salts
	pH of soil
Non-living matter	Organic matter
	Water in soil
Form of soil	Texture
	Structure

All factors mentioned above will have an impact on four of the five safeguard subjects used in the EPS system; i.e. future agricultural productivity, availability of resources, biodiversity and human health. The fifth safeguard subject, aesthetic values, is in contrast to the suggestion by Mattsson et al. (1998) not considered. Most of the factors listed in Table 5 should already be assessed in current Impact Assessment methodology, and do not therefore need further attention. Human health impacts of heavy metals and pesticides are for example included in the "Human Toxicity" category (Cowell & Clift, 2000).

Changes in mass of soil is mainly related to soil erosion, at least when considering a few years. One exception is the change from grassland to cultivated arable land (Cowell & Clift, 2000). The authors suggest that soil depletion should be ranked alongside concerns about depletion of other resources, as the rate of erosion is much larger than the formation of new soil, underlining the fact that current agricultural practice is unsustainable.

Aggregation of data

Whether to strive towards a single index for land use or keep the information without making any aggregation is a debatable issue. Audsley et al. (1997) discuss if it should be possible to aggregate quantitative data for soil quality into a value related to their impact on the potential crop yields. Factors taking into account should be biological (weed population, soil flora and fauna, humus content), physical (erosion, soil density, available water content) and chemical (pH, salinity, nutrient availability, heavy metals, organic contaminants). Their conclusion is that if aggregation of these values into one single is possible, or even appropriate, needs further research.

Cowell and Lindeijer (2000) also discuss the possibilities of integrating indicators into one single score. Weighting different indicators for biodiversity requires that the contribution from each indicator could be establish, with involvement of experts. For the two indicators biodiversity and life support, no relative weighting is proposed due to lack of scientific knowledge (Cowell & Lindeijer, 2000).

Others do not recommend data collected to be aggregated, at least when considering agriculture. Mattsson el al. (2000) mean for instance that it is better to allow the land use category to include both descriptive and non-aggregated parts, compared to other impact categories in an LCA.

GOAL AND SCOPE DEFINITION

Objective

The main objective of this study was to evaluate the consequences on environmental impact and resource management when human urine replaces mineral fertiliser in arable farming. Production of winter wheat and spring barley when only mineral fertilisers were used was compared with a scenario where a combination of human urine and mineral fertilisers was used. The method for assessing the two different scenarios was LCA, *Life Cycle Assessment.*

Sub-objectives were to analyse how different aspects, e.g. transports and materials required for the separated system, influence the total environmental impact, and discuss the magnitude of these impacts to the environmental impact from grain production. For example, how far can human urine be transported before the energy required for transports exceeds the energy saved when urine is used as fertiliser instead of mineral fertiliser? Will the choice of material in the infrastructure needed for use of human urine, e.g. storage tank, pipes etc., be of importance for the environmental outcome? What environmental impact will the handling of urine on the farm have?

Functional unit

The functional unit of this LCA-study was 1 kg of grain (wheat and barley respectively) leaving the farm gate.

Scenarios

Two different scenarios were assessed in each grain production system. In the first scenario, here called the conventional scenario, grain production in accordance to normal practice in the region of Mälaren was considered. In the second scenario, the urine-separating scenario, mineral fertiliser was partly replaced by human urine. This scenario was constructed in accordance with such requirements the farmers could claim, e.g. the handling should be possible to put into practice. For this reason it was for example assumed that urine spreading only occurred in the growing crop and not before sowing. In the clayey soils characteristic for the region surrounding Mälaren, urine spreading during springtime could otherwise result in severe soil compaction, due to the heavy equipage used. The labour consumption is also intense during spring, which makes spreading in the growing crop preferable.

The urine in the urine-separating scenario was assumed to be separated at the source, while the faeces were treated together with other wastewater fractions in a wastewater treatment plant. Purification of both N and P was assumed in the treatment plant. Further description of the scenarios is found in the chapter below and in the inventory.

System boundaries

System description

In the conventional scenario, all activities related to yearly grain production were included. Hereby, agricultural activities performed in the field, production of mineral fertiliser and fuel, as well as transportation was included. Agricultural production using human urine as fertiliser, as in the urine-separating scenario, will fulfil a function beyond the actual food production, i.e. providing wastewater handling. The source-separating system will also affect the water consumption used for flushing. In a systems analysis the systems under comparison must deliver the same functions. This can be done either by adding subsystems that will provide additional functions, or by subtracting subsystems with the excessive functions (Lindfors et al., 1995). Here subtraction was used, i.e. the system boundaries included parts of the water and wastewater system in the urine-separating scenario in the sense that avoided burdens (avoided emissions and use of resources) when separating human urine were subtracted from the agricultural system. The system boundaries are shown in Figure 3.

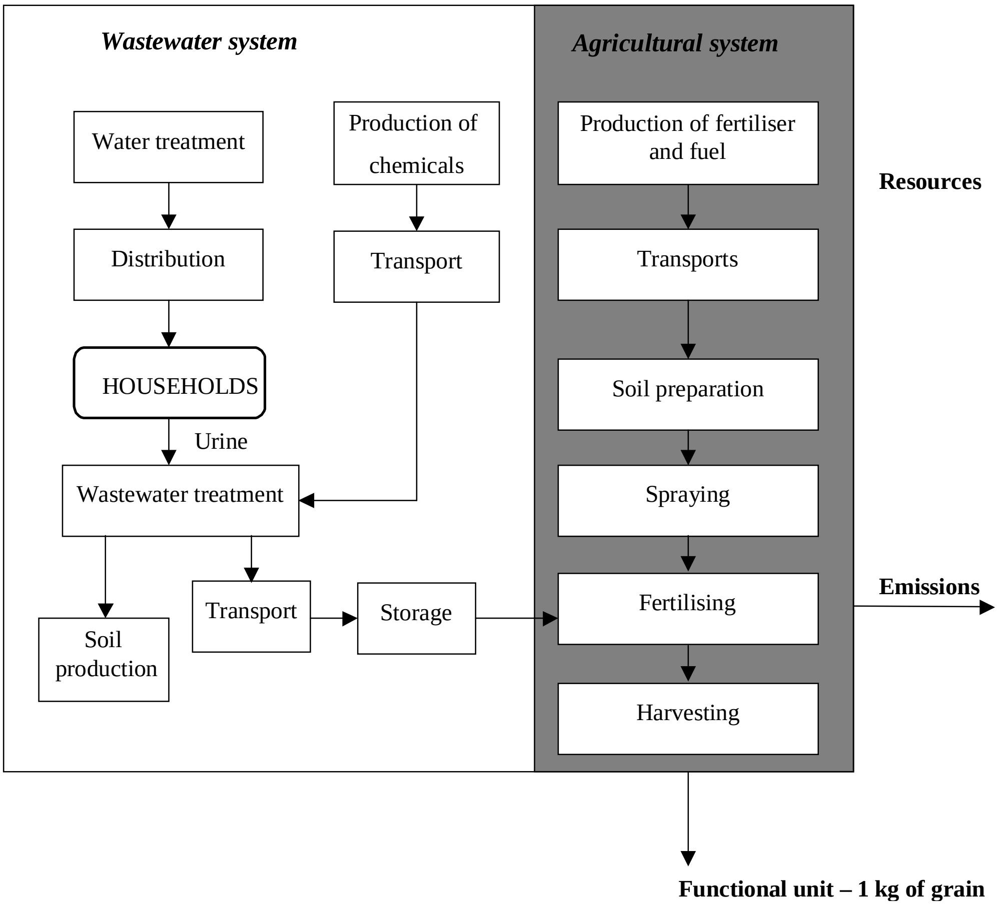

Figure 3. Schematic flow chart of the systems studied. In the conventional system only activities within the agricultural system are taken into account. The urine is assumed to be treated in a wastewater treatment plant, which produces sludge used for soil production. In the urine-separating system the urine is instead collected and then transported to a farm where it is used as fertiliser. Differences arising when separating urine are taken into account as avoided burdens in the urine-separating scenario.

Boundaries related to the production capital

All main changes in the urine-separating scenario compared to the conventional scenario (the reference) were considered, also including production of capital goods, e.g. machines and constructions, not identical in the two scenarios. Therefore, the construction phase of the pipes and storage tanks both on household level and farm level was included in the scenario using human urine, as the construction of those was exclusive for the separating system. For the same reason the energy consumption related to the production of the spreader was taken into account. The urine-separating toilets were considered to be installed only when the old toilets were assumed to be exchanged. The construction phase of those were therefore not taken into account. The way the additional mineral fertiliser was stored in the conventional scenario was not taken into account, as the volume, compared to urine, is very small, and because no special arrangements around the storage is necessary. No other consideration was taken to the resources and energy needed to construct and maintain the machinery and buildings used in agriculture or in the wastewater system. The environmental impacts from this phase are not always negligible when compared with the actual running, but were not considered to be of major importance when comparing these scenarios to each other, since they were similar.

Boundaries in relation to natural system

There is no clear distinction between the technical system and nature in an agricultural system. However, for an LCA, this distinction must be made, as the emissions are accounted for when they pass this boundary. In this specific study, the tilled soil (<20 cm) was considered as part of the technical system, and the subsoil was part of nature.

The wheat and barley straw was assumed to be incorporated in the soil in all systems, which is a common practice in the region studied. For this reason, no allocation between the grain and the straw had to be made.

Data quality and time perspective

Used data and parameters for soil qualities, cultivation practice etc. in this study represented typical condition for the region surrounding the lake Mälaren; i.e. here the counties of Stockholm, Uppland, Södermanland and Västmanland (Figure 4). Data used for the wastewater handling had however a more general character and was taken from case studies and literature reviews.

The time perspective of the study was prospective. However, as the systems studied were analysed as if they were to be shaped in a near future, data from today's agricultural practice and wastewater handling were used. The time-horizon for the field operations was restricted to one year; therefore no attention was paid to field operations not annually performed, such as liming and drainage. These operations were also considered to be the same between the scenarios. As the time perspective was prospective, modelling a change, use of marginal data on electricity is sometimes recommended. In this study average data on electricity was chosen, but the influence from the use of marginal data was examined in a sensitivity analysis.

Figure 4. Map over Mälardalen.

Studied impact categories

Today's use of scarce resources is a threat for future generations, as the reserves worth using are limited. As modern agriculture is highly dependent on purchased inputs such as fossil fuels and phosphorus ore, these categories are considered important here. However, water –a scarce resource in many parts of the world – is under most Swedish conditions not a limiting resource.

Contribution to global warming is closely linked to the use of fossil fuel, as 78% of the green house gases emitted from Sweden originate from combustion and transports (Naturvårdsverket, www). However, in agriculture, as a whole, the contribution of nitrous oxides and methane is considered to have a greater impact on global warming potential than the emissions of CO_2 (Cederberg, 2002).

Eutrophication is one of the environmental quality goal set up by the Swedish Parliament. Agriculture is one of the major contributors to eutrophication, both to lakes and the sea. 37% of the nitrogen load on the Baltic Sea comes from agriculture. The wastewater sector gives a nitrogen contribution of 29% (Naturvårdsverket, 1997). Ways to reduce eutrophication therefore include both actions in the agricultural sector as well as in the wastewater sector, therefore this impact category is highly relevant in this study.

Another Swedish environmental goal is decreased acidification. The objective is to strive towards a situation with only natural acidification and where the acidifying depositions are below the critical load limits. However, the critical loads and fate of the emitted substances are not easily modelled and differ considerable between different areas. In "Svealands slättbygder" (where this scenario study was placed) the reduction required for reaching the goal of "natural acidification only" is considered as low or even none existing (Jordbruksverket, 1999a). In these calculations, depositions from other countries are not included, which have a serious effect on the results. Acidification is however included in this study, due to its relevance for many parts of Sweden, especially the south-western parts. Another reason for including acidification is that a considerable part of the ammonia emissions, which are acidifying, are transported away and may therefore cause environmental damage far away. 40% of the Swedish ammonia emissions are for instance transported and deposited abroad (Jordbruksverket, 1999a).

Photo-oxidant formation, originating from e.g. emissions from the transport sector, may deteriorate the growth of wheat and other crops. Emissions of these substances are not considered to be a specific problem for the agricultural sector or for the wastewater sector.

However, as transports are required in the systems studied, this impact category is therefore included.

According to the above, the following impact categories among the list suggested by Lindfors el al. (1995) are reported in this study.

- Resources – energy, materials and water
- Global warming
- Eutrophication
- Acidification
- Photo-oxidant formation

Discussed, but not quantified, are also the following categories:

- Land use
- Impact on human health

Flows of plant nutrient and cadmium were further quantified and presented for the different scenarios. Use of pesticides was part of the inventory, but was not further assessed in an impact category due to methodological problems.

The categories *not* included here were eco-toxicological impacts (besides the inventory of pesticides used), depletion of stratospheric ozone and impacts on biological diversity, as these impacts were not considered to differ between the scenarios.

INVENTORY OF THE SYSTEMS STUDIED

In this chapter follows a detailed description of the systems under study. Inventory data from the two scenarios and the two different crops are summarized in Appendix 1-4.

Description of the area studied

This study included cultivation of two important Swedish crops, winter wheat and spring barley. Used data and parameters for soil qualities, cultivation practice etc. in this study represented typical conditions for the region surrounding the lake Mälaren. Characteristics for the climate and soils in the area as mean values are shown in Table 6. This region accounts for a considerable part of the Swedish population, today 2.6 million inhabitants (Statistics Sweden, 2001). The region also represents a considerable part of the total grain production in Sweden.

Table 6. Characteristics for climate and soil in the region of Mälaren

Characteristics		References
Type of soil texture	Sedimentary clay soil (80%)	Mattsson, 1996
Humus content	~6%[a]	Eriksson et al., 1997
pH	~6.3	Eriksson et al., 1997
P-AL (labile P)	~10 mg/100 g dry soil (Class IV)	Eriksson et al., 1997
K-AL (labile K)	~20 mg/100 g dry soil (Class IV)	Mattsson, 1996
Cadmium content	~0.3 mg/kg	Eriksson et al., 1997
Vegetation period	180-200 dagar	SNA, 1992
Yearly precipitation	600-700 mm	SNA, 1992
Deposition of N	5.5 kg N/ha	Jordbruksverket, 1999b

[a)] Average figure including organogenic soils. The corresponding figure for the median value is 4.1%.

A common crop rotation on farms without livestock in this region is winter wheat, barley, fallow, winter wheat, rape crop or peas (Strand, pers. com. 2002).

Field operations

A typical production scheme for grain production in the region is schematically described in Figure 5.

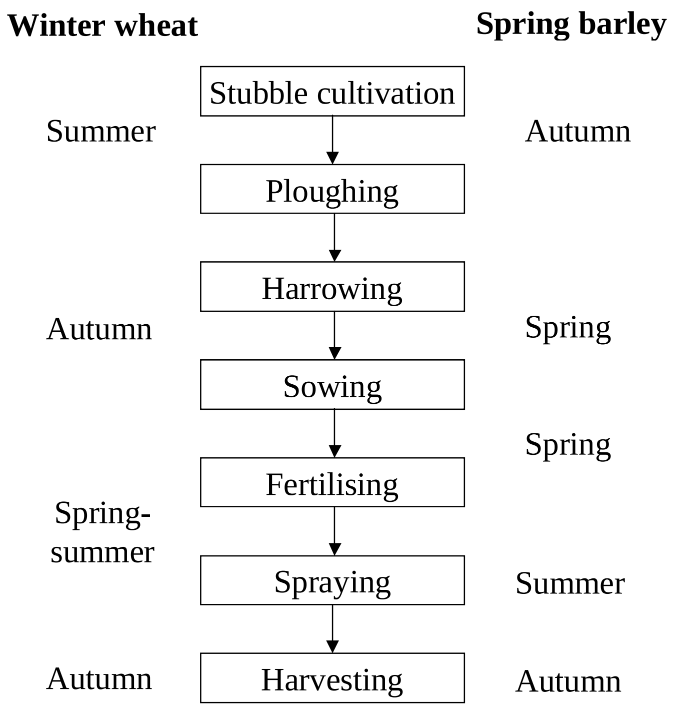

Figure 5. Field operations usually performed in the area. Harrowing is usually performed several times.

Data of exhaust emissions when performing different agricultural field operations (Table 7) were taken from Hansson and Mattsson (1999). The results presented by them reveal that emissions in relation to the energy in the fuel vary considerably between different tractor operations, a fact that makes it inappropriate to use data based on measurements from other vehicles. Generally tractor data used, included for example in the Swedish software for LCA calculations – LCAiT, are based on measurements from road traffic vehicles.

Table 7. Fuel consumption and exhaust emissions when performing different field operations (Hansson & Mattsson, 1999) [1]

Field operations	Fuel consumption (l/ha)	CO (g/ha)	NO (g/ha)	HC (g/ha)
Stubble cultivation	5.41	14.6	143	5.68
Ploughing	16.4	49.3	573	17.1
Harrowing	6.25	9.27	198	3.49
Sowing	3.47	13.2	117	4.20
	(l/km)	(g/km)	(g/km)	(g/km)
Transports [2]	0.6	2.83	16.8	0.687

[1)] These measurements were made on standard diesel with an energy content of 42.8 MJ/kg and a density of 0.826 kg/l.

[2)] The emission values from transports originate from transport of grain from field to farm using a 12-ton tandem-wheel trailer.

Plant nutrients

Recommendations

The expected yield is one important factor for determining the quantities of plant nutrients needed. Especially the rate of nitrogen fertilisation will strongly depend on expected yield as shown in Table 8.

Table 8. Nitrogen recommendations[1] *on wheat and barley (kg/ha) according to yield (Jordbruksverket, 1999c)*

	Expected yield, ton per hectare					
Crop	4	5	6	7	8	9
Wheat		110	130	150	170	190
Barley	70	90	110	130		

[1)] Calcium nitrate is assumed. If ammonium nitrate is used, application rate shall be approximately 10–15% higher (Linder, pers. com.).

When organic fertilisers are used, the amount of nitrogen accounted for is normally based on the content of easily available nitrogen, i.e. nitrogen in the form of ammonium and nitrate. The long-term effect from organically bound nitrogen is estimated to be 10 kg of nitrogen per hectare and year, when in average one ton dry matter of e.g. manure is applied per hectare and year (Jordbruksverket, 1999c).

The requirement for sulphur depends on the cultivated crop and expected yield, and is for cereals approximately 10 kg per hectare. The amount of phosphorus and potassium needed depends both on soil fertility status and expected yield. Recommendations from the Swedish Board of Agriculture on application rates for phosphorus and potassium are shown in Table 9.

Table 9. Recommendations on nutrient application of phosphorus and potassium according to soil class (P-AL and K-AL respectively). The recommendations are based on a yield of 5 tons of grain per hectare (Jordbruksverket, 1999c)

	Amount, kg per hectare, according to soil class (P-AL and K-AL respectively)				
Fertiliser	I	II	III	IV	V
Phosphorus	35	25	15	10	0
Potassium	65	45	25	5	0

A common recommendation if the soil class is III-IV, is to add as much phosphorus as is removed by the crop. Due to weathering of the potassium-rich clay soils, no potassium fertilisers are normally needed in the region of Mälaren (Linder, pers. com.).

In Table 10, the plant nutrient content in the kernels of wheat and barley is shown, which gives the amount of plant nutrients removed from the field if the straw is left on the field.

Table 10. Plant nutrient content in the kernels of wheat and barley (Jordbruksverket, 1999b)

Crop	N	P	K	S
Winter wheat (%)	1.7	0.31	0.43	0.1
Winter wheat (kg per 6 tons) 1)	102	19	26	6
Barley (%)	1.6	0.34	0.43	0.11
Barley (kg per 4.4 tons) 1)	70	15	19	5

1) The exemplified yields of winter wheat and barley are the expected yields in the area where this scenario study is located.

Mineral fertiliser production

Most of the energy required for producing nitrogen fertiliser products arises from the production of ammonia. Natural gas is the dominant energy carrier in this process. When phosphorus fertiliser products are produced, mostly oil and electricity are used. The environmental loads associated with production of mineral fertiliser products (Appendix 5) were taken from a thorough inventory made by Davis & Haglund (1999).

Urine mixture

The figures on concentration of plant nutrients in the urine mixture used in this LCA were from a study by Olsson (1995), in which the nutrient concentrations in seven different systems were reported. These values were also considered as representative according to the compilation of data in Table 2. The median concentrations are shown in Table 11.

Table 11. Concentration of nitrogen, phosphorus, potassium and sulphur (kg/m^3) in human urine mixture used in this LCA-study

Tot-N	NH_4-N +NH_3-N	P	K	S
2.3	2.1	0.23	0.71	0.19

The low concentration of the urine mixture indicates that a considerable dilution has occurred. According to measurements of undiluted urine, the concentration of phosphorus is about 1 g/l

(Hellström & Kärrman, 1995). Dilution can for example be the result of much water added in the toilet, or a result of groundwater leaking into the pipes.

During storage before spreading, considerable losses may occur. These can however be reduced significantly with sufficient coverage. Here 5% losses of the nitrogen content was assumed, which is in comparison with losses during storage of cattle urine under e.g. a roof (Jordbruksverket, 1997).

Cultivation of winter wheat

Winter wheat is largely used for human consumption. It is preferably cultivated on well-drained clay soils with a pH exceeding at least 5.5. Standard yield (i.e. yield to be expected) is 6 tons for winter wheat in the region studied (Statistics Sweden, 2000a).

The seed required for one hectare is normally 210 kg per hectare (Odal, undated). The seed production was taken into account through subtraction of the amount of seed needed, from the total yield.

Tractor operations

Fuel consumption and discharge of emissions when performing different field operations are shown in Table 12. Assumed data for fertilising, spraying and harvesting was taken from Hansson and Mattsson (1999). The emissions related to the energy consumption during fertilisation and spraying was assumed to be the same as the emissions related to the sowing operation. As no specific emission data exist for harvesting, these figures were based on data on ploughing.

Table 12. Fuel consumption and discharge of emissions when cultivating winter wheat in the conventional scenario

Operation	Fuel consumption (l/ha)	CO (g/ha)	NO (g/ha)	HC (g/ha)
Stubble cultivation	5.4	14.6	143	5.7
Ploughing	16.4	49.3	573	17.1
Harrowing (3 times)[1)]	16.1	23.9	511	9.0
Sowing & P-fertilising	3.8	14.5	129	4.6
Fertilising (2 times)	4.4	16.8	148	5.3
Spraying	2.1	8.0	70	2.5
Harvesting	15	45.1	524	15.6

1) A reduction of the energy consumption of 15% is assumed for every following harrowing according to measurements reported by Danfors (1988).

In the scenario using human urine, mineral fertiliser was assumed to be spread once, and a second application of nutrients was made as urine. Spreading of urine using band application with 12 metres working width may consume 3 litres diesel per hectare (de Toro, pers. com.). Filling the spreader's tank will consume additional fuel; here 1 litre of diesel was assumed for a tank of 10 m^3. Emission data for the spreading operation were set to be the same as for transports, i.e. 0.15 g CO/MJ, 0.90 g NO/MJ and 0.037 g HC/MJ (Hansson & Mattsson, 1999).

Emissions of CO_2 during combustion are defined as 74.6 g/MJ (Tillman, 1991). The emissions of SO_2 depend on the sulphur content of the fuel. Here the value of 0.0935 g/MJ in Swedish standard diesel was used (Hansson & Mattsson, 1999).

Fertilisation

Preferably, nitrogen fertilisers should be added only when needed by the crop, i.e. for cereals in spring and summer, otherwise considerable nitrogen losses may occur. The best time for spreading fertilisers is early in the spring. If the application rate will exceed 120 kg N, the recommendation is to split the rate on two or three occasions. The greater part, 90-100 kg N, should normally be spread on the first occasion (Weidow, 1999). The fertiliser product Axan (NS27-3) is frequently used in wheat production in the region (Linder, pers. com.). However, due to lack of data from the production of Axan, inventory data from N28 was used in this inventory.

General assumption: An equal amount of plant-available nitrogen (as ammonium and nitrate) was applied as fertiliser in both scenarios; 145 kg of nitrogen per hectare before volatilisation. The plant-available nitrogen after volatilisation was then assumed to have the same yield-increasing capacity as nitrogen in mineral fertiliser. The amount of phosphorus applied was of the same order as the amount of phosphorus removed by the crop. Due to the potassium-rich clay soils, no potassium fertilisers were used.

Conventional scenario:

Nitrogen was assumed to be applied twice; 145 kg of N per hectare in total. Both applications were as ammonium nitrate (N28). It was further assumed that 19 kg of phosphorus per hectare was applied in the autumn together with the winter wheat seed.

Urine separating scenario:

The strategy chosen was to add half of the available nitrogen as mineral fertiliser and half as urine. 11 kg of phosphorus per hectare was applied in the autumn as mineral fertiliser together with the seed. In springtime, an application of 72.5 kg of nitrogen as mineral fertiliser was assumed. Thereafter 34.5 tons of urine was spread, containing 72.5 kg of plant available nitrogen (before losses) and 7.9 kg of phosphorus. The urine application also supplied the crop with 25 kg of potassium and 6.6 kg of sulphur.

Pesticides

One herbicide treatment yearly with a mixture of Express and Starane, fungicide treatment with Amistar every second year and stubble treatment with Roundup Bio every seventh year is a common practice in the region studied (Linder pers. com.). The amount of active substances applied to the field, based on these estimations, is shown in Table 13.

Table 13. Typical yearly use of pesticides in wheat production in the region studied

Product name	Dose rate per hectare (l)	Active substance (per kg/l)	Total amount of active substances added per hectare
Express 50T	1.5 tablet	500 g tribenuronmetyl (3.75 g/tablet)	2.8 g tribenuronmethyl
Starane 180	0.6	180 g fluroxipyr	108 g fluroxipyr
Amistar	0.8/2	250 g azoxystrobin	100 g azoxystrobin
Roundup Bio	3.5/7	360 g glyphosate	180 g glyphosate

N-emissions

Ammonia emissions

Volatilisation of ammonia from urine was set to 5% of the total NH_4-N in urine, i.e. 3.6 kg per hectare. The emission factor used was in accordance with the results from field trials performed (Richert Stintzing et al., 2001). A prerequisite for low losses is the use of a good technique, for example band application with harrowing within four hours or application in a growing crop.

According to CORINAIR (European Environment Agency’s programme for inventories of airborne emissions) ammonia volatilisation from ammonia nitrate fertilisers are reported to be approximately 1% of the total nitrogen content (Jordbruksverket, 1997). But there are other references pointing out considerably smaller emissions. JTI draw the conclusion from three years of field trial that ammonia emissions from KAS may be less than 0.2% (Svensson et al., 1999). The emission figure used here was 0.6% of the total nitrogen content in the mineral fertiliser products.

According to references in Välimaa and Stadig (1998), ammonia emissions through the leaves are approximately 1.5 kg NH_3-N per hectare and year. This figure was used in both scenarios.

Nitrate leaching

The amount of nitrate leaching from arable land depends on many factors such as soil type, crop and climate, and may therefore show considerable spatial and temporal variations. Both measurements and computer-based models can be used for estimation of nitrate leaching on a local scale. Simplified empirical models for estimations have been developed, for example in the computer programme STANK used for advisory purposes. Its sub-model, developed by Hoffman et al. (1999), was used here. The model takes into account soil type, precipitation, dosage of fertiliser, manure used in the crop rotation, NH_4-content in different organic fertilisers, soil preparation etc. The leaching of nitrate both when using human urine and mineral fertilisers was calculated to be 10 kg nitrogen per hectare (Jordbruksverket, 1999b).

NO_X-emissions

The losses of NO_X (as NO and NO_2) from arable land are considered to be very small. Less then 0.1% of the total nitrogen applied is reported to be emitted (Svensson et al., 1999). The above figure of 0.1% was here used as emission factor.

N_2O-emissions

N_2O-losses from arable land depend on many different factors, including for example soil type, water content and nitrogen availability. No data exist today for relating N_2O-losses specifically to different crops or fertiliser products. An emission factor of 1.25% N_2O-N losses of the total nitrogen applied is proposed by IPCC (1997). According to a study performed by JTI, the total N_2O emissions from a sandy soil with a high organic content in southern Sweden were 2 kg N_2O-N per hectare and year (Svensson et al., 1999). However, the N_2O emissions directly related to the use of mineral fertilisers were only 0.1% of the nitrogen applied. The emission figure used here was 1.25% N_2O-N losses of the total nitrogen applied. The same emission factor from urine as from mineral fertilisers were assumed, but for urine the volatilisation of ammonia was first taken into account through subtraction of the NH_3 emitted from the total N applied.

Nitrogen used in agriculture will also give rise to indirect N_2O emissions. Nitrogen emitted directly to ground water and surface water undergoes nitrification and denitrification, and hereby production of N_2O will occur. Atmospheric deposition of e.g. nitrogen oxides and ammonium fertilise the soil and water and will thus enhance biogenic N_2O formation (IPPC, 1997). A proposal from IPPC (1997) is to calculate indirect emissions of N_2O as 0.01 kg N_2O-N/ kg NO_X-N and NH_3-N emitted. The emission factor proposed per kg N leaching is 0.025 N_2O-N/ kg N. These figures were also used in this study.

P-emissions

Phosphorus lost, as well as nitrate, will contribute to the eutrophication of lakes and the sea. The average losses of phosphorus in Swedish agriculture are estimated to be 0.3 kg per hectare and year, with a huge variation in time and space. Losses in the range of 0.01-3.4 kg per hectare are not unusual (Ulén, 1997). Results from a water quality monitoring programme run by SLU indicate that phosphorus losses from a clay soil in the area studied, could be about 0.5 kg per hectare and year (Johansson et al., 1999). Losses of 0.5 kg phosphorus were here used independent of crop and type of fertiliser product used.

Soil compaction

Intensive field traffic with tractors and heavy vehicles, e.g. slurry spreaders, leads to soil compaction, which may affect plant growth, production costs and environmental effects. As a tool for predicting the effects, a computerized empirical model for estimating crop yield losses has been developed (Arvidsson & Håkansson, 1991). Parts of this model were used here; i.e. the parts covering yield losses caused by structural damage in the topsoil persisting after ploughing, and yield losses due to subsoil compaction when spreading urine.

Yield losses caused by soil compaction

The cumulative yield reduction, due to soil compaction caused by the spreading, in percent of one year's yield was 0.9%, when including the effects in both the topsoil and the upper layer of the subsoil (Table 14, Appendix 6). The persistent effects in the deeper part of the subsoil were calculated to 0.005% yearly. When the reduction during the next 100 years due to spreading urine one year were addressed to the crop under study, these yield losses were another 0.5%.

Table 14. Effects on yield from the spreading of urine

	Topsoil	Subsoil (25-40 cm)	Subsoil (>40 cm)
Yield losses (% of one year's yield)	0.74	0.20	
Future annual yield losses (%)			0.0052 [a)]

a) due to permanent compaction of the deep subsoil. Accumulation during 100 years is 0.5%.

Effect of wheel traffic in the growing crop

Wheel traffic in a growing crop will partly damage the crop and thereby decrease the yield. Results from spraying operations in late springtime indicate that the yield reduction may be of the magnitude of 2% in winter wheat (Jordbruksverket, 2000). This figure was used here.

Yield

The yield reached was set to 6000 kg per hectare in the conventional scenario and 5650 kg in the urine-separating scenario, i.e. the yield in the urine-separating scenario was 94% of the yield in the conventional scenario. The assumed difference in yield was explained by the higher ammonia losses in the urine-separating scenario (2.4%), as well as effects from soil compaction and wheel traffic in the growing crop (1.5 and 2% respectively).

The more immediate as well as future effects from the soil compaction were both included in the yield received in the urine-separating scenario as a hypothetically reduction on the yield of the year under study.

Cultivation of spring barley

Spring barley is well adapted for most well drained soils and is cultivated throughout the country thanks to its cold hardiness. Barley is mainly used as animal feed. The nitrogen demand is less than for wheat and is most often added in combination with sowing (Weidow, 1999). Standard yield for barley is approximately 4.4 tons in the region of Mälaren (Statistics Sweden, 2000a).

The seed required for one hectare is normally 180 kg per hectare (Odal, undated). The seed production was taken into account through subtraction of the amount of seed needed, from the total yield.

As many of the emissions factors used for barley are the same as for wheat, references and background descriptions of what figures to use in the inventory are found in the chapter Cultivation of winter wheat.

Tractor operations

In Table 15, fuel consumption and discharge of combustion emissions are shown when performing field operations in conventional cultivation of barley. The fuel consumption for combi-drilling (when sowing and fertilising are made in the same field operation) was set to be 10% higher than for only the sowing operation, a figure based on measurement on different field operations (Norén et al., 1999). Emission data for harvesting were based on data for ploughing.

Table 15. Fuel consumption and discharge of tractor emissions when cultivating barley in the conventional scenario

Operation	Fuel consumption (l/ha)	CO (g/ha)	NO (g/ha)	HC (g/ha)
Stubble cultivation	5.41	14.6	143	5.68
Ploughing	16.4	49.3	573	17.0
Harrowing (3 times)	16.1	23.9	511	9.0
Sowing & fertilising	3.82	14.5	129	4.62
Spraying	1.36	5.2	45.3	1.6
Harvesting	15	45.1	524	15.5

Emissions of CO_2 are defined as 74.6 g/MJ and emissions of SO_2 as 0.0935 g/MJ.

Fertilisation

General assumption: An equal amount of plant-available nitrogen (as ammonium and nitrate) was applied as fertiliser in both scenarios; i.e. 80 kg of nitrogen per hectare. The amount of phosphorus applied was in the same order as the amount of phosphorus removed by the crop. No potassium fertilisers were used, which is a common practice in the region of Mälaren (Linder, pers. com.).

Conventional scenario:

Nitrogen and phosphorus were applied through combi-drilling. 295 kg per hectare of the fertiliser product Hydro NP Sulphur 27-5 was added in the conventional scenario; i.e. 80 kg of nitrogen, 15 kg of phosphorus and 8.8 kg of sulphur.

Urine separating scenario:

In the scenario using human urine as fertiliser, a first application of 110 kg Hydro NP Sulphur 27-5 was added through combi-drilling; i.e. 30 kg of nitrogen, 5.5 kg of phosphorus and 3.3 kg of sulphur per hectare. Thereafter 24 tons of urine were spread with a content of 55 kg of nitrogen (of which 50 kg were easily available nitrogen), 5.6 kg of phosphorus, 17 kg of potassium and 4.6 kg of sulphur.

Pesticides

According to Linder (pers. com.), fungicide is normally not needed in barley production. One herbicide treatment yearly with Duplosan Super and insecticide treatment with Pirimor every fourth year is a common practice in the region as well as stubble treatment with Roundup Bio every seventh year (Table 16).

Table 16. Typical yearly use of pesticides in barley production in the region studied (Linder, pers. com.)

Product name	Dose rate per hectare (l)	Active substance (per l)	Total amount of active substances added yearly
Duplosan Super	2.0	310 g dichlorprop-p + 160 g MCPA + 130 g mecoprop-p	620 g dichlorprop-p + 320 g MCPA + 260 g mecoprop-p
Pirimor	O.15/4	500 g pirimicarb	19 g pirimicarb
Roundup Bio	3.5/7	360 g glyphosate	180 g glyphosate

N-emissions

Ammonia emissions

Volatilisation of ammonia from urine when using good techniques was set to 5% of the total NH_4-N in urine, i.e. 2.5 kg per hectare. When the mineral fertiliser is covered by 7-8 cm of soil, e.g. through combi-drilling, the ammonia emissions could be negligible (Välimaa & Stadig, 1998). Therefore no NH_3–emissions from mineral fertilisers were assumed here.

The ammonia emissions through the leaves were set to 1.5 kg NH_3-N per hectare and year in both scenarios (Välimaa & Stadig, 1998).

Nitrate leaching

The leaching of nitrate under current conditions was calculated to be 9 kg per hectare for both fertilisers (Jordbruksverket, 1999b).

NOx -emissions

Emissions of NO_X from arable land were estimated to 0.1% of the total nitrogen applied (Svensson et al., 1999).

N_2O-emissions

The emission figure used here was 1.25% N_2O-N losses of the total nitrogen applied (IPCC, 1997). The same emission factor for urine as for mineral fertilisers was assumed. However, ammonia volatilisation was taken into account through subtraction of the amount of N volatilised from the total N applied.

Used factors for indirect N_2O emissions were 0.01 kg N_2O-N/ kg NO_X-N and NH_3-N emitted. The emission factor used per kg N leaching was 0.025 N_2O-N/ kg N (IPCC, 1997).

P emissions

0.5 kg of phosphorus per hectare, independent on cultivated crop and fertiliser products was assumed.

Soil compaction

The yield losses caused by soil compaction were calculated using an empirical model (Arvidsson & Håkansson, 1991). As in the alternative with wheat production, the yield reduction, when spreading urine with a tanker, expressed in percent of one year’s yield was 0.9%, when including the effect in both the topsoil and the upper layer of the subsoil. The persistent effects in the deeper part of the subsoil were calculated to 0.005% yearly; i.e. 0.5% if the reduction during 100 years should be addressed to the crop under study.

Effect of wheel traffic in the growing crop

Wheel traffic in a growing crop will partly damage the crop. Results from spraying operations indicate that the damage may be in the magnitude of 1% in spring barley (Jordbruksverket, 2000), which also was the figure used here.

Yield

The yield reached was set to 4400 kg per hectare in the conventional scenario and 4150 kg in the urine-separating scenario, i.e. the yield in the urine-separating scenario was 94% of the yield in the conventional scenario. The lower yield in the urine-separating scenario was due to higher ammonia-losses (3.1%), soil compaction (1.4%) and the wheel traffic (1%).

In the yield stated in the urine-separating scenario, also the future 100 years effects from the permanent sub-soil compaction caused during this year were included through an accumulative yield reduction.

Transports

Transport distances used in this study was as following:

- Fertiliser products from production plant in Köping: 100 km
- Phosphorus fertiliser product from Western Europe to Köping by ship: 1500 km
- Precipitation chemicals from production plant in southern Sweden to Mälardalen: 600 km
- Urine from households to the farm: 10 km
- Urine from storage on the farm to the field: 1 km
- Other transports between farm centre and field: 1 km

Calculations on transports between farm centre and field are found in Appendix 7 together with emission factors on truck and ship transports. The energy consumption for the truck collecting the urine was set to 1.2 MJ per ton and km (Sonesson, 1996). Filling the truck was set to 13 MJ per ton according to references in Jönsson et al. (2000).

Electricity

Data used for the electricity was based on the Swedish average mix 1999 (Table 17). In a sensitivity analysis, the approach of marginal production of electricity was used, i.e. the avoided use of electricity in the urine-separating scenario was considered as marginal electrical production (Appendix 8).

Table 17. Composition of the Swedish electricity mix 1999 (Uppenberg et al., 2001)

Composition	% of delivered electricity
Hydro power	48.2
Nuclear power	44.3
Wind power	0.23
Combined plants (oil)	1.33
Combined plants (coal)	2.43
Combined plants (natural gas)	0.47
Combined plants (bio fuel)	2.81
Oil condensed	0.2

Aggregated data on emissions are found in Appendix 8.

Water and wastewater treatment

An issue of importance for this study is whether the system with urine separation should be looked upon as an additional part of a conventional system, and thus been given a marginal effect, or as a system that will replace the treatment plant. In this scenario study, the separation of urine was regarded as an additional facility, i.e. the WWTP would occur regardless of the urine separation system, and the other wastewater fractions are assumed to be treated in the WWTP.

How much energy and chemicals will then be saved in the WWTP due to a urine-separating system? Installation of source-separating toilets will decrease the amount of water required for the flushing function. Thus, the need for treatment, distribution and pumping of the drinking water and wastewater will be affected. A decreased amount of phosphorus entering the WWTP will affect the amount of precipitation chemicals required, and therefore affecting the production and transportation of precipitation chemicals. According to Bengtsson et al. (1997), the use of precipitation chemicals can be directly related to the flow of phosphorus entering the WWTP, and the consumption of energy can be directly related to the volumes treated.

Energy

According to studies performed at three different sites in Sweden, the amount of water saved due to urine-separation was in average 8 litres per litre urine mixture (Jönsson et al., 1999). This figure was used here for calculation of the avoided use of energy and avoided emissions in the system.

The drinking water system

In a case study of different wastewater systems in Kronan, Luleå, performed by Bengtsson et al. (1997), the electricity required for treatment and distribution was calculated to 2.7 MJ per m^3. In another LCA-study of drinking water in a plant in Göteborg (Wallén, 1999), the energy required was in total 4.4 MJ per m^3 drinking water, including both fossil fuel and electricity. The electricity demand was 3.1 MJ, a figure higher than was reported by Bengtsson et al. (1997). Here, the figure from Göteborg was used. Approximately 0.5 MJ of the total use of 1.3 MJ of fossil fuel was related to the heating of the plant and was not included in this study, as the need for heating the buildings will persist.

The wastewater system

Bengtsson et al. (1997) report figures on pumping and treating the wastewater. When the heating of the WWTP was excluded (assuming that a smaller amount of incoming water will not affect the heating), the electricity required was 1 MJ per m^3.

In the ORWARE model, the energy required in the WWTP is related to the number of person equivalent (p.e.) connected (Dalemo, 1996). Using the example of Uppsala (140 900 p.e. connected) gives a figure of in total 0.8 MJ/m^3 water handled in the treatment plant. Approximately 45% of the energy requirement is related to the aeration. Additional 0.8 MJ/m^3 is used as a figure for electricity consumption at the pump stations in Uppsala. According to a comparison between different sewage systems by Kärrman (1995), the energy required for the pumping is in general in the magnitude of 0.4 MJ/m^3. As the data from Kronan seem to be in accordance with other data reported, the figures reported by Bengtsson et al. (1997) were used here.

In the region of Mälaren, the largest treatment plants have extended their treatment to include also nitrogen removal. The additional energy requirement for this is not included in the figures mentioned above. According to Dalemo (1996), the electricity needed for aeration during nitrogen removal is approximately 18 MJ/kg N reduced. Balmér et al. (2002) report a figure of 10.3 MJ/kg N reduced. 16 MJ is estimated for nitrification, but 5.7 MJ can be recovered. 10.3 MJ per kg N reduced was the figure used here and 40% of the nitrogen entering the WWTP was assumed to be denitrified.

Chemicals

A decreased amount of phosphorus entering the WWTP will affect the amount of precipitation chemicals needed. According to figures from Uppsala, approximately 20 kg of chemicals, mostly Pix 111, are used per kg P. In the case study of Kronan, 24 kg of precipitation chemicals per kg P was reported. Here, 20 kg was assumed, and the emissions and use of energy related to chemicals are found in Appendix 9.

A minor amount of precipitation chemicals will be saved due to a less requirement of drinking water. In this study, only the avoided use of electricity when saving drinking water was considered, as this was included in the figure included in the figure by Wallén (1999) used here.

Emissions to water and air

Due to the separated system, the emissions to water of phosphorus and nitrogen will decrease. Here, 60% of the nitrogen in incoming water to the WWTP was assumed to be removed, and hence 40% were emitted into the water. The reduction of phosphorus in the WWTP was set to 95%.

No emissions of ammonia or methane from the processes in the treatment plant were accounted for. Based on measurements of Swedish WWTP (Naturvårdsverket, 1994b), 0.15% of the nitrogen in incoming water is estimated to disappear as nitrous oxide (N_2O), and this figure was used here.

Sludge handling

Throughout the 1980's and 1990's, about 30-40% of the Swedish sludge production was used in agriculture. During 1998 the corresponding figure was 26% in Mälardalen (Statistics Sweden, 2000c). Landfilling was however the most common way of disposing the sludge. Due to the coming ban on landfilling of organic wastes from year 2005, and the sludge-boycott from the food companies, other outlets for sewage sludge must be found. Land reclamation, production of soil conditioners, energy recovery by incineration, and use as fertiliser in silviculture and forestry have been proposed (Tideström, 2000). Sludge based soil products can be used on reclaimed land, parks, golf courses and as a protective layers for final covering of landfills. Currently, 10-15% of the total Swedish sludge production is used for these purposes, but the potential for increasing the proportion is considerable (Tideström et al., 2000).

As this study consider a nearby future (within 5-10 years), probably without possibilities on using landfills and farming land for disposal purposes, sludge was here assumed to be used mainly for production of soil products. However, in the long run, plants for incineration and recovering of phosphorus may be an alternative. The chosen alternative could therefore be seen as an intermediate stage. No additional environmental load was assumed to be related to this handling. The level of phosphorus leaching from the soil could differ between different soil products with different content of phosphorus, but this was not included here.

Impact from capital goods

In most LCAs performed, capital goods are excluded. However, according to Weidema et al. (1995) the energy requirement from capital goods used in agriculture is approximately 15% of the total fuel consumption in the production. In this figure repair work is included, which could be approximately 50% of the energy requirement in new equipment.

Impacts related to the production of capital goods considered here was the construction phase of the pipes and storage tanks for urine and energy consumption related to the production of the urine spreader.

The weight of a spreader with the capacity of 10 tons is approximately 5 tons (de Toro, pers. com.). The estimated energy requirement when producing a modern combine is 78 MJ/kg according to Weidema et al. (1995). Using this energy value on a spreader, and assuming that the spreader during its lifetime will be used on 1500 hectare, gives a figure of 260 MJ per hectare and year.

The equipment required for collecting the urine on household level could be of different materials. Whether the urine is collected from a single house, or from a block of house will further affect how the collection will be worked out. Here, data from a storage tank (2.8 m^3) of concrete was used, a tank that is available on the market for collecting urine or closet water. In Table 18, the energy requirement related to the production of the storage tank and excavating is given. The amount of material required was taken from the company Tranås Cement, and the energy required for producing the material was taken from Tillman et al. (1996).

Table 18. The amount of material and machine work required for production of a system where urine is collected in a 2.8 m^3 storage tank in concrete

Material and machine work performed	Amount required	Electricity (MJ)	Fossil fuel (MJ)
Concrete (kg)	2464	185	1907
Reinforcing bars (kg)	36	0	768
Macadam (m^3)	0.9	10	8
Plastic pipes (kg)	25	115	1998
Excavating, tank (m^3)	4		24
Excavating, pipes (m)	8		75
Total		310	4780

The time for writing off the investments was set to 30 years for the storage tanks and the pipes. Using these figures on the data in Table 20 gives that 3.7 MJ of electricity and 57 MJ of fossil fuel could be dedicated to 1 m^3 of urine assuming that the full tank was emptied once a year.

Further and more detailed figures on the impact from storage facilities are found in Appendix 10.

For the storage on the farm, it was assumed that the urine was stored in a concrete tank holding 800 m^3 with a cover in plastic. According to drawings from the company Abetong (www), the weight of an 800 m^3 storage tank is 42.5 tons and 3% of the weight consists of reinforcing bars. It was further assumed that the cover in PVC weights 250 kg and that the storage will be used during 30 years. Data for the environmental impact from the production of PVC was taken from a compilation in Finnveden et al. (1996).

Sensitivities and uncertainties

In a sensitivity analysis, the following changes in the assumptions were made and evaluated according to its influence on the primary energy use.

- Transport distances for the urine mixture were changed from 10 km to 2.5 and 40 km.
- Lower yield. 10% lower yield in the wheat production system and 15% lower yield in the barley production system were assumed.
- Urine storage in two parallel plastic tanks on household level with 1-year storage capacity for each (see Table 19).
- Production of capital goods was not included in the system.
- No avoided burdens in the wastewater system were accounted for.

In Table 19, the difference in energy use due to the choice of material is illustrated. The data on the bigger concrete tank as well as the plastic tank were taken from Tillman et al. (1996). In the sensitivity analysis, figures on the 2 m^3 plastic tank were used.

Table 19. Energy required for production of storage tanks and pipes (MJ per m^3 storage)

Type of storage tank	Electricity (MJ)	Fossil fuel (MJ)
Concrete, 2.8 m^3	111	1707
Concrete, 27 m^3	81	1889
Plastic, 2 m^3	264	5195

For the contribution to global warming, the following aspects were changed.

- Marginal production of electricity was used, which means that the avoided use of electricity was considered as produced from coal.

FLOWS OF PLANT NUTRIENTS AND CADMIUM

Wheat

The flows of nitrogen, phosphorus and cadmium through the soil and plant systems per hectare and year are shown in Table 20. The calculations were based on a yield of 5650 kg in the urine-separating scenario, i.e. also the long-term effects from soil compaction were considered.

Table 20. Flows of N, P and Cd per hectare and year in the plant and soil systems when cultivating wheat according to the two scenarios

	Conventional			Urine-separating		
	N (kg)	P (kg)	Cd (mg)	N (kg)	P (kg)	Cd (mg)
Input						
Mineral fertiliser [a)]	145	19	171	73	11	99
Urine [b)]				79	7.9	20
Deposition [c)]	5.5	0.3	700	5.5	0.3	700
Total input	*151*	*19*	*871*	*157*	*19*	*819*
Removal						
Crops (kernel) [d)]	102	19	224	96	18	211
Leaching [e)]	10	0.5	400	10	0.5	400
Air emissions (N_2 excluded)	4.5			7.5		
Total removal	*117*	*20*	*624*	*114*	*18*	*611*
Accumulation	**34**	**-0.2**	**247**	**44**	**1.2**	**208**

a) The expected content of cadmium (9 mg/kg P) in the fertiliser was taken from Odal (undated) stating that Cd content in P20 is between 6-12 mg/kg P.
b) Cadmium content in urine mixture was set to 0.58 mg/m^3 (Vinnerås, 2001).
c) Deposition of cadmium was taken from Jansson (2002). Data on deposition of P from Wolgast (1994).
d) Concentration of Cd in winter wheat (0.044 mg/kg dw) based on data from Eriksson et al. (2000).
e) Data on Cd in soil solution refereed to in Jansson (2002).

The surplus of nitrogen was higher in the agricultural system using urine due to the fertilisation strategy chosen and the lower yield. In both systems, most of the surplus-N was likely to be emitted as N_2. The system using urine also had a surplus of phosphorus, but the difference was small between the two fertilising strategies. The accumulation of cadmium in the soil was slightly less in the system using urine. Deposition and leaching were the most important factors determining the accumulation.

Barley

The flows of nitrogen, phosphorus and cadmium through the soil and plant systems are shown in Table 21. The calculations were based on a yield of 4150 kg in the urine-separating scenario.

Table 21. Flows of N, P and Cd per hectare and year in the plant and soil systems when cultivating barley according to the two scenarios

	Conventional			Urine-separating		
	N (kg)	P (kg)	Cd (mg)	N (kg)	P (kg)	Cd (mg)
Input						
Fertiliser [a)]	80	15	38	30	5.5	14
Urine [b)]				55	5.5	14
Deposition [c)]	5.5	0.3	700	5.5	0.3	700
Total input	*86*	*15*	*738*	*91*	*11*	*728*
Removal						
Crop (kernel) [d)]	70	15	71	66	14	67
Leaching [e)]	9	0.5	400	9	0.5	400
Air emissions (N_2 excluded)	3.5			5.2		
Total removal	*83*	*16*	*471*	*81*	*15*	*467*
Accumulation	**3**	**-0.2**	**266**	**10**	**-3**	**261**

a) The expected content of cadmium (2.5 mg/kg P) from Odal (undated) stating that Cd content in NP 27-5 lays between 0-5 mg/kg P.
b) Cadmium content in urine mixture was set to 0.58 mg/m^3 (Vinnerås, 2001).
c) Deposition of cadmium from Jansson (2002). Data on deposition of P from Wolgast (1994).
d) Concentration of Cd in barley (0.019 mg/kg dw) based on data from Eriksson et al. (2000).
e) Data on Cd in soil solution refereed to in Jansson (2002).

The surplus of nitrogen was higher in the agricultural system using urine due to the fertilisation strategy chosen and the lower yield. The deficit of phosphorus was however higher in the system using urine, as the content of phosphorus in relation to nitrogen was lower in urine compared with the commercial fertiliser product chosen. The accumulation of cadmium in the soil was of the same magnitude in the two systems.

IMPACT ASSESSMENT

First in this chapter, the different characterisation factors used are presented. The result from the impact assessment is then presented in different impact categories. No valuation methods were used due their limitations to be applied in this specific case. Two important effects when discussing wastewater systems and use of plant nutrients in sewage products are reduction of nutrient emissions to water and recycling of plant nutrient resources, e.g. phosphorus. Neither the EPS-method nor the Eco-indicator 99 takes the negative consequences of nitrogen discharge to water into account. This is however a prioritised Swedish environmental goal. Another valuation method, ET-long, do not take depletion of e.g. phosphorus into account.

The terminology used here was *conventional (scenario)* for the system using only mineral fertiliser products, and *urine-separating (scenario)* for the system using also source-separated human urine as fertiliser.

Characterisation factors

There exist several methods for weighting the many emissions resulting from the inventory. In the following chapter, the weighting factors used for the characterisation in this study are shortly described.

Global warming

An increasing amount of greenhouse gases in the atmosphere leads to an increasing absorption of heat radiation energy. Emissions of CO_2 are considered to give the most important contribution to the global warming, but especially in agricultural systems other gases such as CH_4 and N_2O may contribute more than CO_2 (Cederberg, 1998). The global warming potentials for CH_4 and N_2O relative to CO_2 are shown in Table 22. In this study the time-perspective of 100 years was used.

Table 22. Global Warming Potentials (GWP) as CO_2–equivalents for different trace gases and time-frames (IPCC, 2001)

Trace gas	GWP, 20 years	GWP, 100 years	GWP, 500 years
Carbon dioxide, CO_2	1	1	1
Methane, CH_4	62	23	7
Nitrous oxide, N_2O	275	296	156

Eutrophication

Eutrophication includes both impacts on the terrestrial and the aquatic systems. An increased input of nutrients into an aquatic system may lead to an increasing production of biomass. The decomposition of the biomass will then require oxygen. As different aquatic systems are limited by different nutrients, a general characterisation (as with Global Warming Potential) raises questions. How to account for nitrogen emissions to air is another problematic issue to handle, as some, but not all, of the nitrogen will reach the aquatic system (Lindfors et al., 1995). In Table 25, a maximum scenario of eutrophication is presented, assuming that *all* substances mentioned will contribute to eutrophication. The weighting factors recommended by for example Nordic Guidelines on Life-cycle Assessment (Lindfors et al., 1995) does not account for the oxygen demand of nitrification, i.e. when ammonia to air and ammonium to water are oxidised to nitrate in the receiving water. Therefore, higher weighting factors for oxidation of ammonia and ammonium have been proposed (Kärrman & Jönsson, 2001). These higher weighting factors, marked with bold numbers in Table 23, are used here. In parentheses are the factors used in Lindfors et al. (1995).

Table 23. Weighting factors for eutrophication expressed as a maximum scenario (Lindfors el al., 1995). The factors marked with bold numbers are suggestions from Kärrman & Jönsson (2001)

Substance	Maximum (g O_2 per g)
N to air	20
NO_x to air	6
NH_3 to air	**19.8** (16)
N to water	20
NO_3 to water	4,4
NH_4 to water	**18.6** (15)
P to water	140
PO_4^{3-}	46
COD	1

Acidification

In Table 26, two scenarios for acidification are presented. The differences consider the nitrogen compounds, depending on whether the anion is leached out from the system or not (Lindfors et al., 1995). In reality, the contribution will be something between the two extremes. The amount of nitrogen leaching compared to the input is for example 15% in Scandinavia (Lindfors et al., 1995). In this study, the result is presented as a maximum scenario assuming that all the substances in Table 24 contributed to acidification. The contribution from different substances is however discussed.

Table 24. Weighting factors for acidification for two scenarios, min and max (Lindfors et al., 1995).

Substance	Min [mol H+/g]	Max [mol H+/g]
SO_2	0.031	0.031
HCl	0.027	0.027
NO_x	0	0.022
NH_3	0	0.059

Photo-oxidant formation

Photo-oxidant formation is the formation of reactive compounds, e.g. ozone, by certain air pollutants in the presence of sunlight. Ozone may affect human health, ecosystem and damage crops (Guinée, 2001). The background level of NO_X is important for the ozone production, as NO_X acts as a catalysator. In Guinée (2001), POCP (photochemical ozone creation potentials) is presented for two levels of NO_X; high and low. In the later case, only carbon monoxide and methane of the specific emissions considered in this report are contributing to photo-oxidant formation. In the first case (high levels of NO_X), also nitrogen dioxide and sulphur dioxide contribute. The figures presented as POCP for low levels of NO_X in kg ethylene equivalents per kg emitted substance is 0.04 for CO and 0.007 for CH_4.

Energy use

Energy usage was divided into fossil fuel and electricity. When comparing the total energy required for different activities, the electricity was recalculated to its primary energy carrier. Hereby, the amount of electricity used was multiplied with a factor 2.05 (Arnäs et al., 1997).

Wheat

The use of fossil fuel was slightly higher in the scenario using human urine as fertiliser (Figure 6). A more apparent difference in energy use was the use of electricity, due to the avoided usage of electricity in the urine-separating scenario.

The electricity used in the conventional scenario was entirely related to the production of mineral fertiliser. The production and transportation of mineral fertiliser also represented around 75% of the total fossil fuel required.

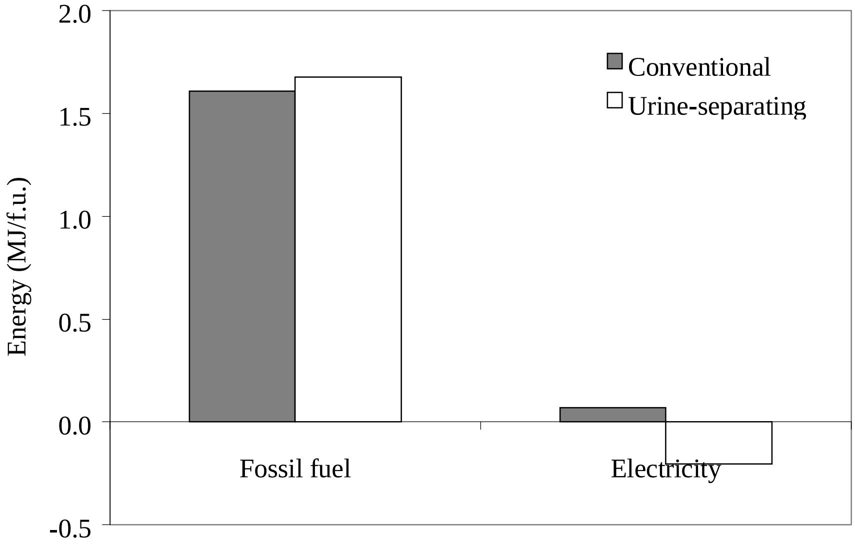

Figure 6. Energy usage expressed as total use of fossil fuel and electricity when cultivating wheat according to the two scenarios.

In both scenarios, production and transportation of mineral fertiliser contributed most to the energy consumption (Figure 7). The difference between the scenarios was however considerable, due to the fact that only half of the quantity of mineral fertiliser was used in the urine-separating scenario compared to the conventional scenario. The differences in energy consumption related to field operations between the two scenarios were related to the spreading of urine (included in field operations) and due to the lower yield in the urine-separating scenario.

The energy use related to the production of capital goods in the urine-separating scenario contributed considerable to the total energy usage, and was in the same magnitude as field operations.

The avoided use of electricity in the urine-separating scenario was due to the less quantity of drinking water necessary to provide, as well as the avoided need for pumping and treating the wastewater. The avoided use of precipitation chemicals was an important factor, but also the avoided need for de-nitrification and pumping of the wastewater contributed considerably.

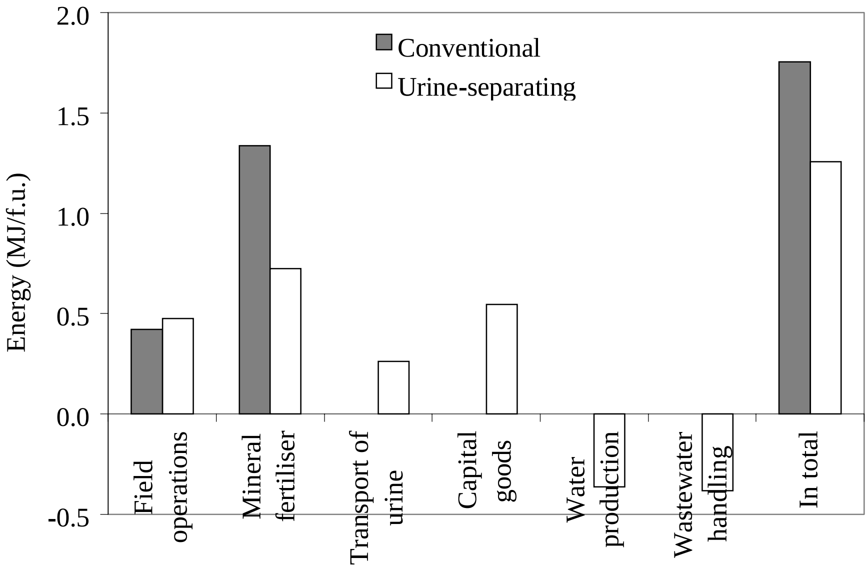

Figure 7. Total energy usage expressed as primary energy partitioned into different activities when cultivating wheat according to the two scenarios.

Barley

As in the wheat production, the total energy used was less in the scenario using human urine as fertiliser (Figure 8, Figure 9). The total energy used was however on a lower level compared to the wheat production system due to the smaller quantity of mineral fertiliser used per functional unit. The urine-separating scenario had a higher consumption of energy as fossil fuel, but a negative use of electricity, i.e. electricity was saved due to the separation of urine.

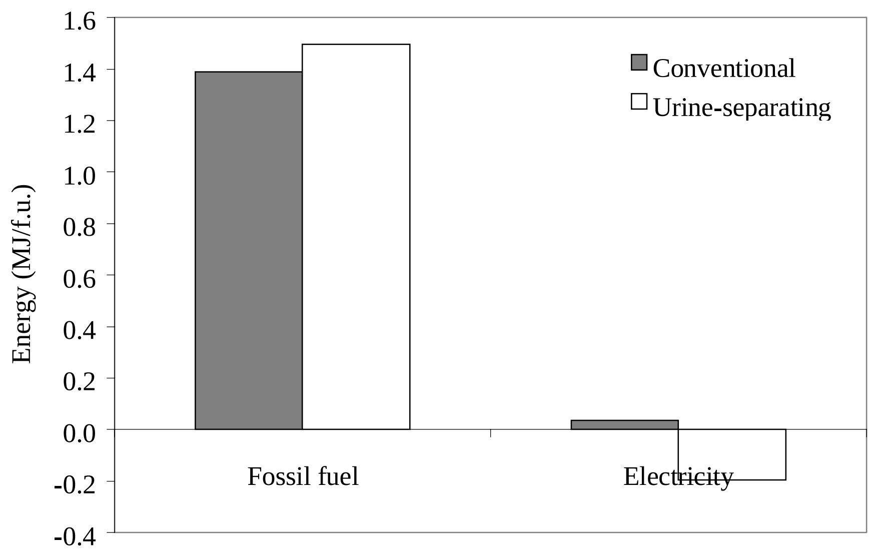

Figure 8. Energy usage expressed as total use of fossil fuel and electricity when cultivating barley according to the two scenarios.

In the conventional scenario, the highest energy use was related to production and transportation of mineral fertiliser products (Figure 9). In the urine-separating scenario, the highest energy consumption was related to the field operations followed by the energy use associated with the production of capital goods. Transportation of urine represented one fourth of the energy required in the urine-separating scenario.

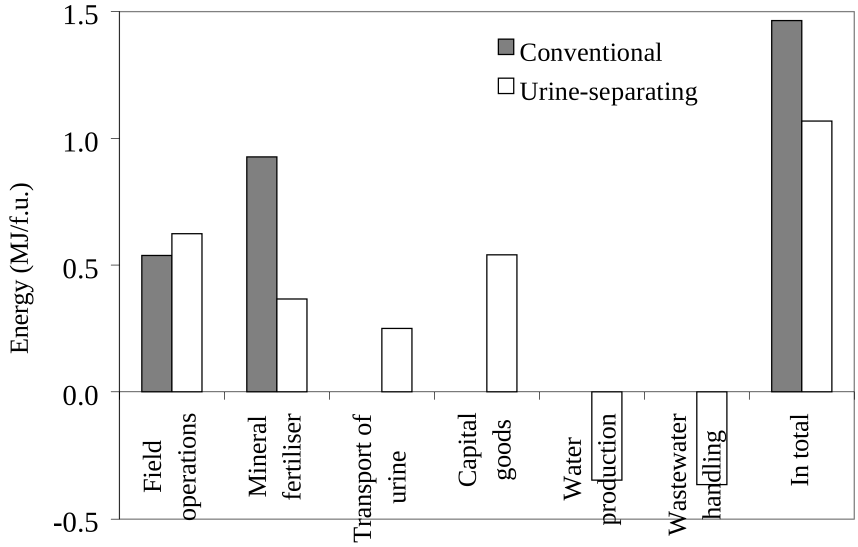

Figure 9. Total energy usage expressed as primary energy partitioned into different activities when cultivating barley according to the two scenarios.

Materials and water

The materials considered here were the use of the plant nutrients phosphorus and sulphur. The applied amount of these nutrients was compared with the removal by the crop. Considering sulphur, the amount applied via urine was the same or even slightly higher than the actual removal. However, the required amount of sulphur is higher than the removal due to e.g. leakage. The recommendation is to apply around 10 kg of sulphur per hectare in grain production (Jordbruksverket, 1999c). In the wheat and barley production systems, 6.6 and 4.6 kg sulphur respectively were applied.

The amount of phosphorus applied as urine, compared with the removal by the kernel was about 40% in both the grain production systems (Table 20 and 21).

As earlier shown (Figure 6 and 8), the use of fossil fuel was slightly higher in the systems using human urine, 4 and 8% respectively for the wheat and barley production systems respectively.

The water saving due to the urine-separating scenarios was 51 and 48 litres respectively of water per kg grain produced in the wheat and barley production systems. The water saving expressed in m^3 per hectare was 276 in the wheat production systems and 192 in the barley production system.

Environmental impacts

Global warming

Wheat

The contribution of greenhouse-gases, expressed as GWP, from the two scenarios was of the same magnitude, even though it was slightly less from the urine-separating scenario (Figure 10). For both scenarios, nitrous oxide gave the highest contribution. It was also the nitrous oxide that gave rise to the differences in contribution to GWP.

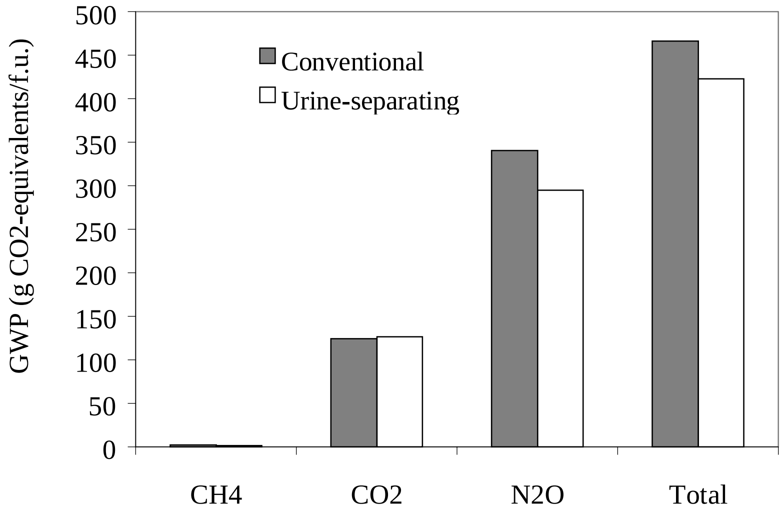

Figure 10. Contributions to Global Warming Potential when cultivating wheat according to the two scenarios.

In Figure 11, contributions to GWP from different activities are shown. Field emissions of nitrous oxide occurring from natural processes in the conversion of nitrogen, accounted for the greatest part the greenhouse-gases, followed by the emissions of nitrous oxides from the production of mineral fertiliser.

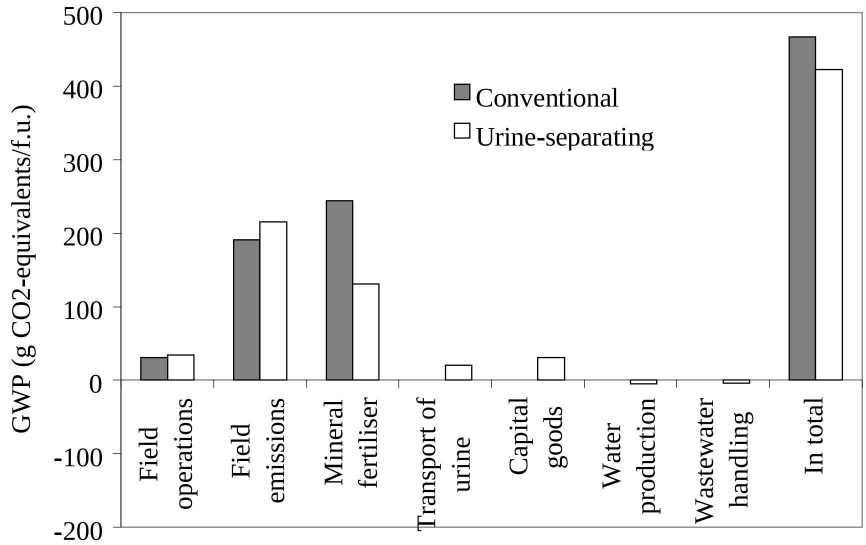

Figure 11. Contributions to Global Warming Potential from different activities when cultivating wheat according to the two scenarios.

Barley

As in the wheat production system, the conventional scenario had a slightly higher contribution of greenhouse-gases than the urine-separating scenario (Figure 12). The total contribution was however on a lower level than in the wheat production system. For both scenarios, nitrous oxide contributed most to GWP. The emissions of nitrous oxide originated mostly from field emissions followed by mineral fertiliser production (Figure 13).

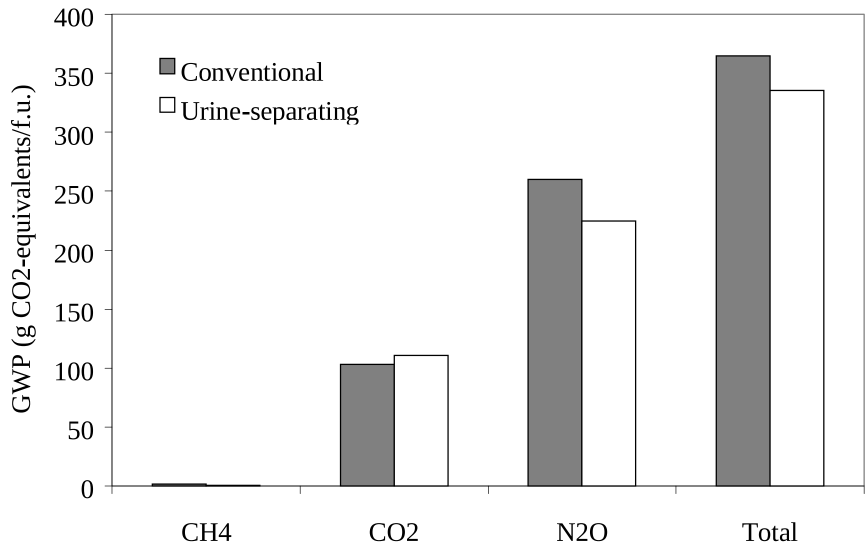

Figure 12. Contributions to Global Warming Potential when cultivating barley according to the two scenarios.

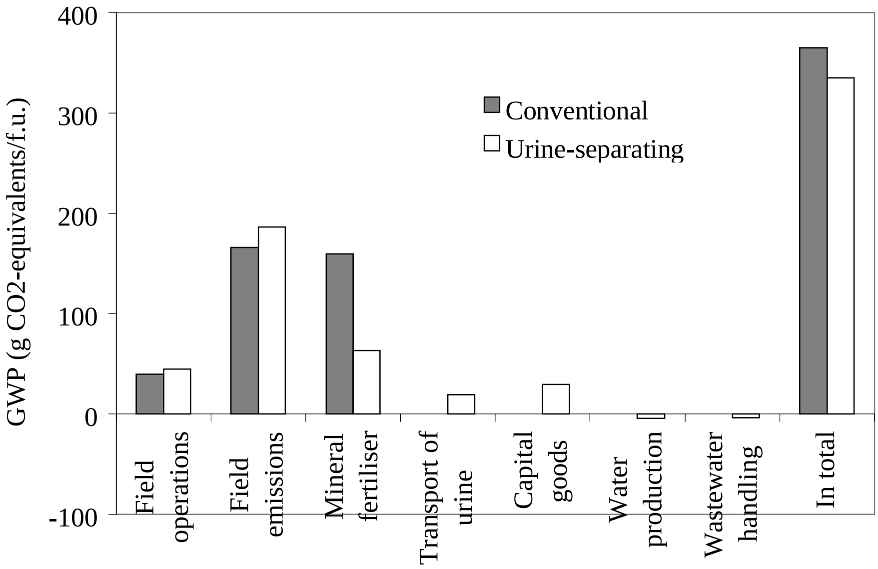

Figure 13. Contributions to Global Warming Potential from different activities when cultivating barley according to the two scenarios.

Eutrophication

Wheat

The difference in contribution to eutrophication was considerable between the two scenarios, due to the avoided emissions of eutrophying substances from the wastewater subsystem of the urine-separating scenario (Figure 14). In the urine-separating scenario, both NO_X and NH_3 to air as well as NO_3 to water was higher than in the conventional scenario. This was however more than compensated for by the avoided emissions of nitrogen from the wastewater treatment plant. The discharge of phosphorus was almost four times higher in the conventional scenario.

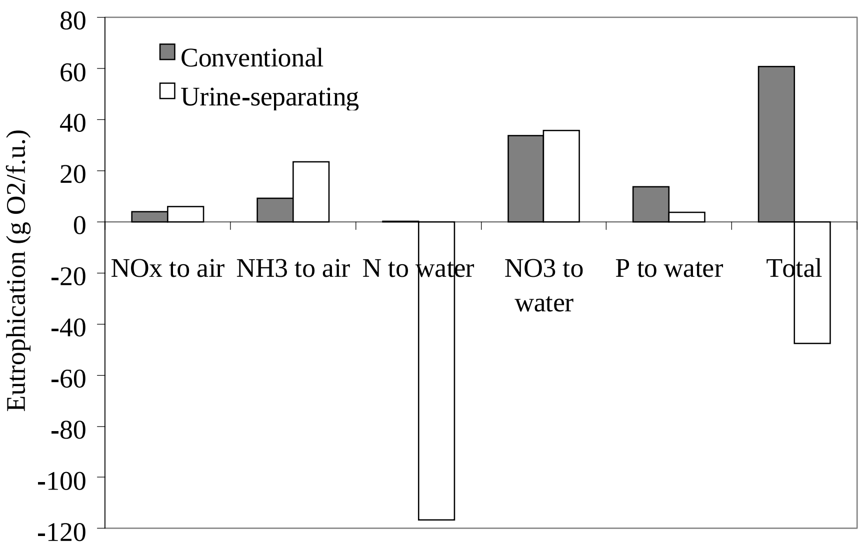

Figure 14. Contributions to eutrophication (max scenario) when cultivating wheat according to the two scenarios.

Barley

In Figure 15, contribution to eutrophication in the barley production system is shown. The main features were similar to those for wheat production. However, due to the lower yield when cultivating barley, the nitrate losses per functional unit was higher than for wheat production, despite the fact that nitrate leaching per hectare was assumed to be higher in the wheat production system.

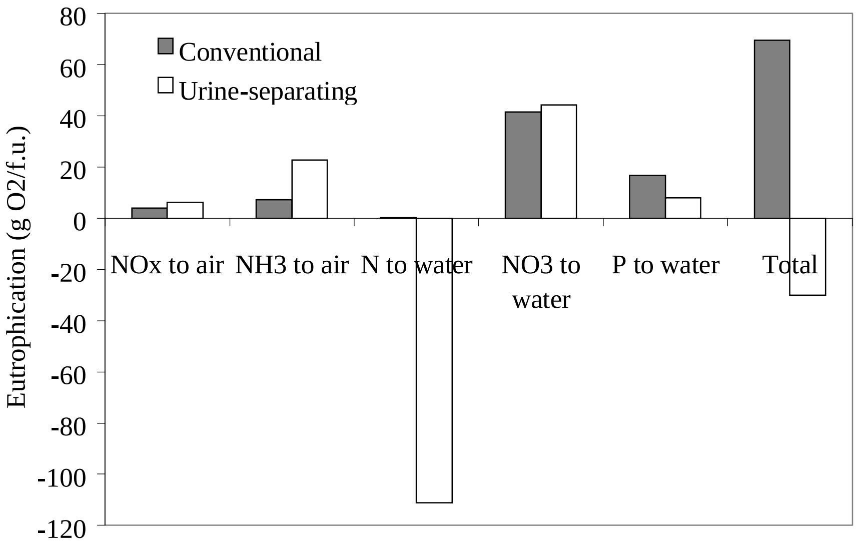

Figure 15. Contributions to eutrophication (max scenario) when cultivating barley according to the two scenarios.

Acidification

Wheat

The urine-separating scenario contributed most to acidification, expressed as a maximum scenario, through its higher emissions of NO_X and NH_3 (Figure 16). In total, the acidification was almost twice as large in the urine-separating scenario as in the conventional scenario. Regarding only the emissions of SO_2, corresponding to a minimum scenario, the conventional scenario had the highest contribution.

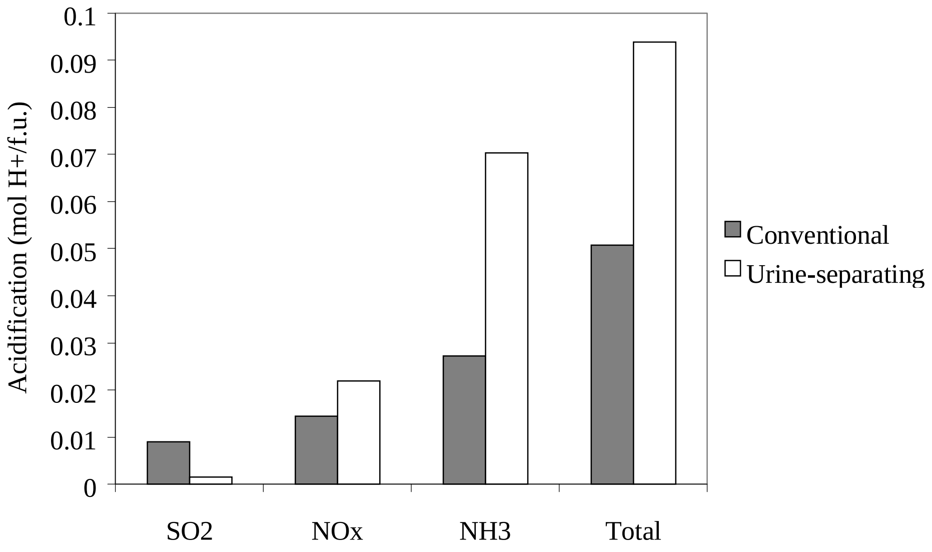

Figure 16. Contributions to acidification when cultivating wheat according to the two scenarios.

Barley

For barley, the acidification result was similar to the result of the wheat production system (Figure 17). The difference in emissions of SO_2 between the scenarios was however smaller than in the wheat production system.

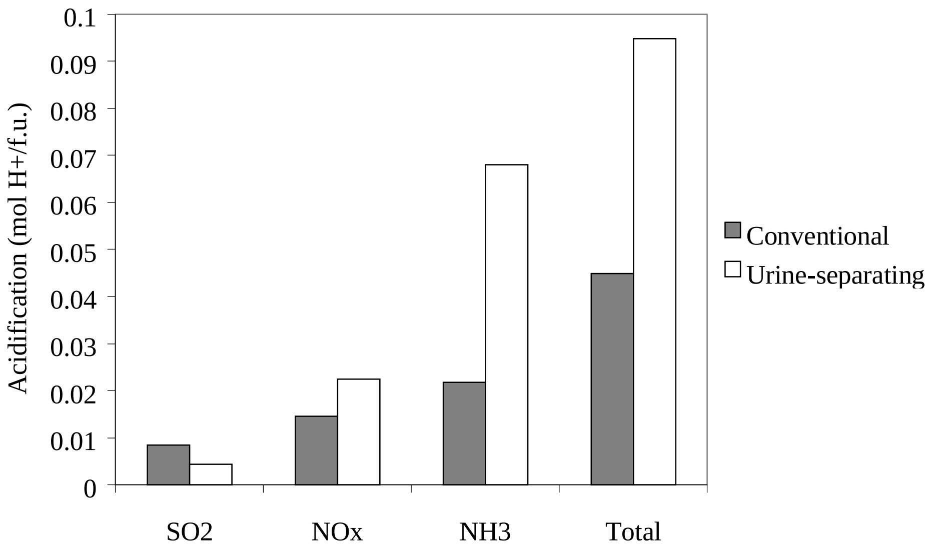

Figure 17. Contributions to acidification when cultivating barley according to the two scenarios.

Photo-oxidant formation

The contribution to photo-oxidant formation from carbon monoxide and methane was of the same magnitude for both scenarios, however slightly less for the urine-separating scenario. These results were the same for both the wheat and the barley production systems.

Table 25. Contribution to photo-oxidant formation (mg ethylene eq../kg grain)

Production systems	Conventional	Urine-separating
Wheat	0.00052	0.00044
Barley	0.00064	0.00057

Sensitivity analyses

Energy

In a sensitivity analysis, changes in the assumptions were made and evaluated according to its influence on the energy use (Figure 18 and 19). Decreasing the transport distance for urine to 2.5 km somewhat decreased the energy usage. Increasing this distance to 40 km mad the primary energy usage approximately equal for the two scenarios. A lower yield due to the fact that NH_4-N, which is the main nitrogen component of urine, could be less direct available than nitrate-N had a minor influence on the result. The way the urine was stored had a major influence on the results. If the hygienisation takes place at the household level, two parallel storage tanks are required. Here, one year of storage capacity for each tank was assumed and the time for writing them off was set to 30 years, as was the case with the concrete storage tank. With such a system, the energy required was much higher for the urine-separating scenario than for the conventional scenario. If on the other hand, capital goods were not included at all, either due to a definition of the system boundary, or due to the fact that no major changes in the investment was made, the energy used in the urine-separating scenario was considerable lower than in the conventional scenario. The rightmost bar shows the effect on the use of primary energy when no avoided burdens from the water and wastewater system were taken into account. This could be the case if no water is saving due to the installation of the urine-separating system and if the use of precipitation chemicals in the wastewater treatment plant is based on the flow rather than the amount of incoming nutrients.

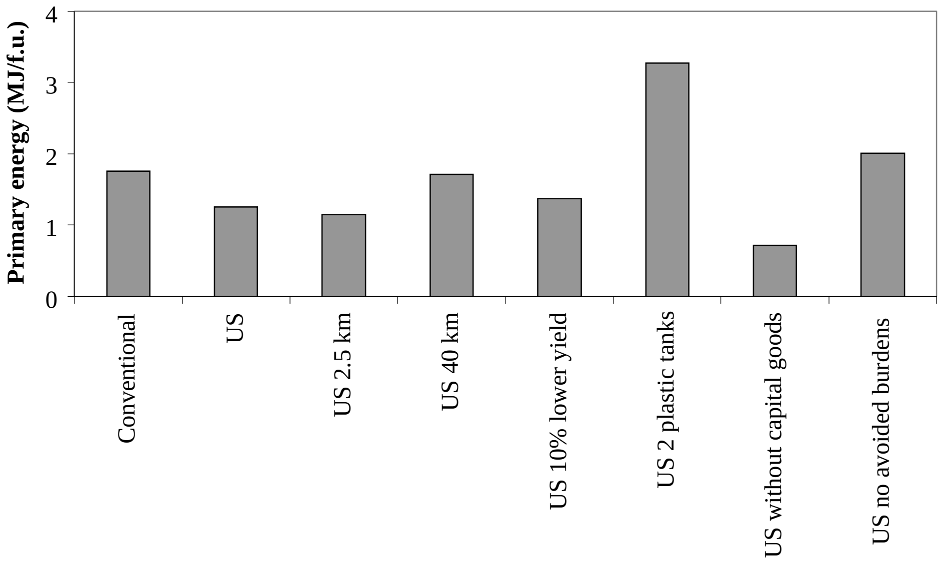

Figure 18. Primary energy usage in the wheat production system comparing the conventional and urine-separating (US) scenarios with changes according to the sensitivity analyses. The two bars to the left, show the primary energy used in the two scenarios without any changes in the assumptions. The abbreviation US stands for the urine-separating scenario.

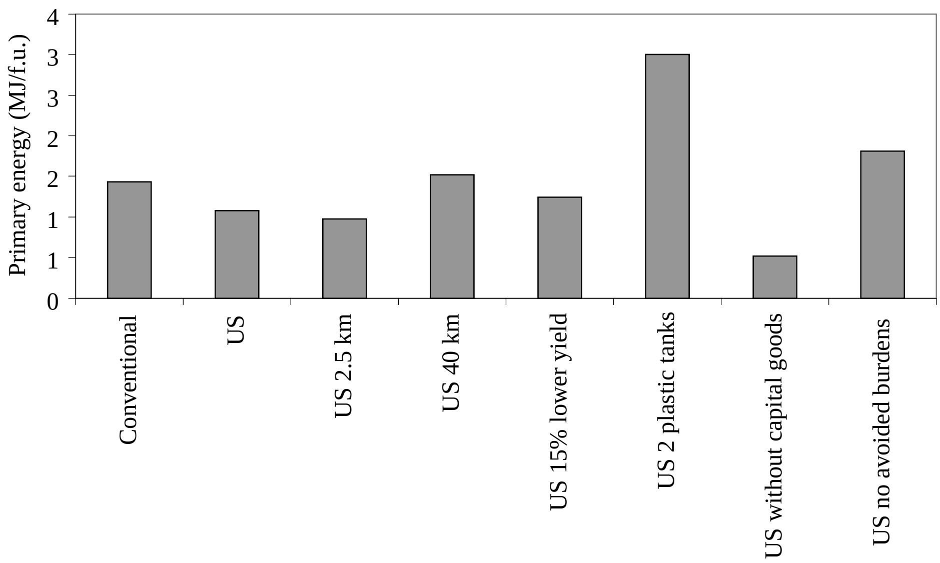

Figure 19. Primary energy usage for the barley production system comparing the conventional and urine-separating (US) scenarios, with changes according to the sensitivity analysis. The two bars to the left, show the primary energy used in the two scenarios without any changes in the assumptions. The abbreviation US stands for the urine-separating scenario.

Global warming

Emissions of CO_2 were the second largest contributor to global warming from the systems (Figure 10 and 12). However, the emission of CO_2 from electricity production highly depends on the electricity mix used. A calculation therefore was made where the saved electricity was considered as being based on marginal production of electricity, i.e. produced from coal. The result, Figure 20, shows that this change in the assumption had a noticeable impact on the emissions of CO_2, despite the minor use of electric energy compared to other forms of energy.

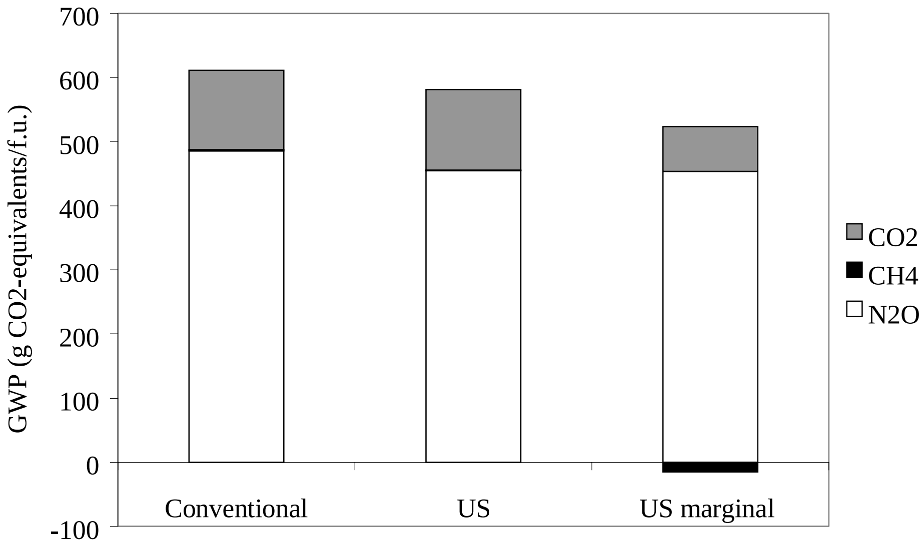

Figure 20. Contribution to Global Warming Potential from the conventional and urine-separating wheat production scenarios, compared to a urine-separating scenario using marginal production of electricity, i.e. coal condensed electricity

NORMALISATION

In normalisation, the change in environmental impact when implementing a urine-separating system in big-scale was compared with the total impact in Sweden (Table 26). The urine was here assumed to be used on 30 000 hectares, corresponding to approximately 10^6 m^3 or roughly one million users of the urine separating toilets. Figures from the wheat production system were used.

Table 26. The change in environmental impact when implementing a urine-separating system in big-scale spreading on 30 000 hectares compared to the total impact in Sweden

Emission	Total impact in Sweden (ton)	Change (in ton)	Change (in %)
GWP-gases (CO_2-equivalents)	69 000 000 [a]	-11 250	-0.02
NH_3 to air	55 450 [b]	115	0.21
SO_2 to air	58 000 [a]	-21	-0.04
N to water (after retention)	87 000 [c]	-954	-1.1
P to water	2 700 [d]	-13	-0.47

[a] Statistics Sweden, 2002b.
[b] Statistics Sweden, 2000b.
[c] Naturvårdsverket, 1997b. Figure from 1995.
[d] Naturvårdsverket, 1997a.

The normalisation shows that the most apparent change appeared when looking at the discharge of N and P to water. The change was however small compared to the total impact in

Sweden, mainly because the urine-separating scenario was compared with a conventional scenario with efficient removal of N and P.

Assuming that urine was used on 30 000 hectares in Mälardalen and comparing the total reduction of nitrogen to the contribution of nitrogen to the sea from the region of Mälaren

(10 500 tons according to Naturvårdsverket 1997b), gave a 9% regional reduction of nitrogen to sea. The discharge of phosphorus to the lake Mälaren is approximately 400 tons per year (Wallin, 2000). Comparing the change in phosphorus to water with the total load to the lake Mälaren, showed that the theoretical reduction would be 3%.

DISCUSSION

Agricultural aspects in systems analyses of wastewater management

Earlier performed environmental systems analyses of wastewater management and recycling of nutrients in Sweden take parts of the agricultural system into account, focusing on how much plant nutrients might be substitute by replacing mineral fertiliser with sewage products. This has been made either by accounting for the avoided burdens when sewage products replace mineral fertilisers (e.g. Bengtsson et al., 1997), or by expansion of the systems boundary, stating that all systems shall deliver the same amount of plant nutrients to the crop (e.g. Dalemo, 1999). In this study, compared to these previous studies, the focus was changed from a wastewater perspective to an agricultural perspective, looking upon changes in the handling of wastewater as avoided burdens. What are the advantages of such a change in focus? One advantage is related to how the information gained in the study can be further communicated. Farmers have a crucial part in the establishment of new systems aiming at recycle plant nutrients in sewage products to arable land. They might act as entrepreneurs, and the way the sewage product is handle (e.g. storage, spreading, choice of fertilised crop) will have a great influence on the environmental impact. It could therefore be pedagogical to relate environmental impact from a change to a source-separating nutrient recycling sewage system to the environmental impact from conventional production of grain. Using an agricultural perspective will also highlight how the use of a sewage product fits into the existing production, focusing upon the practical aspects a farmer needs to take into account when using different fertiliser products.

Other advantages when including agricultural production in the systems analysis are that other factors than plant nutrient substitutability, e.g. yield reduction due to e.g. soil compaction and wheel-traffic in a growing crop, can be included.

One conclusion from the study is that the main differences between the systems in this study are related to changes in the wastewater system as well as the avoided use of mineral fertiliser, and not related to other changes in the agricultural system. For a course assessment of environmental impact from urine-separating systems compared to conventional wastewater treatment, it is therefore sufficient to take only the substitutability of mineral fertiliser into account. However, for a better understanding of how an agricultural system using human urine could be shaped, a more detailed agricultural perspective gives more information and generates data useful for the communication with farmers.

Including capital goods in the system boundary?

Many environmental systems analyses performed in Sweden include only the operation phase, i.e. the environmental impact related to the production of capital goods is disregarded. This is e.g. the case for the ORWARE model used for calculating emissions, energy turnover and plant nutrient recycling related to the handling of organic waste. However, LCAs of water and wastewater systems including the production of capital goods (e.g. Tillman et al., 1996; Bengtsson et al., 1997) show that this phase is not always negligible. In wastewater systems with separate treatment of e.g. urine, the production phase may account for about 40% of the total energy required calculated as primary energy (Tillman et al., 1996). For conventional treatment of wastewater, the relative share from the production phase is smaller. Tillman et al. (1996) concluded that in area with a majority of the households being one-family houses, the energy usage related to the production phase should be included in an LCA.

In this study, additional investments in the urine-separating scenario relative to the reference scenario - conventional production of grain - were taken into account. Through this definition of the system under study, production of storage tanks, additional pipes etc. were included in the urine-separating scenario as those were considered to complement an already existing system for handling the household wastewater. Using a change-oriented perspective in this study gave interesting information on the environmental impact. Hereby, major changes in e.g. the construction of capital goods could be taken into account, without considering the construction phase in its whole.

How far can the urine be transported?

The long distance sometimes required for transporting the urine is often pointed out as a weakness with a source-separating system without further concentration of the urine mixture. Therefore, studies aiming at concentrate the urine have been carried out (e.g. Johansson & Hellström, 1999; Lind et al., 2000). How far the urine can be transported before the total energy for the urine-separating scenario exceeds the conventional scenario will however depend on assumptions made in the goal and scope definition and in the inventory. Including production of capital goods in this study contributed considerably to the primary energy used in the urine-separating scenario as well as assumptions that the production of drinking water and the treatment of the wastewater were affected. Here, it was assumed that the installation of urine-separating toilets would decrease the total water consumption as well as the requirement for pumping and treating the wastewater. Using the assumption given in this study allowed transportation of 42.5 km (one way) in the wheat production system, before the total energy usage of the urine-separating scenario exceeded the one of the conventional scenario. If the production phase was excluded the urine could be transported 78 km. If instead the reduced requirement of electricity due to saving of the water plus denitrification was excluded, the energy balance was negative already for the base urine-separating scenario.

These examples show that, the transport distance the urine mixture could be transported without getting a negative energy balance to a great extent depends on the underlying assumptions. Concentration of urine as an additional step in order to reduce the energy usage required for transportation should therefore be compared to the infra-structural changes needed for this handling, including the construction phase. There might however be economical reasons for systems where the urine is further concentrated.

Site-specific versus general data for eutrophication and acidification

In this study, placed in Mälardalen, in the eastern part of Sweden, mainly site-specific data were used. However, when assessing the environmental impact from the systems, maximum scenarios were used for eutrophication and acidification to give the results a more general character. Eutrophication is an impact category where a maximum scenario could be used also in this regional context as both reduction of nitrogen to sea as well as phosphorus to lakes are prioritised environmental goal. Independent of which plant nutrient is limiting, the urine-separating scenario turned out to be better in this aspect.

For acidification, the situation is somewhat different as only part of the total nitrogen-input is leaching out from the soil. Here, the use of a maximum scenario is misleading, as it will overestimate the contribution from nitrogen compounds to acidification. Depending on whether a maximum or a minimum scenario is used, different solutions are pointed out as most environmentally favourable. In the maximum scenario, the urine-separating scenario turned out to have the highest contribution, whereas in the minimum scenario, the conventional scenario contributed the most.

How to include chemical risks related to wastewater handling?

Issues of high interest in the Swedish debate concerning future wastewater systems are the risks associated with pharmaceuticals and organic pollutants (Palmquist, 2001). These aspects are not easily dealt with in traditional LCAs. Other existing methods of risk assessment are also insufficient as they lack important information on long-term effects and interaction between chemicals as they describe only one single substance at the time (Palmquist, 2001). For decision-making however, these aspects have to be taken into account. One way to include these aspects is to make qualitative comparisons between different alternatives, highlighting barriers of different characters, e.g. physical, functional, technical barriers. The indicated risks with pharmaceuticals and other organic pollutants are today identified as related to effects on fishes and other organisms living in the receiving water.

Separation of urine introduces a fraction containing a concentration of both plant nutrients and pharmaceuticals as those are mainly excreted in the urine. New types of exposure routes could therefore not be disregarded if human urine is spread on arable land. However, concentration of pharmaceuticals in a smaller fraction as urine may also open up for future possibilities where the urine is treated in order to neutralize harmful substances.

In terms of barriers, the soil can be seen as an extra filter providing a barrier where the soil organisms could degrade different substances. Nevertheless, it is of crucial importance that more studies concerning chemical risks associated with both conventional treatment of wastewater and, especially, different alternatives as for example urine-separating systems are initiated.

Why should a urine-separating system be implemented?

Earlier performed systems analyses of wastewater systems (e.g. Bengtsson et al., 1997; Kärrman et al., 1999; Jönsson et al., 2000) as well as this study point out urine separation as a more favourable system in many environmental aspects compared to a conventional system. Urine-separation will decrease the nutrient emissions and increase the recycling potential of non-renewable plant nutrient resources. Considering the energy use as well as acidification, the result from this study is more ambiguous, and will be more dependent on the context. In the short term, introduction of urine-separating systems could be efficient for reducing the discharge of nutrients to water, especially in areas with poor on-site treatment or in areas with conventional wastewater treatment without additional nitrogen removal.

Recycling of plant nutrients to farmland in order to avoid depletion of resources might in the long run however be the main reason for introducing new systems. Up to today, focus has been mainly on phosphorus. Not only phosphorus however but also potassium and sulphur should be considered when discussing future wastewater systems, as the reserves worth mining are limited (Balmér et al., 2002). Even though the need for potassium is limited on clay soils as exemplified in this study, the need for potassium is apparent on sandy soils. Today, there is no urgent need for plant nutrient from sewage products in conventional farming. In organic farming on the other hand, there is a lack of easily available plant nutrient. If source-separated sewage products as urine are allowed in organic farming, it is most likely that the interest from the farmers would be higher.

Urine-separating systems are robust systems, even though there is a need for further technical development of different parts of the system (Jönsson et al., 2000). A future big-scale introduction of source-separated systems requires however that experiences are made through successive implementation.

Agricultural aspects on the design of a urine-separating system

A urine-separating system could be designed in many ways. Urine-separation combined with dry handling of the faeces makes two fractions, which require different treatments, both at household level and at later stages. Alternatively, supplementing urine-separation to an already existing system could be an easy way to upgrade the system. In this study, faeces were assumed to be treated in a wastewater treatment plant. Hereby, the source-separated system affected the handling in the wastewater treatment plant only on the marginal. Another solution could be on-site systems with existing infiltration upgraded with a urine-separating system. In this way, many on-site systems on the countryside with deficient reduction would be more environmentally adapted.

How shall the urine storage be designed?

The urine should preferably be placed near the arable land where the spreading occurs. Hereby, the spreading could be optimised in time, a necessity for a high utilization. Central storage facilities near farmland where the urine is spread will also facilitate the control of the hygienisation, i.e. that the urine is stored long enough, and the possibility to make analyses of the plant nutrient content before spreading. For sufficient hygienisation, two alternating storages are safer than only one, if urine is collected all around the year.

Urine in a crop rotation and in time

Substitutability of mineral fertiliser with a sewage product is an important environmental aspect when assessing wastewater systems. This implies that the plant nutrient content is known, and use of an optimised technique for spreading it on a suitable crop. Urine should be spread during spring or in the growing season preferably on grain. Spreading on grassland is not recommendable as the ammonia emissions can be high as shown in experiments with cattle urine on ley (Rodhe et al., 1997). The best handling in theory is however not being done in practice. A survey of experiences of human urine in agriculture illustrated this dilemma (Fernholm, 1999). Out of ten farmers, only one used trailing hoses. Five of the farmers spread urine on ley and only three farmers had analyses of the plant nutrient content. It was also common to spread urine during the autumn, despite the fact that this leaves only a very small proportion of the nitrogen in the urine for the succeeding crop.

These practical experiences highlight the importance of disseminating recommendations for the agricultural use of human urine. Implementation of urine-separating systems requires a holistic perspective where the handling on the farm is crucial for the sustainability of the whole system.

CONCLUSIONS

- For sustainability, the future society must, to a much larger extent than today, be based on recycling. Source-separation of human urine is one promising technology in this respect.
- A change according to the urine-separating scenario as described in this study, will decrease the emissions of N and P to water, and increase the recycling of plant nutrients. A minor decrease of emitted GWP-gases will also appear, due to the avoided use of mineral fertiliser when urine is used as fertiliser. This decrease is larger if the electricity production is based on fossil fuels.
- Using an agricultural perspective gives an understanding of how an agricultural system using human urine could be shaped. Hereby, other factors than plant nutrient substitutability, e.g. yield reduction, can be included.
- Whether an introduction of urine-separating systems will increase or decrease the contribution to acidification depends on whether nitrogen compounds will contribute to acidification. If a maximum scenario is used, the urine-separating scenario contributes more to acidification than the conventional scenario. The opposite is true when a minimum scenario is used.
- The use of a change-oriented perspective on the investments gave interesting information on the environmental impact. Hereby, major changes in e.g. the construction could be taken into account, without considering the whole construction phase.
- Whether a urine-separating scenario will decrease the total energy use depends on many factors, and is not obvious. The construction phase might give a considerable contribution, and effects on the existing wastewater treatment are also an important factor.
- With the assumptions in this study, the urine could be transported more than 40 km one way without exceeding the total primary energy used in the conventional scenario. However, minimising transports is just one of several key issues from an energetically point of view.
- When urine is used as fertiliser, it is of great importance that the composition regarding heavy metals, organic pollutants, pathogens and plant nutrients can be guaranteed. The level of heavy metals in human urine is very low. The contribution of e.g. cadmium is even lower than in some “cadmium-free fertilisers”. However, the risks related to pharmaceuticals in urine must be further investigated.

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APPENDIX 1. INVENTORY RESULT FROM THE WHEAT PRODUCTION SYSTEM - CONVENTIONAL SCENARIO

Operation	Fossil fuel (MJ)	Electricity (MJ)	CO_2 (g)	N_2O (g)	CH_4 (g)	NO_x (g)	HC (g)	CO (g)	SO_2 (g)	PO_4^{3-} (g)	P (g)	NH_3 (g)	N to water (g)	NO_3 (g)
Precombustion (farm operations)	138		8064		4.6	71	76	4.6	44					
Stubble cultivation	191		14267			143	5.7	14.7	17.9					
Ploughing	580		43249			573	17.1	49.3	54.2					
Harrowing (3 times)1	569		42457			511	9.0	23.9	53.2					
Sowing	123		9151			117	4.2	13.2	11.5					
Fertilising (2 times)	156		11603			148	5.3	16.9	14.5					
Spraying	74		5538			70	2.5	8	6.9					
Harvesting	530		39557			524	15.6	45.1	49.6					
Transports within the farm	71		5301			65	2.6	10.9	6.6					
Emissions from soil and fertiliser				2829		311					500	1061		44300
Plant emissions												1500		
Indirect emissions				915										
Production of 518 kg N28	6326	189	467754	2911	448	787	0.012	54.4	663	0.053	0.00047	106	59	
Production of 95 kg P20	442	167	61370	5.7	113	365	0.0042	85	763	0.024	66	0.029	2.0	
Transport of P fertiliser	55		4047			85	3	4	0.52					
Transport of N fertiliser	67		4973			44	4.5	5	0.63					
Per hectare:	9322	356	717330	6660	566	3814	146	335	1685	0.077	566	2668	61	44300
Per kg:	1.61	0.062	124	1.15	0.098	0.659	0.025	0.058	0.291	0.000013	0.098	0.46	0.011	7.7

APPENDIX 2. INVENTORY RESULT FROM THE WHEAT PRODUCTION SYSTEM - URINE-SEPARATING SCENARIO

Operation	Fossil fuel (MJ)	Electricity (MJ)	CO_2 (g)	N_2O (g)	CH_4 (g)	NOx (g)	HC (g)	CO (g)	SO_2 (g)	PO_4^{3-} (g)	P (g)	NH_3 (g)	N to water (g)	NO_3 (g)
Precombustion field operations	155		9006		5.1	80	85	5.1	49					
Stubble cultivation	191		14267			143	5.7	15	18					
Ploughing	580		43249			573	17.1	49	54					
Harrowing (3 times)1	569		42457			511	9	24	53					
Sowing	123		9151			117	4.2	13	11					
Fertilising (once)	78		5802			74	2.7	8.4	7.3					
Bandspridning urin	226		10548			32	1.3	5.3	13.2					
Spraying	74		5538			70	2.5	8	6.9					
Harvesting	530		39557			524	15.6	45	50					
Transport-band spreading	144		10759			111	4.5	19	13					
Transport-others within the farm	66		4905			52	2.1	8.8	6.2					
Emissions from soil and fertiliser				2924		326					500	4934		44300
Plant emissions												1500		
Indirect emissions				1032										
Production of 259 kg N28	3163	95	233877	1456	224	394	0.006	27.2	332	0.027	0.0002	53	30	
Production of 55 kg P20	257	97	35724	3.3	65.8	212	0.002	49.2	444	0.014	38.3	0.017	1.2	
Transport P-fertiliser	32		2356			49	2.0	2.6	0.30					
Transport N-fertiliser	34		2486			22	2	2.3	0.32					
Transport urin 10x2 km (34.5 ton)	1277		98291	4	10	957	42	17	27					
Pipes and storage	2070	131	161108	6.3	23	1560	0.38	30	4.83					
Machinery		302	2368	0.2	15	5	1	5	3.9					
Avoided burdens														
Drinking water, treatment & distribution	-221	-856	-23711	-1.3	-44	-178	-10	-18	-16					
Wastewater, pumping & treatment		-311	-2435	-0.22	-15	-4.7	-0.9	-5.6	-4.0					
Extra nitrogen removal		-327	-2564	-0.23	-16	-4.9	-0.9	-6	-4.2					
Production of precipitation chemical	-154	-276	-10839			-183	-6.5	-24	-78					
Transport of precipitation chemical	-62		-4571			-40	-4.1	-4.3	-1.0					
N and P emissions to water											-397		-31740	
Per hectare:	9132	-1144	687328	5423	263	5402	175	276	991	0.0407	142	6487	-31709	44300
Per kg:	1.68	-0.210	126	1.00	0.0483	0.993	0.0322	0.0508	0.182	7.49E-06	0.0260	1.19	-5.83	8.14

APPENDIX 3. INVENTORY RESULT FROM THE BARLEY PRODUCTION SYSTEM - CONVENTIONAL SCENARIO

Operation	Fossil fuel (MJ)	Electricity (MJ)	CO_2 (g)	N_2O (g)	CH_4 (g)	NOx (g)	HC (g)	CO (g)	SO2 (g)	PO_4^{3-} (g)	P (g)	NH_3 (g)	N to water (g)	NO_3 (g)
Precombustion (farm operations)	128		7490		4.3	66	71	4.3	41					
Stubble cultivation	191		14267			143	5.7	15	18					
Ploughing	580		43249			573	17	49	54					
Harrowing (3 times)1	569		42457			511	9.0	24	53					
Sowing and fertilising	135		10074			129	4.6	15	13					
Spraying	74		5538			70	2.5	8	6.9					
Harvesting	530		39557			524	16	45	50					
Transports within the farm	60		4509			48	2.0	8.1	5.7					
Emissions from soil and fertiliser				1571		171					500			39870
Plant emissions												1500		
Indirect emissions				790										
Production of 295 kg NP 27-5	3552	90	266385	1345	273	540		64	917	0.046	0.56	59	31	
Transport of NP fertiliser	38		2832			25	2.5	3	0.36					
Per hectare:	5859	90	436358	3707	277	2801	130	235	1159	0.046	501	1559	31	39870
Per kg:	1.4	0.021	103	0.9	0.066	0.66	0.031	0.056	0.27	0.00001	0.12	0.37	0.007	9.4

APPENDIX 4. INVENTORY RESULT FROM THE BARLEY PRODUCTION SYSTEM - URINE-SEPARATING SCENARIO

Operation	Fuel (MJ)	Electricity (MJ)	CO_2 (g)	N_2O (g)	CH_4 (g)	NO_x (g)	HC (g)	CO (g)	SO_2 (g)	PO_4^{3-} (g)	P (g)	NH_3 (g)	N to water (g)	NO_3 (g)
Precombustion field operations	146		8628		5	76	81	5	47					
Stubble cultivation	191		14267			143	5.68	14.7	17.9					
Ploughing	580		43249			573	17.1	49.3	54.2					
Harrowing (3 times)1	569		42457			511	9	23.9	53.2					
Sowing and fertilising	135		10074			129	4.6	14.5	12.6					
Bandspridning urin	191		10548			31.8	1.31	5.3	13.2					
Spraying	74		5538			70.4	2.5	8	6.9					
Harvesting	530		39557			524	15.6	45.1	49.6					
Transport-band spreading	102		7595			81	3.3	13.6	9.5					
Transport-others	60		4509			48	2.0	8.1	5.7					
Emissions from soil and fertiliser				1621		182					500	3050		3987
Plant emissions												1500		
Indirect emissions				884										
Production of 110 kg NP 27-5	1320	34	99330	502	102	201		24	342	0.017	0.21	22	12	
Transport NP fertiliser	14		1056			9	1	1.0	0.13					
Transport urin 2x10 km (24 ton)	888		68376	3	7	666	29	12	19					
Pipes and storage	1440	91.2	111595	4.4	16	1085		21						
Machinery		302	2368	0.2	15	5	1	5	7.3					
Avoided burdens														
Drinking water, treatment & distribution	-154	-595	-16495	-0.88	-30	-124	-7	-13	-11					
Wastewater, pumping & treatment plant		-216	-1694	-0.15	-11	-3	-1	-4	-2.8					
Extra nitrogen removal		-227	-1783	-0.16	-11	-3	-1	-4	-3.0					
Production of precipitation chemical	-107	-192	-7540			-127	-5	-17	-54					
Transport of precipitation chemical	-43		-3180			-28	-3	-3	-1					
N and P emissions to water											-276		-22080	
Per hectare:	5937	-804	438455	3012	88	4051	158	210	566	0.0171	224	4572	-22068	3987
Per kg:	1.50	-0.202	110	0.76	0.0221	1.02	0.0398	0.0528	0.143	4.29E-06	0.056	1.15	-5.56	10

APPENDIX 5. PRODUCTION OF MINERAL FERTILISER PRODUCTS

Table 1. Energy and emissions related to production of mineral fertiliser products (Davis & Haglund, 1999)

Energy and emissions	Fosfor (P20)	Kväve (N28)	NP Sulphur (27-5)
Energy			
Diesel [MJ]	1.91	0.261	0.407
District heat, Swedish average [MJ]		-1.22	-0.988
Electricity, European average [MJ]	1.76	0.101	0.216
Electricity, Swedish average [MJ]		0.365	0.306
Fuel oil, ship (2-stroke) [MJ]	0.23	0.0122	0.0225
Hard coal [MJ]		1.32	1.17
Natural gas (>100kW) [MJ]		10.6	9.36
Oil, heavy fuel [MJ]	2.51	1.24	2
Steam [MJ]	2.26	-0.489	0.250
Total energy consumption [MJ]	6.40	12.7	12.3
Emissions			
CO_2 [g]	646	903	903
CO [g]	0.890	0.105	0.218
HC (aq) [g]	0.000044	0.0000241	0.0000245
N_2O [g]	0.0601	5.62	4.56
NH_3 [g]	0.000306	0.205	0.2
NO_2 [g]		0.627	0.509
NO_X [g]	3.84	1.52	1.83
Phosphate (aq) [g]	0.000118	0.0000161	0.0000252
PO_4^{3-} (aq) [g]	0.000254	0.000103	0.000155
SO_2 [g]	8.03	1.28	3.11
SO_3 [g]	0.278		0.0899
SO_4^{2-} (aq) [g]			0.173
CH_4 [g]	1.19	0.865	0.925
COD (aq) [g]	0.00735	0.000434	
P-tot (aq) [g]	0.692	9.16E-07	0.00190
N-tot (aq) [g]	0.0209	0.114	0.106

APPENDIX 6. SOIL COMPACTION

The traffic intensity in the model is expressed in Mgkm (the product of the weight of the vehicle and the distance driven). Soil moisture content and wheel equipment are taken into account by multiplying by a correction factor.

Assumptions: The tractor weighted 5 tons and the one-axled spreader weighted 4 tons empty and in 14 tons loaded. Tyre inflation pressure in tractor front wheels was 100 kPa and in rear wheels 100 kPa. The pressure in spreader wheels was 150 kPa. The distance driven in the field was set to 2.1 km.

Topsoil compaction

According to the soil compaction model, the yield reduction due to compaction in the topsoil is assumed to be proportional to Mgkm-corr and the clay content as shown in the formula below.

Cumulative yield loss (%)=0.00154 × Mgkm-corr × clay content (%)

With the correction factors shown in Table 1 and a clay content of 30%, the yield losses due to compaction in the topsoil was calculated to 0.7%.

Subsoil compaction

In the model, yield losses due to subsoil compaction is assumed to be proportional to Mgkm-corr. The model divides the subsoil into two layers. In the 25-40 cm layer damage is assumed to persist during 10 years, but the cumulative yield reduction is expressed in percent of one year's yield. When estimating the Mgkm in the model, the axle load is reduced by 4 Mg. The formula for this compaction is:

Yield losses (%)=Mgkm-corr /40

In deeper layers, damage will be exceptionally persistent. Therefore, the future yield losses are expressed as a percentage of annual yield reduction. When estimating the Mgkm in the model, the axle load is reduced by 6 Mg. The formula for this compaction in the deeper layer is as follows:

Annual yield loss (%)=Mgkm-corr /400

Yield losses caused by soil compaction

The yield reduction expressed in percent of one year's yield was 0.9%, when including the effect in both the topsoil and the upper layer of the subsoil (Table 1). The persistent effects in the deeper part of the subsoil were calculated to 0.005% yearly. If the reduction during 100 years should be addressed to the crop under study, these yield losses were another 0.5%.

Table 1. Effects on yield from the spreading of urine

	Topsoil	25-40 cm	40 cm
Correction factor, 100 kPa	0.50	0.80	0.80
Correction factor, 150 kPa	0.60	1.0	1.0
Mgkm-corr. front wheels	2.1	0	0
Mgkm-corr. rear wheels	5.2	1.7	0
Mgkm-corr. spreader	8.8	6.3	2.1
Yield losses (% of one years yield)	0.74	0.20	
Future annual yield losses (%)			0.0052

APPENDIX 7. TRANSPORTATION AND FUEL REQUIREMENT

Transports between farm centre and field

The average distance between farm and field was set to 1 km. When the transports between farm and field were calculated, it was assumed that field operations were performed during four hours before returning to the farm. Figures on velocity, working width and correction factors were taken from SLU (1996) and time required for one hectare was calculated as follows:

Time required = 10/(working width × velocity) × correction factor.

The environmental load per hectare related to these transports was therefore divided with the number of hectare cultivated during four hours (Table 1).

Table 1. Time required for field operations depending on velocity, working width and correction factors

Field operation	Velocity	Working width	Corr. factor	Time/hectare	Hectare/4 hours
Stubble cultivation	6	3	1.4	0.81	5
Ploughing	5	1.6	1.4	1.75	2
Harrowing	6	6	1.4	0.39	10
Sowing	7	4.5	1.8	0.57	7
Fertilising	7	5	1.6	0.47	8
Spraying	8	12	1.2	0.13	32
Harvesting	4	4.6	2.1	1.14	4

Transports with ship and truck

Figures on the energy required and emissions occurring during transports of mineral fertilisr and chemicals were taken from the NMT – nätverket för transport och miljö (network for transport and environment) website (NTM, www).

Table 2. Fossil energy required (MJ) and emissions (g) during transports by ship and truck and trailer per ton-kilometre including pre-combustion

Transport	Fossil energy (MJ)	CO_2 (g)	NO_x (g)	HC (g)	CO (g)	SO_2 (g)
Ship (2´- 8´dwt)	0.30	22	0.54	0.018	0.025	0.36
Truck and trailer	0.65	48	0.42	0.043	0.045	0.01

In Table 3, emissions and energy related to only precombustion and the total environmental load related to transportation with heavy vehicles including precombustion is shown. Data on combustion were taken from Arnäs et al. (1997) and data on the total environmental load from IVL (2001).

Table 3. Energy and emissions related to pre-combustion (per MJ fuel)

	Fossil energy (MJ)	CO_2 (g)	NO_x (mg)	CH_4 (mg)	CO (mg)	SO_2 (mg)
Combustion	0.06	3.5	31	2	2	19
Diesel total	0.06	77	750	8	13	21

APPENDIX 8. ELECTRICITY

Figures on environmental impact per MJ produced electricity used in this report (Table 1) was based on the average mix of electricity used in Sweden (Uppenberg et al., 2001).

Table 1. Environmental impact per MJ produced electricity

Resources used	
Total energy used [MJ]	0.032
Emissions to Air	
NO_x [mg]	15
SO_x [mg]	13
CO [mg]	18
HC+NMVOC+VOC [mg]	2.9
CO_2 [mg]	7842
N_2O [mg]	0.71
CH_4 [mg]	49
Particles [mg]	2.5
NH_3 [mg]	0.22

For calculations on the environmental impact from the marginal production, figures from production of electricity from coal power plants were used (Table 2).

Table 2. Environmental impact per MJ produced electricity from coal power plants

Emissions to Air	
NO_x [mg]	98
SO_x [mg]	160
CO [mg]	92
HC+NMVOC+VOC [mg]	4.6
CO_2 [mg]	210 000
N_2O [mg]	3.4
CH_4 [mg]	2500
Particles [mg]	59
NH_3 [mg]	5.5

APPENDIX 9. PRODUCTION OF PRECIPITATION CHEMICALS

Figures from the production of PIX 111 (Table 1) were taken from Frohagen (1997).

Table 1. Energy and emissions related to production of PIX 111 (per kg)

Energy and emissions	
Energy consumption	
Fossil fuel (MJ)	0.97
Electricity (MJ)	1.7
Additional	0.06
Emissions to air	
CH_4 (g)	
CO (g)	0.15
CO_2 (g)	68
HC (g)	0.041
NO_x (g)	1.15
SO_x (g)	0.49
Emissions to water	
NH_3 (g)	0.035
N-tot (g)	
P-tot (g)	

APPENDIX 10. ENERGY CONSUMPTION RELATED TO PRODUCTION OF STORAGE FACILITIES

In Table 1, the energy required for producing different materials is shown. Data were taken from Tillman et al. (1996).

Table 1. Energy consumption related to material and machine work performed

Material used and machine work performed	Unit	Electricity (MJ/unit)	Fossil fuel (MJ/unit)
Concrete	kg	0.0751	0.774
Reinforcing bars	kg		21.34
Macadam	m^3	11.15	8.95
Plastic pipes	kg	4.58	79.91
Excavating, tank	m^3		3.05
Excavating, pipes	m		4.66

Table 2 and 3 illustrate the energy required for production of a bigger storage tank in concrete and a smaller one in plastic. In both cases, data were taken from Tillman et al. (1996).

Table 2. Amount of material required for a system with a 27 m^3 storage tank in concrete and the energy consumption related to this

Material	Amount required	Electricity (MJ)	Fossil fuel (MJ)
Concrete (ton)	27	2028	20898
Reinforcing bars (kg)	1300	0	27742
Macadam (m^3)	4.5	50	40
Plastic pipes (kg)	25	115	1998
Excavator (m^3)	85.5		261
Machinery work (m^3)	20		61
Total		2192	51000

Table 3. Amount of material required for a system with a 2 m^3 storage tank in plastic and the energy consumption related to this

Material	Amount required	Electricity (MJ)	Fossil fuel (MJ)
Concrete (ton)	0.7	53	542
Reinforcing bars (kg)	35	0	747
Glass fibre (kg)	140	434	8470
Macadam (m^3)	0.7	8	6
Plastic pipes (kg)	7.5	34	599
Excavator (m^3)	7		21
Wheel loader (m^3)	1.5		4
Total		529	10389

Table 4. Environmental impact from the production of 1 kg of PVC (Boustead, 1994 in Finnveden et al., 1996). Figures used for the storage cover in PVC

Fossil fuel (MJ)	Electricity (MJ)	CO (g)	CO_2 (kg)	SO_2 (g)	NO_x (g)
63	4	2.7	1.9	13	16

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