Science of the Total Environment 493 (2014) 209–219

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Environmental impact of recycling nutrients in human excreta to agriculture compared with enhanced wastewater treatment

J. Spångberg [a], P. Tidåker [b], H. Jönsson [a,*]

[a] Swedish University of Agricultural Sciences, Department of Energy and Technology, Box 7032, 750 07 Uppsala, Sweden
[b] Swedish Institute of Agricultural and Environmental Engineering, P.O. Box 7033, 750 07 Uppsala, Sweden

HIGHLIGHTS

- Environmental impacts of using blackwater and urine as fertilisers were assessed.
- Three scenarios assessed blackwater, urine and chemical fertilisers, respectively.
- Toilet fraction nutrients not recycled as fertiliser were removed in enhanced WWTP.
- Blackwater and urine proved better for GWP and energy use than chemical fertiliser.
- Blackwater and urine caused more eutrophying emissions than chemical fertiliser.

ARTICLE INFO

Article history:
Received 14 February 2014
Received in revised form 22 May 2014
Accepted 26 May 2014
Available online 17 June 2014

Editor: Simon Pollard

Keywords:
Wastewater treatment
Fertilisers
Life cycle assessment
Urine
Blackwater
Cadmium

ABSTRACT

Human excreta are potential sources of plant nutrients, but are today usually considered a waste to be disposed of. The requirements on wastewater treatment plants (WWTPs) to remove nitrogen and phosphorus are increasing and to meet these requirements, more energy and chemicals are needed by WWTPs. Separating the nutrient-rich wastewater fractions at source and recycling them to agriculture as fertiliser is an alternative to removing them at the WWTP. This study used life cycle assessment methodology to compare the environmental impact of different scenarios for recycling the nutrients in the human excreta as fertiliser to arable land or removing them in an advanced WWTP. Three scenarios were assessed. In blackwater scenario, blackwater was source-separated and used as fertiliser. In urine scenario, the urine fraction was source-separated and used as fertiliser and the faecal water treated in an advanced WWTP. In NP scenario, chemical fertiliser was used as fertiliser and the toilet water treated in an advanced WWTP. The emissions from the WWTP were the same for all scenarios. This was fulfilled by the enhanced reduction in the WWTP fully removing the nutrients from the excreta that were not source-separated in the NP and urine scenarios. Recycling source-separated wastewater fractions as fertilisers in agriculture proved efficient for conserving energy and decreasing global warming potential (GWP). However, the blackwater and urine scenarios had a higher impact on potential eutrophication and potential acidification than the WWTP-chemical fertiliser scenario, due to large impacts by the ammonia emitted from storage and after spreading of the fertilisers. The cadmium input to the arable soil was very small with urine fertiliser. Source separation and recycling of excreta fractions as fertiliser thus has potential for saving energy and decreasing GWP emissions associated with wastewater management. However, for improved sustainability, the emissions from storage and after spreading of these fertilisers must decrease.

1. Introduction

Eutrophication is caused by excessive inputs of nutrients to a water body. These nutrients cause large algal growth and sometimes algal blooms, with oxygen depletion when the algae die and decay. Eutrophication threatens many coastal ecosystems around the world (Randall, 2003; UNEP, 2006). The main sources of these nutrients are anthropogenic, such as wastewater systems, agriculture and atmospheric deposition largely due to the burning of fuels. One eutrophied water is the Baltic Sea, where the Baltic Sea Action Plan (BSAP) aims at recovering good environmental status. To achieve this, the surrounding countries have agreed on sharp decreases in eutrophying emissions by 2021 (HELCOM, 2011).

The direct nutrient discharges from Swedish municipal wastewater treatment plants (WWTPs), account for about 20–30% of the Swedish anthropogenic nitrogen and phosphorus discharges to the Baltic Proper

* Corresponding author. Tel./fax: +46 18 671886.
E-mail address: hakan.jonsson@slu.se (H. Jönsson).

http://dx.doi.org/10.1016/j.scitotenv.2014.05.123

(SEPA, 2009). In Sweden, WWTPs already reduce about 60% of incoming nitrogen and 95% of incoming phosphorus (SEPA, 2013a). To achieve the reductions required by the BSAP, WWTPs have been suggested to reduce at least 80% of incoming nitrogen and to emit a maximum of 0.2 mg phosphorus per litre outgoing water (SEPA, 2009). This will increase the use of resources such as precipitation chemicals, carbon sources and energy at WWTPs (SEPA, 2009). Current target for reduction of the Swedish emissions are 9240 tonnes of nitrogen and 530 tonnes of phosphorus (HELCOM, 2013).

The global population is expected to grow by about 35% by 2050 (UN, 2013), increasing the demands on agricultural production and use of chemical fertilisers. Today human excreta are almost universally looked upon as a hazardous waste to be disposed of. However, the nutrients in urine and faeces derive from ingested food and, if recycled, might be important as fertiliser in future agriculture. This would be in line with the waste hierarchy in the Waste Directive of the European Union (EC, 2008a), where re-use and recycling are given higher priority than disposal, thus promoting a change of view on human excreta from waste to resource. It would also agree with Rockström et al. (2009), who claim that the global flows of reactive nitrogen are ought to be reduced.

The urine fraction (excluding flush water) contributes about 1% to the total flow of urine, faeces and greywater (Jönsson et al., 2000), but gives the largest contribution to the flow of macronutrients, about 80% of the nitrogen and 60% of the phosphorus. Blackwater (urine, faeces, toilet paper and flush water) contains about 90% of the nitrogen and 90% of the phosphorus in the excreta (Jönsson et al., 2005). The nutrient content, before losses, in urine and faeces excreted by the Swedish population corresponds to 28% of the total nitrogen and 44% of the total phosphorus in chemical fertilisers sold in Sweden 2010/11 (Statistics Sweden, 2012a). The nitrogen in urine mainly consists of ammonium and has 85–100% of the plant availability of the nitrogen in chemical fertilisers (Jönsson et al., 2000). The phosphorus in urine is mainly in the form of phosphate ions and is as available to plants as soluble phosphorus fertilisers (Kirchmann and Pettersson, 1995). The nutrients in faeces are somewhat less available, since some of them are bound to non-degraded organic material. About 50% of the nitrogen in faeces is water-soluble and thus immediately available for plants (Jönsson et al., 2005). The phosphorus in faeces is largely bound to calcium and is comparable to that in chemical fertilisers, although with slower solubility (Frausto da Silva and Williams, 1997).

There is no law controlling the use of human excreta as fertiliser in conventional farming in Sweden, although to a certain extent it is covered by the regulation regarding safe use of sewage sludge (EEC, 1986). According to the EU Directives on organic production, human excreta are not allowed as fertilisers in organic farming (EC, 2008b), even though human excreta well fulfil the intention of the Directive that "in order to minimise the use of non-renewable resources, wastes and by-products of plant and animal origin should be recycled to return nutrients to the land" (EC, 2007).

Over recent years, a number of source-separation techniques, especially for urine separation, have been investigated. One review by Maurer et al. (2006) concluded that there are many urine treatment processes available both for hygienisation and nutrient-recovery, e.g. struvite precipitation and ammonia stripping, but that further work is needed to optimise the processes. For separating urine, special toilets have been developed with a front bowl collecting the urine and a rear bowl collecting the faeces and toilet paper. The urine is piped to a storage tank for further treatment. Collection of source-separated blackwater (urine, faeces, flush water and toilet paper) in collection tanks for vehicle transport to a WWTP is fairly common in Sweden and many other countries.

Proper hygiene control is important when using human excreta as fertiliser. Urine is sterile in the bladder of healthy individuals, and after excretion it contains low counts of normal skin flora (Jönsson et al., 2000). The hygiene risk of faeces, which frequently contain bacterial, virus and parasitic pathogens, is high. Therefore, for urine the main hygiene risk is associated with faecal cross-contamination (Schönning and Stenström, 2004). For hygienisation of urine, storage is a low-tech and low resource-demanding alternative. The recommendations are storage for 1 to 12 months depending on storage temperature and crop to be fertilised (Schönning and Stenström, 2004; WHO, 2006).

A low-tech hygiene treatment of faeces is storage for at least 2 years (WHO, 2006). The storage time can be greatly reduced by adding e.g. pH-increasing additives such as lime or urea, which could come from urine (Fidjeland et al., 2013; Schönning and Stenström, 2004). The excreta fractions are relatively low in heavy metals (Jönsson et al., 2005), and urine contains far smaller amounts of heavy metals than faeces. The concentrations of most metals are much lower, by at least 10-fold, in urine than in animal manure (Winker et al., 2009). Excreta are the main contributor of pharmaceutical substances and hormones to wastewater, where the problems caused by sex hormones emitted with wastewater effluents are very well documented (Liney et al., 2006; Vajda et al., 2008).

Ammonium nitrate is the most commonly used compound for chemical nitrogen fertiliser in Europe (Fertilizers Europe, 2013). About 80% of the global ammonium nitrate production is by fixation of atmospheric nitrogen using natural gas as a source of both hydrogen and energy (Brentrup and Pallière, 2008). The global warming impact from nitrogen fertiliser production is mainly due to the large emissions of carbon dioxide (CO_2) when using natural gas and of nitrous oxide (N_2O) from the nitric acid production, a step within the nitrate production process (Brentrup and Pallière, 2008). The use of phosphate rock for the production of chemical fertilisers is also a concern, as the life time of economic reserves of phosphate rock is finite and is estimated to be exceeded in the next 30–370 years (Cordell and White, 2011; USGS, 2013). Another environmental issue regarding fertiliser use is the cadmium flow to arable land. Cadmium exposure in Sweden, mainly from smoking and food intake, is many times above or at safety levels that can have harmful effect on bones and kidneys (KEMI, 2008). This not only emphasises the health risk to humans but also that humans excrete relatively large amounts of cadmium. KEMI, the Swedish Chemical Agency, recommends a limit of 12 mg cadmium per kg phosphorus added to soil to keep safe levels (KEMI, 2008), but analyses of chemical fertilisers sold in Europe show median concentrations of 87 mg per kg phosphorus (Nziguheba and Smolders, 2008).

Recycling the nutrients in human excreta to arable land as fertiliser can reduce the use of energy and non-renewable resources for production of chemical fertilisers. It can also reduce the use of energy and chemicals at WWTPs, both because less nutrients need to be removed and because the biological wastewater process, and especially the nitrogen removal process, function more efficiently when urine is source-separated from the influent to the WWTP (Wilsenach, 2006; Wilsenach and van Loosdrecht, 2003).

A number of studies have demonstrated the environmental benefits of using human excreta as fertiliser on arable land (Benetto et al., 2009; Remy and Jekel, 2008; Tidåker et al., 2007a, 2007b). However, most of these studies focus on the urine fraction and no previous study has compared systems with the same direct emissions of nutrients from wastewater to water. The present study aimed to fill this gap.

2. Goal and scope

The goal of this study was to assess the environmental impact of separating and recycling nutrients in human urine and faeces for use as fertiliser on arable land, compared with treating these fractions at a WWTP with enhanced treatment and fertilising the arable land with chemical fertilisers. Three scenarios in a Swedish setting were evaluated in a life cycle perspective for a new housing district in the Stockholm area. In the blackwater scenario, ultra-low-flush vacuum toilets were used and the blackwater was hygienically treated and later spread on arable land. In the urine scenario, the urine was separated at source, stored and spread on arable land while the faeces were piped to and treated at an advanced WWTP. In the NP scenario, chemical fertiliser was used and the blackwater was treated at an advanced WWTP. The direct emissions of nitrogen and phosphorus from excreta to water were kept constant in all three scenarios. Life cycle assessment (LCA) methodology based on the ISO 14000 series standards (ISO, 2006a, 2006b) was followed and the environmental impacts considered relevant were assessed. Avoided activities in the systems were included by using system expansion (Guinée et al., 2002; ISO, 2006b). The Ecoinvent 2.2 databases (Ecoinvent Centre, 2010) and the SimaPro v 7.3.3 software (PRé consultant, 2012) were also used to obtain and analyse data.

2.1. Functional unit

In LCA, a functional unit (FU) is chosen to make all scenarios comparable. The FU consists of the functions which it is essential that all scenarios fulfil. Two functions were included in the FU of this study, to reflect both the need for fertiliser and the required treatment of the wastewater.

- Production of fertiliser containing 1 kg plant-available nitrogen and 0.15 kg phosphorus after emissions at storage and after spreading.
- Source separation or removal of 1.21 kg nitrogen and 0.15 kg phosphorus from the wastewater, thus maintaining zero nutrient emissions from excreta with the effluent from the WWTP in all scenarios.

The 1 kg of plant-available nitrogen that is added to agricultural soil comes with 0.15 kg of phosphorus when it is spread as blackwater, as this is the composition of blackwater. It is required that the same amounts of nutrients are spread also in the other scenarios. Source-separating this amount of blackwater decreases the load to the WWTP by 1.21 kg of nitrogen and 0.15 kg of phosphorus. The difference between 1.21 kg nitrogen removed from wastewater and 1 kg plant available nitrogen recycled is due to not all nitrogen being plant available and also due to nitrogen losses in the recycling system.

2.2. Impact categories

The impact categories studied were global warming potential (GWP), potential eutrophication, potential acidification and use of total and of non-renewable primary energy. GWP was expressed in CO_2-equivalents and calculated using indices for a 100-year time horizon (IPCC, 2007). Potential eutrophication was expressed in PO_4^{3-}-equivalents and potential acidification in SO_2-equivalents, both calculated according to CML 2001 (Guinée et al., 2002). Both emissions to air, to water and to soil can cause potential eutrophication and potential acidification (Guinée et al., 2002) The energy balances included cumulative primary energy use, distinguished into non-renewable energy (fossil and nuclear) and renewable energy (biomass, hydropower, solar, wind, etc.). The flow of cadmium to arable soil, use of phosphate rock and potential carbon storage in soil were also assessed.

2.3. System boundaries

Included scenarios (see Fig. 1) included all relevant processes associated with the fertiliser use and treatment of the wastewater fractions, including production, transport and spreading. Treatment of greywater was not included in the study. In all three scenarios, the emissions to water from the WWTP of nitrogen and phosphorus from the same amount of excreta were set to zero. In the blackwater scenario, this was achieved by the blackwater being source-separated and recycled as fertiliser. Thus, it did not reach the WWTP. In the Urine scenario, the faeces and 25% of the urine, which was assumed to be not correctly source-separated, were piped to the WWTP. Therefore, the resources needed to fully remove these nutrients in the WWTP were calculated. In the NP scenario, all excreta were assumed to be piped to the WWTP and the resources for fully removing the nitrogen and phosphorus in the WWTP were calculated.

The effects of source-separation on sewage sludge production are for several parameters, e.g. phosphorus, simple but for most parameters complex, requiring a full dynamic simulation of the WWTP for covering the full effects on the sludge and such a simulation was outside the scope of this study. Furthermore, the main changes caused in this study by the source-separation were the changes in the required amounts of nutrients to be removed in the WWTP and these changes should be reflected by the data collected in the data in SEPA (2009) collected in the inventory. About 25% of the Swedish sewage sludge is spread on agricultural land, while 20% is used for covering landfills and 32% for soil production (Statistics Sweden, 2012b). In this study the sewage sludge was assumed to be put to use, but not on agricultural land, which is the most common situation in Sweden. This is also the situation for the sludge from the largest WWTP in Sweden, which for many years will be used for re-vegetation of a mine landfill (Stockholm Vatten, 2012a). This means that even the obvious changes in e.g. amount of phosphorus in the sludge did not affect any of the assessed environmental impacts.

For capital goods, only the additional infrastructure required in the separating scenarios was included. As spreading of blackwater and urine often involves new spreading equipment, production of tractor and agricultural equipment was included for these scenarios. The lifetime of the collection system was assumed to be 40 years and the lifetime of the lagoons 30 years. Final disposal of capital goods was included for all materials and vehicles used except for pumping equipment, which was assumed to be negligible, and the pipes and the storage tanks in the blackwater and the urine scenarios, which were assumed to be left on-site after end-of-use. The electricity in the scenarios was assumed to be a Nordic electricity mix, NORDEL, including the electricity supply of Denmark, Finland, Norway and Sweden (at grid) (Ecoinvent Centre, 2010).

The agricultural soil was included in the production system. This means that the fertilisers ending up in the soil were not included in the calculations of potential eutrophication and potential acidification. The field on which the fertilisers were spread was assumed to be at a distance of 30 km from the urban area of Stockholm city. Nutrient leaching from the arable soil was assumed to be similar for all scenarios, and was therefore not included.

3. Description of the scenarios

3.1. General description of the blackwater and urine scenarios

The source-separating blackwater and urine systems were assumed to be installed in a new house of three floors with three apartments on each floor and with two people living in each apartment. The extra infrastructure needed consisted of the pipes from the toilet to a collection tank outside the house (see Fig. 2). These pipes were in addition to the pipes needed for the collection of greywater. With a conventional wastewater system just one pipe collecting all sewage fractions is needed.

The collected blackwater and urine were brought to on-site storage, i.e. constructed lagoons near the field, by tanker trucks. Each lagoon was dimensioned to store sufficient blackwater or urine to supply 100 ha (with 50 kg of available nitrogen per hectare). For data on infrastructure included in the different scenarios, see Table 1.

Data for dimensioning of the systems, i.e. the amounts of urine, faeces, water, nitrogen and phosphorus etc. produced per person (Table 2), were mainly based on Jönsson et al. (2005) and on studies of the residential areas Palsternackan (Jönsson et al., 2000) and Gebers (Andersson and Jensen, 2002) in the Stockholm region. It was estimated that people were at home on average 65% of the day (Jönsson et al., 2005). The vacuum toilets used about 3 kWh per person and year (Wostman, 2013). For other data on materials and processes see

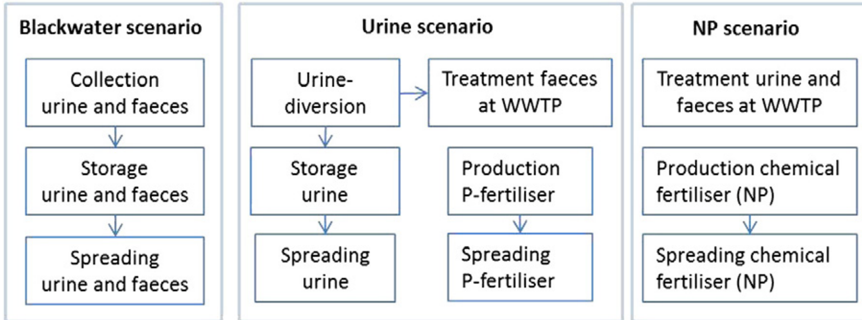

Fig. 1. System boundaries for the three scenarios studied. All three scenarios produced the functional unit (P-fertiliser = phosphate rock).

Appendix A and for details of transport included in the study see Appendix B.

3.2. Blackwater scenario

Blackwater was assumed to be collected for recycling, while the greywater was piped to a WWTP. State-of-the-art toilets for this type of system, ultra-low-flush vacuum toilets (0.5 l per flush), were assumed to be used for collection. The number of flushes during 65% of an average day that a person spends at home was estimated at 4.6 (Jönsson et al., 2005). In total, about 22 m^3 of blackwater were collected annually from the 18 persons in the house. To get a safety margin for the tank not to overflow, it was assumed to be emptied twice a year. Following WHO guidelines on safe use of excreta and wastewater in agriculture, the blackwater was assumed to be stored for two years (WHO, 2006). To fulfil this criterion, three constructed lagoons were placed by the field, one of which could be emptied and spread on the field, one could be filled up and one left for undisturbed storage during two years (filled until spring in one year, enclosed storage for two years, and then spread in spring). The lagoons were assumed to be mainly constructed of polyethylene (HDPE) with a covering liner of polyvinylchloride (PVC).

The blackwater was assumed to be spread in the same way as urine and data on engine emissions during spreading were taken from Lindgren et al. (2002). No studies on emissions from collection, storage and spreading of blackwater were found, but a few such studies on urine were available. As urine contributes almost 90% of the total nitrogen and more than 95% of the ammonia-nitrogen in blackwater (Table 2), the results for the emissions from collection, storage and spreading of urine were considered relevant also for blackwater. Studies on urine collection show that the ammonia-nitrogen emissions from a well-designed collection and tank system and its emptying and transport to a storage lagoon amount to 0.1% of total nitrogen (Jönsson et al., 2000), while the ammonia emissions from storage of animal urine in lagoons has been measured at 4% of total nitrogen (Karlsson and Rodhe, 2002). Emissions from spreading were set to 5% of ammonia-nitrogen, based on a study on ammonia emissions after application of human urine with trailing hoses in spring followed by incorporation 4 h later (Rodhe et al., 2004). Since both blackwater and urine have low dry matter content and low viscosity, the same emission factor was used for blackwater. The emissions from storage in the blackwater scenario were assumed to be 8%, twice as high as emissions in the urine scenario due to the 2–3 year storage time for blackwater, compared with 0.5–1.5 years for urine. Nitrous oxide emissions from spreading were included, directly as 1% (N_2O-N) of total nitrogen added to soil and indirectly as 1% (N_2O-N) of the ammonia-nitrogen emitted to air (IPCC, 2006). Indirect nitrous oxide emissions from ammonia emissions at collection and storage were included in the same way as for spreading.

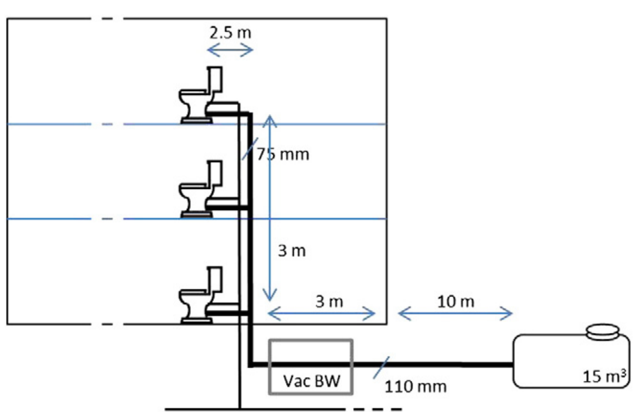

Fig. 2. Diagram of the toilet system assumed in the study. The pipe system with the thicker line plus the collection tank is the extra infrastructure needed. The type of toilet varied between the scenarios, but as the environmental impact depends more on the make of toilet than whether it is a vacuum, urine-diverting or conventional toilet, the toilet was not included in the additional infrastructure. The "Vac BW" box represented the vacuum system of the blackwater scenario.

3.3. Urine scenario

In the urine scenario, water-flushed, urine-separating toilets were used. The flush water amounted to 0.4 l per person and day at home for urine and 3.9 l for faeces. Based on measurements in several residential areas, 25% of the urine was assumed not to be separated but followed the faecal flow (Jönsson et al., 2000). About 8 m^3 of urine–flush water mixture was collected from the 18 persons in the house each year. The 15 m^3 collecting tank (Fig. 2) was assumed to be emptied once a year. In large systems where urine is collected from several households in the Swedish temperate climate, the urine should be stored for at least six months before being considered hygienically safe to fertilise food and fodder crops which are processed before consumption (Jönsson et al., 2000; WHO, 2006). This was the minimum storage time assumed in this study. Thus, this scenario had two lagoons per 100 ha, one for filling-up while the other was left for enclosed, undisturbed storage during at least six months. The ammonia emissions were assumed to be 0.1% from collection and transport (Jönsson et al., 2000), 4% from storage in the lagoon (Karlsson and Rodhe, 2002) and 5% after spreading

Table 1
Components included in the blackwater and urine scenarios.

Component	Blackwater scenario	Urine scenario
Pipes	38.5 m (75/100 mm)[a]	38.5 m (75/100 mm)[a]
Tank	15 m^3	15 m^3
Lagoon	3 × 3000 m^3	2 × 1500 m^3
Vacuum unit	1400 w/230 V[b]	

[a] 75 mm diameter indoors and 100 mm outdoors.
[b] Wostman (2013).

Table 2
Data on total flows of urine, faeces, nitrogen, ammonia, phosphorus, cadmium and carbon per person and day, 24 h (Jönsson et al., 2005).

	Amount	Total-N (g/p,d)	NH_4-N (g/p,d)	Total-P (g/p,d)	Cd (mg/p,d)	Carbon (g/p,d)
Urine	1.5 l/p,d	11.0	10.3	0.9	0.0005	4.3[a]
Faeces	53.1 g/p,d	1.5	0.3	0.5	0.0150	19.1[b]

[a] Andersson and Jensen (2002).
[b] Including toilet paper.

the urine fraction (Rodhe et al., 2004). Nitrous oxide emissions were calculated as in the blackwater scenario.

The faeces, together with the 25% of the urine which was not source-separated, were assumed to be piped to an advanced WWTP. To fulfil the FU of 1.21 kg nitrogen and 0.15 kg phosphorus removed from wastewater in this scenario, about 0.05 kg phosphorus and 0.13 kg nitrogen were removed at the WWTP (see Section 3.5). In addition, 0.05 kg phosphorus fertiliser was needed to fulfil the FU. This was done by including production of ground phosphate rock, which is a phosphorus fertiliser permitted for use in both organic and conventional farming.

3.4. NP scenario

The abbreviation NP refers to chemical fertilisers, which often is denoted as NP or NPK. The chemical fertiliser was calculated as being composed of triple superphosphate (TSP) and ammonium nitrate (AN), with an N:P ratio of 1:0.15. Data on energy use and GWP emissions for the production of these two fertiliser components were taken from Brentrup and Pallière (2008) and data on eutrophication and acidification emissions from Davis and Haglund (1999). For energy use and GWP emissions, the best available technique (BAT) was assumed to be used for production of AN. However, for TSP, European average data for 2006 were used due to lack of other information. The chemical fertiliser was assumed to be spread with a combi-drill. Based on comparison of total emissions from combi-drilling and only seed drilling, 16% of the total engine emissions and fuel use were allocated to the spreading (Lindgren et al., 2002). Indirect nitrous oxide emissions of 1% of total nitrogen added to soil were included (IPCC, 2006). At the WWTP the same amount of nitrogen and phosphorus as contributed by the urine and faeces was assumed to be removed and the energy and other resources needed for this were included (see Section 3.5). Transport vehicles, distances and data sources are given in Appendix B.

3.5. Treatment of nitrogen and phosphorus in the WWTP

In Sweden, roughly 95% of urban wastewater undergoes both biological and chemical treatment (SEPA, 2013a). The requirements on nutrient reduction at Swedish WWTPs are about to increase from an already fairly high level. This reduction on the margin will be easier to achieve if all excreta nutrients from new houses are source-separated and do not reach the WWTP, as was the case for the blackwater scenario. Thus, to compare this scenario in a fair way with other scenarios, the urine and the NP scenarios included the use of energy, chemical precipitants and carbon source (for nitrogen removal) required to fully remove the same amount of nitrogen and phosphorus in the WWTP as it receives from non-source-separated excreta. The resources needed for improving the nutrient reduction in the WWTPs from a high to an even higher level have been estimated by the Swedish Environmental Protection Agency (SEPA, 2009). For the additional high level nutrient reduction in the WWTP, 5.8 kWh and 2.2 kg BOD_7 (Biological Oxygen Demand during 7 days), e.g. carbon source, are needed per kg nitrogen removed and 179 kg precipitation chemical per kg phosphorus removed (SEPA, 2009).

Methanol was the carbon source assumed to be used at the WWTP. Data on environmental impact from its production were taken from Ecoinvent Centre (2010). For precipitation, iron-based precipitants were assumed to be used in this study. Data on its environmental impact were based on three common (iron-based) precipitants, with the brand name PIX and which are commonly used in Sweden (IVL, 2003).

The IPCC guidelines consider nitrous oxide emissions from wastewater treatment to be negligible (IPCC, 2006), although studies have shown that they can vary considerably (Foley et al., 2011; Westling, 2011). In a review by Westling (2011), the range of the nitrous oxide emissions was 0 to 2% (N_2O) of removed nitrogen. The Australian Government uses a similar number, 1.6%, as the N_2O emissions factor of nitrogen removed at a WWTP (DCCEE, 2010). In the present study 1% of removed nitrogen (influent minus effluent) was assumed to be emitted as nitrous oxide (N_2O), which is close to the emissions factor used in Finland (Westling, 2011), where the climate and treatment techniques are similar to those in Sweden.

3.6. Cadmium flow to soil and potential carbon sequestration

The cadmium content of the phosphate rock was assumed to be 3 mg cadmium per kg phosphorus, the level in phosphate rock from the Kola Peninsula, which is the source used for most of the phosphorus fertilisers in Sweden (Yara, 2010). The cadmium content in urine and faeces is shown in Table 2.

Adding organic matter, such as human excreta, to soil increases the soil carbon pool and can result in carbon sequestration. As carbon turnover in soil depends on many factors, e.g. soil type, temperature and soil microbial activity, accurate prediction of the carbon stored after application of organic material is difficult and uncertain. Estimated sequestration, expressed as the fraction of the added carbon that remains in the soil after 100 years, ranges between 2 and 14% in studies on digestate and compost applied to farm land (Bernstad and la Cour Jansen, 2012). In the model EASEWASTE, the estimated carbon storage factor is 2 to 16% in a 100-year perspective for digested municipal food waste (Hansen et al., 2006). Since the carbon in the urine is highly degradable and the main part is lost as urea, only the carbon from the faeces and toilet paper in the blackwater was assumed to contribute to potential carbon sequestration in this study. A carbon storage factor after 100 years of 7% of initial carbon added was assumed for the blackwater.

3.7. Normalisation

The results were normalised based on the environmental impacts from using the amount of nitrogen and phosphorus in blackwater that one person produces in one year as fertiliser. One person produces 2.2 kg plant-available nitrogen and 0.3 kg phosphorus in blackwater collected at home. In the normalisation, the environmental impacts associated with these amounts of nitrogen and phosphorus were calculated based on the results of the blackwater scenario for all three scenarios and then compared with the yearly total impact on energy input, GWP, potential eutrophication and potential acidification per capita in Sweden.

3.8. Sensitivity analysis

A life of 40 years for the toilet system (additional pipes and urine tank) and 30 years for the lagoon can be questioned, as fittings and infrastructure might change quickly. In a sensitivity analysis, the expected lifetime of the materials was reduced by half.

Data on ammonia emissions after spreading human urine and human excreta are scarce or fully lacking. In a sensitivity analysis, the ammonia emissions after spreading were doubled and reduced by half.

Increasing the nutrient reduction at a WWTP that already has a high nutrient reduction is energy and resource-intensive. In a sensitivity analysis, the electricity source was changed to coal power (UCTE hard coal power plant mix) (Ecoinvent Centre, 2010).

Part of the phosphorus reaches the WWTP bound to particles and will partly precipitate without chemical input. A large part of the phosphorus is dissolved. To remove this, most Swedish WWTPs use chemical precipitants. In a sensitivity analysis, the amount of precipitation chemical needed for phosphorus removal was changed from 179 kg precipitation chemical per kg phosphorus estimated for marginal removal by SEPA (2009) to 16 kg per kg phosphorus, which at the reduction level would be the average amount of chemical per kg phosphorus removed. This was calculated by using the present and the marginal use of precipitation chemicals according to SEPA (2009), dividing these by the total amounts of phosphorus currently removed in Sweden (Statistics Sweden, 2012b). In this way an average was calculated on the use of precipitation chemicals, including both the marginal amount for a stricter removal and the present use. The present amount of precipitation chemicals used was estimated by scaling up from the data for four WWTPs in Sweden with a load of more than 100,000 person equivalents (Stockholm Vatten, 2012a; Syvab, 2012; UMEVA, 2012; Uppsala Vatten, 2012).

As the emissions of nitrous oxide from WWTPs vary widely, in a sensitivity analysis the emissions were doubled from 1% to 2% nitrous oxide emitted per kg nitrogen removed. A recently developed technique, which might be more common in the future, is the use of autotrophic nitrogen removal, e.g. the nitritation–anammox process, which reduces the energy use by more than 50% for nitrogen removal and requires almost no external carbon source (Maurer et al., 2003; Siegrist et al., 2008). A sensitivity analysis was therefore also done on halving the energy use and removing the methanol input at the WWTP.

4. Results

4.1. Primary energy use

The results demonstrated that the NP scenario used 3 to 4 times as much energy as the blackwater and urine scenarios (Fig. 3). The installation of the separating system in the house (section Bw and urine collection system in Fig. 3) used most primary energy in the blackwater scenario, where construction of the collecting tank contributed about 60% of the primary energy used. In the urine scenario, about half the total primary energy was used for the reduction of nitrogen and phosphorus at the WWTP. About half this energy use derived from phosphorus removal and half from nitrogen removal. In the NP scenario 78% of the primary energy use derived from nitrogen and phosphorus removal at the WWTP, with 26% deriving from phosphorus removal and 74% from nitrogen removal. Of the total primary energy use, 90–94% derived from non-renewable energy sources in all scenarios.

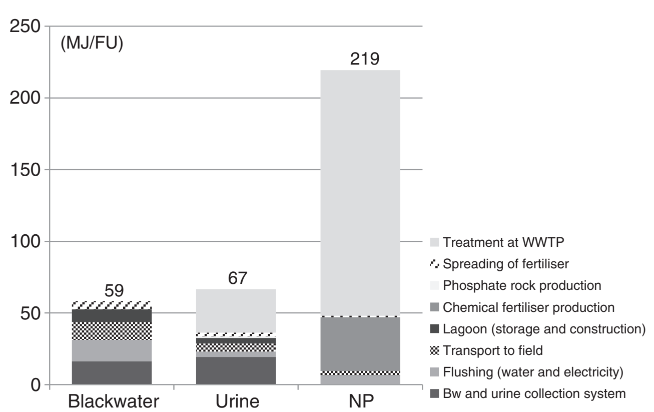

Fig. 3. Total primary energy use for the three scenarios divided between processes. Total primary energy use is also given, as the number above each bar. Bw = blackwater.

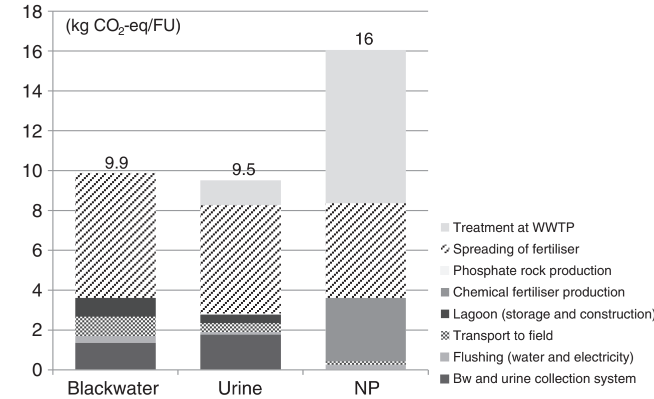

Fig. 4. Global warming potential for the three scenarios divided between processes. Total GWP is also given, as the number above each bar. Bw = blackwater.

4.2. Global warming potential (GWP)

For GWP, the impact from the NP scenario was 1.5 to 2 times that from the blackwater and urine scenarios. The indirect nitrous oxide emissions related to the emissions after spreading were the main contributor to GWP for the blackwater and urine scenarios (Fig. 4). The second largest contribution in these scenarios came from the collection system, 15 and 19% respectively. For the NP scenario, emissions after spreading also contributed greatly, almost 30%, to GWP, due to the direct and indirect nitrous oxide emissions from soil and ammonia emissions. In this scenario, however, the treatment of nitrogen and phosphorus at the WWTP had a greater impact, about 48% of total GWP.

4.3. Eutrophication

Since a criterion for all scenarios was to have the same water emissions of nitrogen and phosphorus emissions from the WWTP, there were no direct eutrophying emissions to water from the excreta fractions. Furthermore, the eutrophying water emissions from the fertilised field were considered similar and excluded from this comparative study. Thus, the potentially eutrophying emissions in the blackwater and urine scenarios were mainly direct gaseous emissions from the collection, handling, storage and spreading, in addition to upstream processes. In the NP scenario, the main potentially eutrophying emissions derived from the production of chemical fertilisers and the production of resources (precipitation used for nutrient removal at the WWTP). The impact on eutrophication was about 25 and 19 times as large for the blackwater and the urine scenarios as for the NP scenario (Fig. 5).

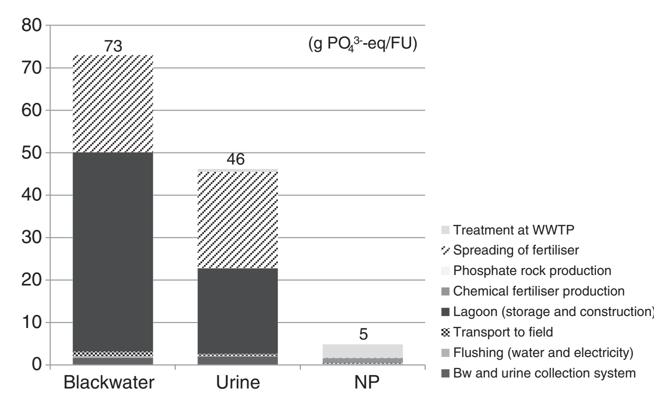

Fig. 5. Potential eutrophication for the three scenarios divided between processes. Total potential eutrophication is also given, as a number above each bar. Bw = blackwater.

The ammonia emissions from storage and spreading of blackwater and urine contributed most to eutrophication. Due to the longer storage time of blackwater compared with urine, the blackwater scenario caused greater ammonia emissions and thus greater eutrophication potential than the urine scenario. About 99% of the potentially eutrophying emissions from the storage and construction process derived from direct ammonia emissions at storage in the blackwater and urine scenarios. In the NP scenario the treatment at the WWTP was the main contributor to potential eutrophication, mainly from the electricity use and methanol production.

4.4. Acidification

Ammonia emissions also contributed to potential acidification and therefore the results for potential acidification (Fig. 6) showed a similar pattern to those of potential eutrophication (Fig. 5). The acidifying emissions were about 8 and 5 times as large for the blackwater and urine scenarios, respectively, as for the NP scenario. The ammonia emissions from storage and after spreading were the largest contributors to acidification for the blackwater and urine scenarios. For the NP scenario, the production of chemical fertilisers and treatment at the WWTP were the main sources of potentially acidifying emissions.

4.5. Use of phosphate rock, cadmium flow to soil and potential carbon sequestration

The blackwater scenario had by far the largest flow of cadmium to the soil (Fig. 7), about four times as large as in the NP scenario and 8 times as large as in the urine scenario. In the urine scenario, 0.05 kg phosphorus deriving from phosphate rock was needed to fulfil the FU and this contributed about 0.16 mg cadmium per FU, and thus more than the urine itself, which contributed two-thirds of the phosphorus flow. In the NP scenario, all of the phosphorus added to soil, 0.15 kg, derived from phosphate rock, resulting in a cadmium flow of 0.5 mg per FU.

The flow of carbon to soil with faeces and toilet paper in the blackwater scenario was 2.7 kg. This resulted in carbon storage of 0.2 kg per FU in the blackwater scenario, corresponding to a reduction of 0.7 kg CO_2-equivalents in the blackwater scenario. This represented a decrease of 7% in the GWP results of the blackwater scenario.

4.6. Normalisation and sensitivity analysis

The normalisation showed that the emissions at spreading and storage in the blackwater and urine scenarios contributed most to potential eutrophication and potential acidification and are thus of relevance in a wider perspective (Table 3).

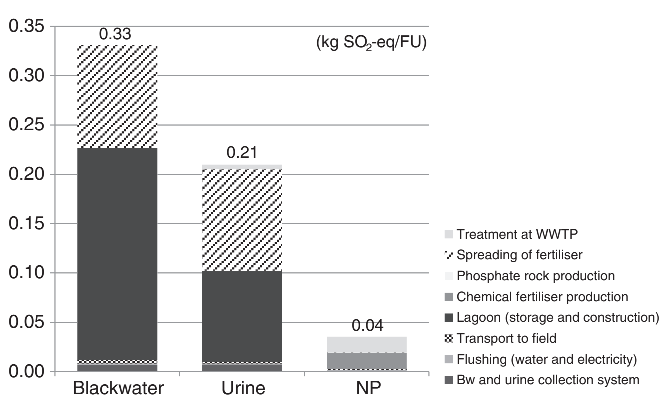

Fig. 6. Potential acidification for the three scenarios divided between processes. Total potential acidification is also given, as a number above each bar. Bw = blackwater.

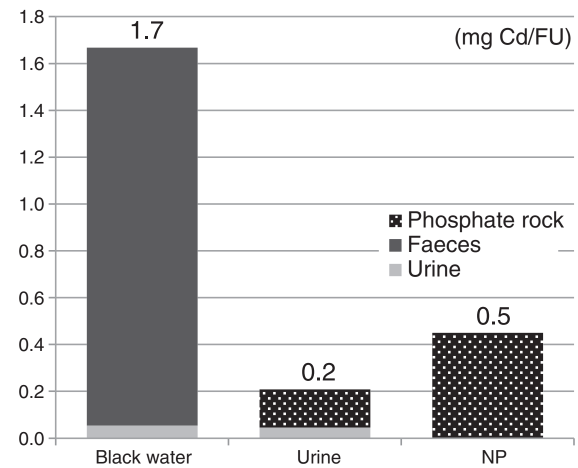

Fig. 7. Flow of cadmium to arable soil for the three scenarios. Total cadmium flow is given as the number above each bar.

For the blackwater scenario, decreasing the lifetime of the additional infrastructure by 50% in the sensitivity analysis had the largest impact on primary energy use, while doubling the ammonia emissions after spreading increased the eutrophication and acidification impact by almost 50% (Table 4). For the urine scenario, the impact on primary energy of halving the infrastructure lifetime was much less, 7%, while doubling the ammonia emissions after spreading increased eutrophication and acidification by more than 50%. For the NP scenario, the impact of the electricity mix was large. If the electricity came from coal power the impact on potential eutrophication would almost double and potential acidification would be almost 3.5 times as high as with Nordic electricity mix. If the phosphorus from excreta can be fully removed using just the average precipitation chemical dosage at the WWTP, and not the marginal, this would greatly decrease all impact categories studied, by 8–26%. If the nitritation–anammox process were to be used at the WWTP, with reduced energy use and no organic carbon source needed, the primary energy use would be reduced by 17 and 45%, respectively, for the urine and the NP scenarios and the other categories in the NP scenario would be reduced by 12–28%. Doubling the amount of nitrous oxide emitted at the WWTP would increase the GWP by 22% in the NP scenario.

5. Discussion

The two main processes affecting the overall results were the ammonia emissions from storage and spreading in the blackwater and urine scenarios and the enhanced treatment of nitrogen and phosphorus at the WWTP in the urine and NP scenarios. Compared with the blackwater scenario, the urine scenario had a smaller impact within all impact categories studied except energy use. These smaller impacts for the urine scenario were mainly due to the smaller volume handled and the shorter storage time and the larger energy use due to the nutrient removal in the WWTP. The NP scenario had a greater impact than both the urine and blackwater scenarios for GWP and primary energy use, but a much smaller impact for potential eutrophication and potential acidification.

The potential eutrophication in this study is mainly due to gaseous ammonia emissions and cannot be directly compared with directly eutrophying emissions to water, as not all of the potential eutrophication will actually cause negative eutrophication in the end. As essentially all the ammonia emissions occur in rural areas, from the storage lagoons by the fields and after spreading, and as ammonia is mainly a regional pollutant (Asman et al., 1998), a large fraction of it will be redeposited on agricultural fields, where eutrophication is seen as

Table 3
Normalisation of results on energy, GWP, potential eutrophication and potential acidification for the three scenarios.

	Primary energy (MJ)[a]	GWP (kg CO_2-eq)[b]	Eutrophication (kg PO_4^{3-}-eq)[c]	Acidification (kg SO_2-eq)[d]
Blackwater	0.1%	0.3%	1.3%	3.7%
Urine	0.1%	0.3%	0.8%	2.3%
NP	0.3%	0.5%	0.1%	0.4%

[a] Total energy input per person in Sweden based on 2011 data (Statistics Sweden, 2012c).
[b] Total GWP emissions in Sweden based on 2011 data (SEPA, 2013b).
[c] Total eutrophying emissions based on estimated amount of nitrogen and phosphorus reaching the Swedish coast in 2010 (Miljömål, 2013), total ammonia emissions in 2009 (SEPA, 2011) and emissions of nitrogen oxides (Miljömål, 2013).
[d] Total acidifying emissions in Sweden, based on emissions of nitrogen oxides and sulphur dioxide in 2009 (Miljömål, 2013) and ammonia emissions in 2009 (SEPA, 2011).

increased fertility, rather than as a negative impact. In this study the scenarios were designed such that the direct emissions to water from the WWTP were the same for all scenarios. This was achieved by advanced wastewater treatment, requiring different amounts of energy and chemicals to remove the incoming load of nitrogen and phosphorus from the different scenarios. The study illustrates the impacts of the improvements required at WWTPs to fulfil the commitments of the Baltic Sea Action Plan (HELCOM, 2011). Without these enhanced requirements on the WWTPs being a part of the FU, primary energy use and GWP would have been more similar for all scenarios, at the expense of different eutrophication impacts from the direct water emissions from the WWTPs. This is also the main reason for the difference in results compared with other studies comparing the environmental impact of urine separation and blackwater systems versus wastewater treatment at a WWTP (Benetto et al., 2009; Remy and Jekel, 2008; Tidåker et al., 2007a) and compared with small-scale treatment with precipitation chemicals and urea added to a septic tank (Tidåker et al., 2007b).

Ammonia emissions from storage and after spreading heavily influenced the results for eutrophication and acidification and should be further scrutinised. When emissions related to spreading were halved in the sensitivity analysis, potential eutrophication and acidification were reduced by 11 and 13%, respectively, in the blackwater scenario and by 26% for both categories in the urine scenario (Table 4). Field trials have shown that ammonia emissions after spreading of human urine to growing cereals, or by direct injection of the urine, may be scarcely detectable (Rodhe et al., 2004). A proper spreading strategy is thus imperative to reduce the losses associated with application. This would have a great impact on potential eutrophication and potential acidification, as can be seen in the sensitivity analysis (Table 4). If better techniques to reduce ammonia emissions from storage could also be developed, e.g. use of a gas proof lining, potential eutrophication and potential acidification could be further decreased. A well-studied technique is also acidification of manure, where pH is reduced by addition of e.g. sulphuric acid, which has been shown to decrease the emissions of ammonia at storage and after application significantly (e.g. Kai et al., 2008). The importance of these ammonia emissions can also be seen in the normalisation of the results (Table 3), where mainly eutrophication and acidification were of significant importance when comparing normalised per capita emissions.

Due to the much larger cadmium content of the faeces than in the other fertilisers, the blackwater scenario added significantly larger amounts of cadmium to the soil and the urine scenario added the least, about 50% of that added in the NP scenario (see Fig. 7). If the cadmium content of the phosphate rock had been higher, e.g. 87 mg per kg phosphorus as is the European average for phosphorus fertilisers (Nziguheba and Smolders, 2008), the NP scenario would add about eight times as much cadmium as the blackwater scenario and about three times as much as the urine scenario, even though the urine scenario also needed some phosphate rock to be added. These results stress that urine is an exceptionally clean fertiliser, with a cadmium content of 0.6 mg cadmium per kg phosphorus (compared with 11 mg for blackwater).

The blackwater scenario recycled the largest amount of phosphorus and organic material to the soil. Soil organic matter is valuable for keeping a soil fertile. However, the potential carbon sequestration in soil had a relatively small impact on the overall GWP results (see Section 4.5). In addition, urine and faeces also add micronutrients important for crop growth.

As the human excreta fractions were assumed to be stored according to WHO (2006) for hygienically safe use on agricultural land, hygiene risks with these scenarios should be insignificant, provided that the fertilisers are handled in a proper way. The environmental risk of pharmaceuticals spread on arable land is still something that has to be further studied before any conclusions can be drawn on the actual effect. Over 80 pharmaceutical compounds have been found over the level of micrograms per litre in sewage effluent, surface waters and

Table 4
Results on primary energy, GWP, potential eutrophication and potential acidification from the sensitivity analysis, presented in relation to the whole scenario.

	Primary energy (MJ/FU)		GWP (kg CO_2-eq/FU)		Eutrophication (g PO_4^{3-}-eq/FU)		Acidification (kg SO_2-eq/FU)	
Blackwater scenario (BW)	59		9.9		73		0.33	
Infrastructure life 50% shorter (BW)	78	+33%	11.3	+14%	74	+1%	0.33	+1%
+100% NH_4-emission spreading (BW)	62	+6%	10.6	+7%	104	+43%	0.47	+43%
−50% NH_4-emission spreading (BW)	58	−1%	9.8	−1%	65	−11%	0.29	−13%
Urine scenario (U)	67		9.4		46		0.21	
Infrastructure life 50% shorter (U)	71	+9%	10.7	+13%	47	+2%	0.21	+1%
+100% NH_4-emission spreading (U)	67	0%	10.4	+9%	72	+57%	0.33	+57%
−50% NH_4-emission spreading (U)	67	0%	9.4	−1%	34	−26%	0.16	−26%
Electric. WWTP from coal power (U)	70	+5%	9.8	+3%	47	+3%	0.21	+2%
Average P chemical WWTP (U)	52	−22%	9.0	−5%	46	−1%	0.21	−2%
Anammox process WWTP (U)	56	−17%	9.2	−2%	46	0%	0.21	0%
Increased N_2O WWTP (U)			9.9	+4%				
NP scenario (NP)	219		16.0		5		0.04	
Electric. WWTP from coal power (NP)	247	+13%	22.3	+40%	17	+244%	0.07	+93%
Average P chemical WWTP (NP)	179	−18%	14.7	−8%	4	−24%	0.03	−26%
Anammox process WWTP (NP)	121	−45%	13.9	−13%	3	−28%	0.03	−12%
Increased N_2O WWTP (NP)			19.6	+22%				

groundwater (Heberer, 2002). Source separation and treatment can be one way to control this problem (Larsen et al., 2004; Lienert et al., 2007). While the hormone flow with animal manure spread on land is far larger than the flow with wastewater, the problems are considered smaller, probably because the degradation of sex hormones is fast in arable land (Colucci et al., 2001; Colucci and Topp, 2001).

Recent studies have investigated nitrous oxide emissions from the removal of nitrogen at WWTPs. There are still uncertainties on the level of these emissions, as they depend on the equipment and technique used. With the emission factor of 1% used in this study, they cause about 25% of the GWP emissions of the NP scenario, but as shown in Foley et al. (2011) and Westling (2011) the emissions can vary. On doubling the nitrous oxide emissions, the GWP increased by 4% for the urine scenario and 22% for the NP scenario (Table 4). One process that was not included in this study was the potential use of sludge produced from the WWTP. This could be used as a fertiliser on arable land, replacing chemical fertilisers, and would also cause emissions from storage and spreading. While source separation would naturally decrease the amount of phosphorus in the sludge, it is not clear how source separation would affect either the total amount of sludge or the amount of nitrogen in the sludge. The amount might increase due to a decreased need for nitrogen reduction (Wilsenach and van Loosdrecht, 2003), or it might decrease due to less COD entering the WWTP.

When the life time of the additional infrastructure was reduced by half, the impact on GWP and primary energy use increased significantly in the urine and blackwater scenarios, but they still did not exceed the impacts of the NP scenario (Table 4). Disposal of materials had an insignificant impact on the results, since the majority of the infrastructure included in the study was assumed to be left on-site, and thus not disposed of. In this study a tractor with a mega trailer with a maximum cargo load of 33 tonnes was assumed to be used for transporting the toilet fractions from the households to the field. If the vehicle had been smaller, the relative impact per FU would have been much higher, as less volume could be transported per route. Use of European electricity mix by the WWTP resulted in significant increases in all impact categories for the NP scenario (Table 4), although the potential eutrophication and acidification were still lower than for the blackwater and urine scenarios.

If urine and blackwater are to be sustainable fertiliser alternatives, technological development is required. A system resulting in emissions of 1–4% of the annual emissions per person of potential acidification and eutrophication is not yet sustainable.

6. Conclusion

Source separation and use of human excreta as fertiliser proved to be more energy efficient and caused less global warming impact than enhanced reduction of nitrogen and phosphorus at a wastewater treatment plant, complemented with use of chemical fertiliser. However, the technologies assumed here for use of blackwater and urine as fertiliser caused larger emissions of ammonia, with great impacts on the potential eutrophication and potential acidification, especially when blackwater was recycled as fertiliser instead of being treated at an advanced wastewater treatment plant. If small flows of cadmium to arable soil are important, then fertilising with source-separated urine should be seriously considered, as its degree of contamination with heavy metals is extremely low. For systems with source-separation and recycling of toilet fractions as fertilisers, lower ammonia emission technologies need to be further developed to support more sustainable food production, based on a closed loop of nutrients between food and excreta.

Conflict of interest

This study was financially supported by the research fund Swedish Research Council Formas. J. Spångberg, H. Jönsson and P. Tidåker state that there are no conflicts of interest.

Acknowledgement

The financial support provided by the Swedish Research Council Formas (220-2007-1683) for this study is gratefully acknowledged.

Appendix A

Data used for materials and processes included in the study are given in Table A.

Table A
Materials, processes and references for emissions connected to the blackwater and urine scenarios. Amounts are for blackwater scenario/urine scenario, respectively.

Material/process	Type material/process used	Amount (per FU)	Reference emissions
Pipes	Polypropylene, granulate, at plant	0.035/0.046 kg	Ecoinvent[a]
Collection tank	Concrete, normal, at plant	7.024/9.304 kg	Ecoinvent[a]
	Reinforcing steel, at plant	0.144/0.207 kg	Ecoinvent[a]
Excavation	Excavation, hydraulic digger	0.010/0.014 m^3	Ecoinvent[a]
Tap water use	Electricity and chemical consumption as for tap water production Stockholm	379/663 m^3	Stockholm Vatten (2012b); Ecoinvent[a]
Emptying of collection tank	Diesel consumption pumping	0.059/0.028 kg	Baky et al. (2010)
	Emissions pumping		NTM (2013)
Lagoon			
Inside cover	HDPE, granulate, at plant	0.039/0.016 kg	Ecoinvent[a]
Floating cover	PVC, at regional storage	0.028/0.011 kg	Ecoinvent[a]
Loading area	Concrete, normal, at plant	0.240/0.133 kg	Ecoinvent[a]
Excavation	Excavation, hydraulic digger	0.023/0.009 m^3	Ecoinvent[a]
Emptying of lagoon	(The same as for emptying collection tank)		
Spreading			
of blackwater	Average of urine and semi-liquid manure, with Valtra 6650		Lindgren et al. (2002)
of urine	Spreading of urine, with Valtra 6650		Lindgren et al. (2002)
of phosphate rock	16% of combi-drilling, with Valtra 6660		Lindgren et al. (2002)
of NP fertiliser	16% of combi-drilling, with Valtra 6660		Lindgren et al. (2002)
Tractor	Tractor, production	0.006/0.002 kg	Ecoinvent[a]
Spreading equipment	Agriculture machinery, general, prod.	0.014/0.006 kg	Ecoinvent[a]

[a] Ecoinvent Centre (2010).

Appendix B

Data on transports included in the study are given in Table B.

Table B
Transport included in the study.

Transport	Transport mean	Load factor (%)	Distance (km)	Data source for emissions
Pipes				
China-Rotterdam	Transoceanic freight ship	65	19,500	Ecoinvent[c]
Rotterdam-hh[a]	Lorry 16–32 t, EURO5	Ecoinvent[b]	1,120	Ecoinvent[c]
Collection tank				
Cement, general	Lorry 16–32 t, EURO5	b	50	Ecoinvent[c]
Concrete, general	Lorry 16–32 t, EURO5	b	15	Ecoinvent[c]
Reinforcing steel				
Mo i Rana-Halmstad	Transoceanic freight ship	65	1,800	Ecoinvent[c]
Halmstad-hh[a]	Lorry 16–32 t, EURO5	b	500	Ecoinvent[c]
Transport hh[a] to farm	Tractor + megatrailer <30 t, EURO3	50	30	NTM (2013)
From farm to field	Included in the data on spreading			
Chemical fertiliser				
Uusikaupuuki-Åhus	Transoceanic freight ship	65	700	Ecoinvent[c]
Åhus-Växjö	Lorry 16–32 t, EURO5	b	160	Ecoinvent[c]
Växjö-farm	Lorry 16–32 t, EURO5	b	430	Ecoinvent[c]
Phosphate rock				
Kola peninsula-farm	Lorry 16–32 t, EURO5	b	1,600	Ecoinvent[c]

[a] hh = household.
[b] Load factor calculated by the model TREMOVE based on European data (Ecoinvent Centre, 2010).
[c] Ecoinvent Centre (2010).

References

Andersson Å, Jensen A. Flows and composition of greywater, urine, faeces and solid biodegradable waste in Gebers (in Swedish) (Master thesis) Swedish University of Agriculture. Department of Agricultural techniques; 2002.

Asman WAH, Sutton MA, Schjörring JK. Ammonia: emission, atmospheric transport and deposition. New Phytol 1998;139:27–48.

Baky A, Sundberg M, Brown N. Kartläggning av jordbrukets energianvändning. Ett projekt utfört på uppdrag av Jordbruksverket. Uppsala: JTI – Institutet för jordbruks- och miljöteknik; 2010.

Benetto E, Nguyen D, Lohmann T, Schmitt B, Schosseler P. Life cycle assessment of ecological sanitation system for small-scale wastewater treatment. Sci Total Environ 2009; 407:1506–16.

Bernstad A, la Cour Jansen J. Review of comparative LCAs of food waste management systems— current status and potential improvements. Waste Manag 2012;32:2439–55.

Brentrup F, Pallière C. GHG emissions and energy efficiency in European nitrogen fertiliser production and use. The International Fertiliser Society. Proceeding; 2008. p. 639.

Colucci MS, Topp E. Persistence of estrogenic hormones in agricultural soils: II. 17 α-etonylestradiol. J Environ Qual 2001;30:2077–80.

Colucci MS, Bork H, Topp E. Persistence of estrogenic hormones in agricultural soils: I. 17 β-estradiol and estrone. J Environ Qual 2001;30:2070–6.

Cordell D, White S. Peak phosphorus: clarifying the key issues of a vigorous debate about long-term phosphorus security. Review. Sustainability 2011;3:2027–49.

Davis J, Haglund C. Life cycle inventory (LCI) of fertiliser production: fertiliser products used in Sweden and Western Europe. SIK report 654. Göteborg, Sweden: The Swedish Institute for Food and Biotechnology; 1999.

DCCEE (Department of Climate Change and Energy Efficiency). National greenhouse and energy reporting system measurement. Technical guidelines for the estimation of greenhouse gas emissions by facilities in Australia. Canberra: Commonwealth of Australia; 2010 [June].

EC (European Council). Council Regulation (EC) no 834/2007 of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) no 2092/91; 2007.

EC (European Council). Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain directives; 2008a.

EC (European Council). Commission regulation (EC) no 889/2008 of 5 September 2008 laying down detailed rules for the implementation of Council Regulation (EC) no 834/2007 on organic production and labelling of organic products with regard to organic production, labelling and control; 2008b.

Ecoinvent Centre. Ecoinvent data v2.2. Final reports no. 1–25. Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories; 2010.

EEC (European Economic Community). Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture; 1986.

Fertilizers Europe. EU fertilizer market— key graphs. Available from www.fertilizerseurope.com/fileadmin/user_upload/publications/statistics_publications/Stat_website.pdf, 2013. [07–31].

Fidjeland J, Magri ME, Jönsson H, Albihn A, Vinnerås B. The potential for self-sanitisation of faecal sludge by intrinsic ammonia. Water Res 2013;47(16):6014–23.

Foley J, Yuan Z, Keller J, Senante E, Chandran K, Willis J, et al. N_2O and CH_4 emission from wastewater collection and treatment systems. Technical report. London, UK: Global Water Research Coalition; 2011.

Frausto da Silva JJR, Williams RJP. The biological chemistry of the elements — the inorganic chemistry of life. Oxford: Oxford University Press; 1997.

Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, et al. Handbook on life cycle assessment: operational guide to the ISO standards. Dordrecht: Kluwer Academic Publishers; 2002.

Hansen TL, Bhander GS, Christensen TH. Life cycle modelling of environmental impacts of application of processed organic municipal solid waste on agricultural land (EASEWASTE). Waste Manag Res 2006;24:153–66.

Heberer T. Occurrence, fate, and removal of pharmaceutical residues in the aquatic environment: a review of recent research data. Toxicol Lett 2002;131:5–17.

HELCOM. The fifth Baltic Sea pollution load compilation (PLC-5) Balt Sea environ proc no. 128; 2011.

HELCOM. Declaration highlights. Reaching good environmental status for a healthy Baltic SeaMinisterial meeting Copenhagen 3 October 2013; 2013.

IPCC. Agriculture, forestry and other land use. In: Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K, editors. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, vol. 4. Japan: IGES; 2006.

IPCC. Changes in atmospheric constituents and in radiative forcing. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, et al, editors. Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate ChangeCambridge, United Kingdom and New York, NY, USA: Cambridge University Press; 2007.

ISO (International Organization for Standardization). ISO 14040: 2006. Environmental management–life cycle assessment — principles and framework; 2006a.

ISO (International Organization for Standardization). ISO 14044: 2006. Environmental management–life cycle assessment — requirements and guidelines; 2006b.

IVL. Jämförande miljöbedömning av tre fällningskemikalier PIX-111, PIX-110 och PIX-118. Kemira AB; 2003.

Jönsson H, Vinnerås B, Höglund C, Stenström TA, Dalhammar G, Krichmann H. Källsorterad humanurin i kretslopp. VA-forsk rapport; 2000:1; 2000.

Jönsson H, Baky A, Jeppsson U, Hellström D, Kärrman E. Composition of urine, faeces, greywater and biowaste: for utilisation in the URWARE model. Urban water report; 2005:6. Chalmers University of Technology; 2005.

Kai P, Pedersen P, Jensen JE, Hansen MN, Sommer SG. A whole-farm assessment of the efficacy of slurry acidification in reducing ammonia emissions. Eur J Agron 2008;28:148–54.

Karlsson S, Rodhe L. Översyn av Statistiska Centralbyråns beräkning av ammoniakavgången i jordbruket – emissionsfaktorer för ammoniak vid lagring och spridning av stallgödsel. JTI Uppdragsrapport; 2002.

KEMI (the Swedish Chemicals Agency). Kadmiumhalten måste minska - för folkhälsans skull: En riskbedömning av kadmium med mineralgödsel i fokus. Rapport från ett regeringsuppdrag by the Swedish Chemicals Agency; 2008. [Sundbyberg].

Kirchmann H, Pettersson S. Human urine — chemical composition and fertilizer use efficiency. Fertil Res 1995;40:149–54.

Larsen TA, Lienert J, Joss A, Siegrist H. How to avoid pharmaceuticals in the aquatic environment. J Biotechnol 2004;113:295–304.

Lienert J, Bürki T, Escher BI. Reducing micropollutants with source control: substance flow analysis of 212 pharmaceuticals in faeces and urine. Water Sci Technol 2007;56(5):87–96.

Lindgren M, Pettersson O, Hansson P-A, Norén O. Jordbruks- och anläggningsmaskiners motorbelastning och avgasemissioner - samt metoder att minska bränsleförbrukning och avgasemissioner. JTI-rapport nr. 308; 2002.

Liney KE, Hagger JA, Tyler CR, Depledge MH, Galloway TS, Jobling S. Health effects in fish of long-term exposure to effluents from wastewater treatment works. Environ Health Perspect 2006;114(Suppl. 1):81–9.

Maurer M, Schwegler P, Larsen TA. Nutrients in urine: energetic aspects of removal and recovery. Water Sci Technol 2003;48(1):37–46.

Maurer M, Pronk W, Larsen TA. Treatment processes for source-separated urine. Water Res 2006;40:3151–66.

Miljömål. Environmental objectives portal (in Swedish). Sweden. Available from http://miljömål.se/sv/Miljomalen, 2013. [04-22].

NTM. Goods and logistics. Calculation tool from the network for transport and environment (in Swedish); 2013 [(last updated March 2010) Available from www.ntmcalc.com [08–13]].

Nziguheba G, Smolders E. Input of trace elements in agricultural soils via phosphate fertilizers in European Countries. Sci Total Environ 2008;390:53–7.

PRé Consultant. SimaPro 7.3.3 LCA software; 2012 [Amersfoort, Netherlands].

Randall CW. Potential societal and economic impacts of wastewater nutrient removal and recycling. Water Sci Technol 2003;48(1):11–7.

Remy C, Jekel M. Sustainable wastewater management: life cycle assessment of conventional and source-separating urban sanitation systems. Water Sci Technol 2008; 58(8):1555–62.

Rockström J, Steffen W, Noone K, Persson Å, Chapin S, Lambin E, et al. A safe operating space for humanity. Nature 2009;461:472–5.

Rodhe L, Richert Stintzing A, Steineck S. Ammonia emissions after application of human urine to a clay soil for barley growth. Nutr Cycl Agroecosyst 2004;68:191–8.

Schönning C, Stenström TA. Guidelines on the safe use of urine and faeces in ecological sanitation systems. EcoSanRes programme. Report 2004:1; 2004.

SEPA (Swedish EPA). Sveriges åtaganden i Baltic Sea Action Plan; Konsekvensanalyser. Report 5984; 2009.

SEPA (Swedish EPA). Emissions of ammonia to air in Sweden in 2009 (in Swedish). SM1101 ; 2011. SM1101, 2011.

SEPA (Swedish EPA). Rening av avloppsvatten i Sverige 2010 978-91-620-8629-9; 2013a.

SEPA (Swedish EPA). Växthusgaser - nationella utsläpp 1990–2011. Available from http://www.naturvardsverket.se/Sa-mar-miljon/Statistik-A-O/Vaxthusgaser-nationella-utslapp/, 2013. [03–12].

Siegrist H, Salzgeber D, Eugster J, Joss A. Anammox brings WWTP closer to energy autarky due to increased biogas production and reduced aeration energy for N-removal. Water Sci Technol 2008;57(3):383–8.

Statistics Sweden. Use of fertilisers and animal manure in agriculture in 2010/11 (in Swedish). SM1203 ; 2012a.

Statistics Sweden. Discharges to water and sewage sludge production in 2010. Municipal wastewater treatment plants, pulp and paper industry and other industry; 2012b [SM1201].

Statistics Sweden. Annual energy balance sheets 2010–2011 (in Swedish); 2012c. SM1206 (corrected version), 2012.

Stockholm Vatten. Miljörapport 2011. Stockholm Vatten VA AB; 2012a.

Stockholm Vatten. Verksamheten 2011. Årsberättelse. Stockholm Vatten AB; 2012b.

Syvab. Sjölunda avloppsreningsverk. Malmö. Miljörapport enligt Miljöbalken för år 2011; 2012.

Tidåker P, Mattsson B, Jönsson H. Environmental impact of wheat production using human urine and mineral fertilisers — a scenario study. J Clean Prod 2007a;15:52–62.

Tidåker P, Sjöberg C, Jönsson H. Local recycling of plant nutrients from small-scale wastewater systems to farmland — a Swedish scenario study. Resour Conserv Recycl 2007b;49:388–405.

UMEVA. Miljörapport 2011. Öns avloppsreningsverk. Umeå Vatten och Avfall AB; 2012.

UN (United Nations). World population prospects: the 2012 revision, highlights and advance tables. Department of Economic and Social Affairs, Population Division; 2013 [ESA/P/WP.228].

UNEP (United Nations Environment Program). Challenges to international waters — regional assessments in a global perspective. Nairobi, Kenya: United Nations Environment Programme; 2006.

Uppsala Vatten. Miljörapport 2011. Kungsängsverket. Uppsala Vatten och Avfall AB; 2012.

USGS. Phosphate rock. Statistics and informationU.S. Geological Survey, Mineral Commodity Summaries; 2013 [(Jan). Available from http://minerals.usgs.gov/minerals/pubs/commodity/phosphate_rock [2013-09-11]].

Vajda AM, Barber LB, Gray JL, Lopez EM, Woodling JD, Norris DO. Reproductive disruption in fish downstream from an estrogenic wastewater effluent. Environ Sci Technol 2008;42:3407–14.

Westling K. Lustgasemissioner från avloppsreningsverk - en litteraturstudie. IVL Svenska Miljöinstitutet AB. Report B1977; 2011.

WHO. WHO guidelines for the safe use of wastewater, excreta and greywater. Excreta and greywater use in agricultureWHO publications, vol. IV; 2006 [Geneva].

Wilsenach JA. Treatment of source separated urine and its effects on wastewater systems (Thesis) The Netherlands: Delft University of Technology; 2006.

Wilsenach JA, van Loosdrecht M. Impact of separate urine collection on wastewater treatment systems. Water Sci Technol 2003;48(1):103–10.

Winker M, Vinnerås B, Muskolus A, Arnold U, Clemens J. Fertiliser products from new sanitation systems: their potential values and risks. Bioresour Technol 2009;100:4090–6.

Wostman. Manual EcoVac vacuum toilet system. Available from http://wostman.se/PDF/vac_manual.pdf, 2013. [04–12].

Yara. Kadmiumhalterna fortfarande låga. Hyltén-Cavallius. I. Växtpressen No. 2; 2010.