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Department of Energy and Technology

Blackwater sanitization with urea in Sweden
– sanitization effect and environmental impact

Brenda Vidal Estévez

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SLU, Swedish University of Agricultural Sciences
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Author: Brenda Vidal Estévez

Main supervisor: Björn Vinnerås, Department of Energy and Technology, SLU
Co-supervisors: Annika Nordin, Department of Energy and Technology, SLU
Günter Langergraber, University of Natural Resources and Applied Life Sciences Vienna (BOKU), Institute of Sanitary Engineering and Water Pollution Control
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Abstract

The increasing global population is placing pressure on the water resources and nutrients needed for food production, while increasing the amount of waste generated and the environmental contamination. Technical solutions and changes in the social perception of the environmental problems are needed to address these problems.

The blackwater (wastewater from toilets only) represents the major proportion of the plant nutrients found in household wastewater (about 90 % of the phosphorous and nitrogen from the total effluent) and thus they should be recycled in order to close the nutrients loop in the environment. The aim of this research was to analyze the possible use of blackwater as fertilizer in agriculture by comparing three different scenarios: conventional wastewater treatment, separate storage with addition of urea 1 % and separate storage with addition of urea 0.5 % and heat from different sources. The methodology included a system analysis to evaluate the environmental impact of the treatments, considering primary energy use, electricity use and global warming potential. Furthermore, a sanitation part was carried out in the laboratory as a pilot study by monitoring of indicator organisms over time in blackwater treated with 1 % urea. The urea added *in situ* is degraded into ammonium, which has a sanitization effect due partly to the increment of the pH that consequently inactivates the pathogens. The results showed that the urea treatment was better than the conventional wastewater treatment both from environmental and sanitation perspective. The indicator organisms studied in the lab showed good inactivation rates.

The importance of this research relies on the possibility to minimize waste and environmental pollution, close the nutrients cycles by an efficient use of the available resources and, at the same time, decrease the demand of chemical fertilizers by the agricultural sector. In addition, an adequate sanitation process is ensured to reduce the hygienic risks associated.

Keywords: blackwater, fertilizer, LCA, Life Cycle Assessment, sanitation, urea treatment

Acronyms:

BAT: best available technology

BW: blackwater

CAN: calcium ammonium nitrogen (fertilizer)

FU: functional unit

GWP: global warming potential

LCA: life cycle assessment

MJ: mega joules

NPK: fertilizer based on nitrogen, phosphorous and potassium

TSP: triple superphosphate (fertilizer)

TTC: total thermotolerant coliform bacteria

WWTP: wastewater treatment plant

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1. Introduction

Wastewater, and human excreta in particular, has been regarded for a long time as a problem due to the hygienic hazards involved, instead of a resource that is always available in all societies (Langergraber and Muellegger, 2005; Heinonen-Tanski and van Wijk-Sijbesma, 2005). This misconception has resulted in the development of unsustainable sanitary systems (Magri *et al.*, 2013) characterized by a linear flow going from the use of sources to the production of the so called waste.

As an alternative to this conventional view, the ecological sanitation approach presents circular flows where the water and nutrient cycles are closed and moreover, the wastewater and the human excreta are conceived as resources instead of waste (Langergraber and Muellegger, 2005).

Innovative sanitation systems based on ecological sanitation technologies are currently being investigated and have been proved to be effective in reducing pathogens which are normally found in human excreta, providing health safety and decreasing the environmental pollution (Winker *et al.*, 2009).

Additionally, these new emerging ”waste” products generally show promising characteristics when used in agriculture, by improving the soil structure and fertility, enhancing productivity and providing plant available nutrients -therefore reducing the need for artificial fertilizers based mostly on non-renewable resources (Jönsson *et al.*, 2004; Stenström and Schönning, 2004; Winker *et al.*, 2009).

Blackwater (a liquid mixture of urine, faeces, toilet paper and flush water) is one of these wastewater products, with considerable amounts of carbon and high total nutrient recovery (Winker *et al.*, 2009). This fraction of the household waste requires treatment for stabilization and sanitization in order to achieve hygienically safe conditions (Vinnerås, 2002). Several treatment options have been studied to reduce pathogenic microorganisms, generally based on factors such as temperature, moisture, pH and ammonia content. Some sanitization treatments that can be used for blackwater treatment in a Swedish context are wet composting, anaerobic digestion and chemical sanitation based on ammonia (Nordin, 2010). The last one was the focus of this thesis.

2. General aim

The general aim of this thesis was to analyze the potential use of blackwater as fertilizer in agriculture by: monitoring indicator organisms and physicochemical parameters over time in blackwater treated with urea in a farm located in the municipality of Uddevalla (Part I. Sanitization); and evaluating the environmental impact of this treatment in comparison to the conventional wastewater treatment (Part II. Systems Analysis).

3. Literature review

3.1 Blackwater as fertilizer

Blackwater is considered to include the water collected from toilets, consisting of a mixture of urine, faeces, toilet paper and flush water (WHO, 2006). From chemical point of view, urine and faeces are high quality fertilizers that contain low levels of contaminants such as heavy metals. Faeces are rich in organic matter and macronutrients like potassium and phosphorus; urine is characterized by high nitrogen content, which is usually a limiting factor regulating the growth of plants (Jönsson *et al.*, 2004).

Due to this variety of plant-available nutrients, the human excreta have been used in agriculture for thousands of years all over the planet and by almost all cultures. Examples of it can be found especially in China and Southeast Asia (Heinonen-Tanski and van Wijk-Sijbesma, 2005; EcoSanRes, 2008a; Nordin, 2010).

The use of wastewater has increased in both developed and developing countries. The principal driving forces are related to the increase in global population, especially in urban and periurban areas in developing countries, using larger amounts of water and therefore producing greater amounts of wastewater in comparison to rural areas. The degradation of freshwater resources and the increase of water scarcity and stress also contribute to the usage of wastewater. Moreover, there is a major concern and recognition about the nutrients value of the wastewater and the possibility of using it as a resource (WHO, 2006a). The agricultural sector is known to be the largest user of freshwater, accounting for approximately 70% of all surface water supplies (Ongley, 1996). As the scarcity of freshwater increases due to population growth, effects from climate change and urban expansion, so does the use of wastewater in agriculture. As stated by the United Nations, water is a key to food security and it needs to be protected in order to ensure access to safe and nutritious food for a productive and healthy life. Other driving factor for the increasing use of wastewater is the Millennium Development Goals (MDGs), settled by the United Nations and adopted by the General Assembly in September 2000. Particularly the Goals 1 and 7 related to eliminating extreme poverty and hunger and ensuring environmental sustainability (United Nations, 2000).

The use of wastewater for agricultural purposes can cause positive and negative environmental impacts. Although there is always an inherent environmental risk associated, such risk can be minimized considerably with careful and controlled planning and management. The impact will depend on how the wastewater is used and treated, and how it interacts with the environment (WHO, 2006).

Besides the valuable plant nutrient content, human excreta may contain pathogenic microorganisms that could constitute a threat to human and animal health. Therefore an appropriate sanitation process is required before reuse in order to reduce the concentration of pathogens to a safe level (Schönning *et al.*, 2004).

3.2 Benefits and drawbacks of using blackwater as a fertilizer

The nutrients cycles in the environment are constantly being altered by agricultural activity. During the food production, a stream of nutrients is lost when supplying human society with food (Vinnerås, 2007), creating a linear, non-recycling and open-ended system (EcoSanRes, 2008b). To compensate this shortage, chemical fertilizers are usually added to the soil while, on the other hand, the nutrients from society are discharged as waste or contaminants into water recipients (Vinnerås, 2007). Toilet water represents the major proportion of the nutrients found in households (Vinnerås *et al.*, 2006) and thus they should be recycled in order to close the nutrients loop in the environment (Figure 1).

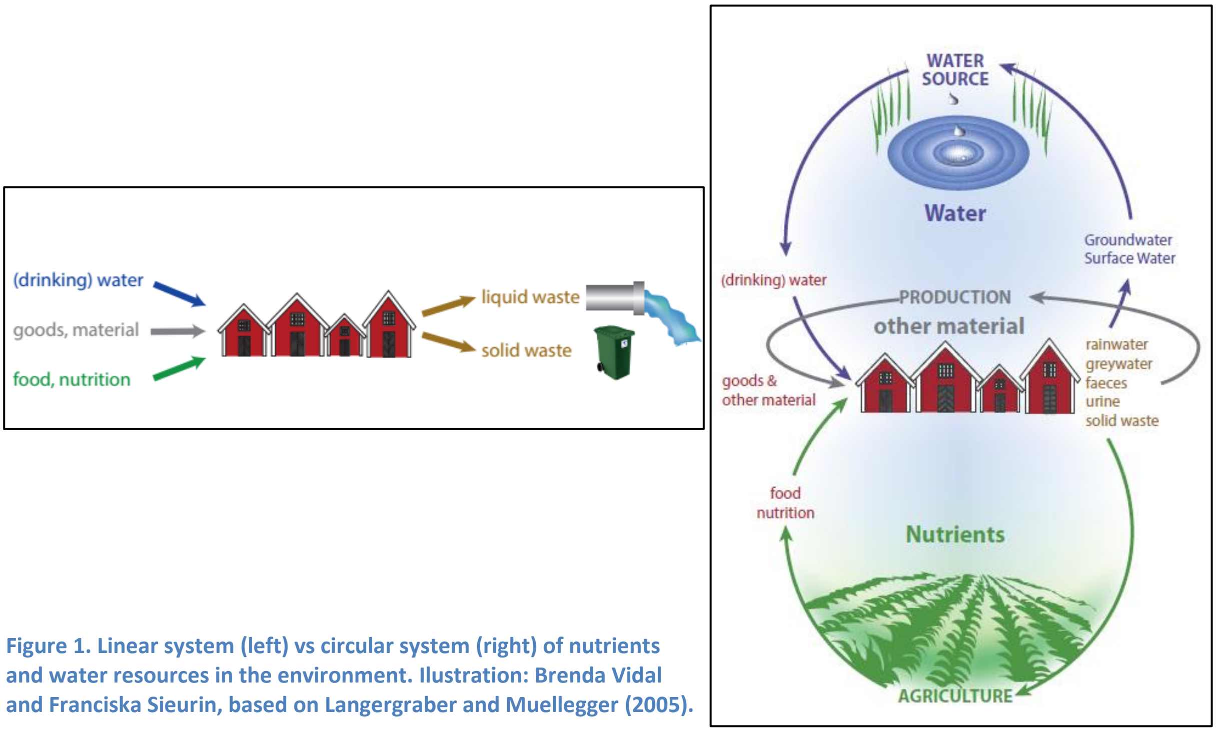

Figure 1. Linear system (left) vs circular system (right) of nutrients and water resources in the environment. Illustration: Brenda Vidal and Franciska Sieurin, based on Langergraber and Muellegger (2005).

In regions where **water scarcity** is a reality, recycling wastewater is sometimes the only feasible solution for farmers, especially in arid and semi-arid conditions. Additionally, the pressure on recipient water bodies decreases because direct discharges are minimized (WHO, 2006).

One of the main benefits of using blackwater is the **closing of the nutrients loop**. Human excreta is an important source of plant available macronutrients such as nitrogen, phosphorous and potassium and micronutrients such as copper, iron or zinc; the risk of deficiency of these nutrients in the fields when using human excreta is minimal as it contains all the elements needed for plant growth (Jönsson *et al.*, 2004).

For instance, urine from one person contains about 4.0 kg of nitrogen, 0.4 kg of phosphorous and 0.9 kg of potassium per year, representing about 88 % of the nitrogen, 64 % of the phosphorous and 73 % of the potassium of the total excretion of a person. Faecal material contains about 0.5 kg of nitrogen and 0.18 kg of phosphorous. Even though the toilet fraction represents only a small part of the total effluent of a house, it contains about 90 % of the phosphorous and nitrogen and 80 % of the potassium in the total household drain, as well as other micronutrients that may be important for plants uptake (Hjelmqvist *et al.*, 2012). The existing knowledge about the nutrients plant availability is still limited. When estimating the expected nutrients availability it is important to know the mineral and organic fraction of the nitrogen. According to Winker *et al.* (2009), the expected nutrient availability is 100 % for the mineral nitrogen and 10 % the organic nitrogen in the year of application; for phosphorous and potassium, the expected availability is 100 % for a crop rotation of three years.

Considering these numbers, the use blackwater ought to decrease the use of mineral fertilizers since a certain part of the demand would be covered by the nutrients contained in the blackwater. This would result in less energy needed to produce fertilizers and fewer sources of non-renewable resources being exploited (WHO, 2006).

This fact gains even more importance when referring to **phosphorous**. Due to its chemical properties, much of the phosphorous in soil is not available for plants and hence it must be added additionally in agriculture. As a non-renewable resource and because of the high rate it has been mined, the projections about its irreversible depletion are debated. Since the vast majority of phosphorus (around 90 %) is used for food and feed production (Cordell *et al.*, 2010) and normally the same amount that is consumed by animals and humans is excreted (EcoSanRes, 2008b), it is reasonable to recover this precious nutrient safely and return it back to the nutrient loop to decrease the need of mined phosphorous in agriculture. For the case of the nitrogen, the atmosphere contains approximately 79 % nitrogen which, in order to be used by plants, needs to be fixed by specific nitrogen fixing microorganisms. Both industrial and biological nitrogen production and fixation require large amounts of energy (Heinonen-Tanski and van Wijk-Sijbesma, 2005).

However, the use of excreta is not exempt from constrains. The major constrain for using toilet water as a fertilizer in agriculture is the presence of **excreta-related pathogens** (Winker *et al.*, 2009). Excreta-related diseases are caused by infective agents such as bacteria, viruses and parasites like protozoa and helminths that are released through excreta (faeces and urine) of infected people (WHO, 2006) that do not necessarily manifest clinical symptoms of infection. In the case of urine, it is normally sterile in the urinary bladder although some microorganisms can be collected along the urinary tract (Schönning *et al.*, 2004). A content of 10^3 organisms per ml of urine does not indicate an infection and these saprophytic organisms are considered generally harmless (Schönning *et al.*, 2004). The main case of enteric pathogens in urine is faecal contamination in the toilet. Faecal material normally contains large amounts of microorganisms, in the range of 10^{11}-10^{13} per gram (Heinonen-Tanski and van Wijk-Sijbesma, 2005; Schönning *et al.*, 2004), including many opportunistic pathogens. The likelihood or risk of being infected is a function of exposure; the susceptibility of the individuals and the infective dose of the pathogen. Some pathogens are important as zoonotic agents, they can be transmitted between animals and humans and their excreta, infecting a wide range of species (Schönning *et al.*, 2004).

In general terms, the leading agent causing gastrointestinal illness is **bacteria**, especially in developing countries. Bacterial pathogens found in excreta include enterohaemorrhagic *E. coli* (EHEC), *Leptospira interrogans*, *Campylobacte*r spp., *Yersinia* spp. and *Salmonella* spp. In regions lacking appropriate sanitation, *Salmonella typhi*, and *paratyphi* (causing typhoid fever), *Vibrio cholera* (causing cholera) and *Shigella* (causing diarrhea) are commonly found and constitute serious risk regarding contamination of water and gastrointestinal illness (Schönning *et al.*, 2004; WHO, 2006; Nordin, 2010).

Regarding **viruses**, in developed countries it is considered the main agent causing gastrointestinal-related illnesses, being excreted more commonly in faeces than in urine. It has been estimated that more than 120 different types of viruses can be found in faeces (WHO, 2006). Hepatitis A, Norovirus, parvovirus, rotavirus, enterovirus and enteric adenovirus are the most frequent groups, causing symptoms such as diarrhea, vomiting, enteritis, fever and stomach cramps (Schönning *et al.*, 2004; WHO, 2006).

Parasites, on the other hand, are an important cause of enteric diseases in developing countries. Schistosomiasis constitutes a major parasitic infection affecting humans in Africa. It is caused by *Schistosoma* spp. a helminth with indirect life cycle that depending on the subspecies can be excreted either by the urine or by the faeces (Nordin, 2010). The transmission is requiring the presence of freshwater sources where the aquatic larvae can emerge from the intermediate host, fresh water snail (Schönning *et al.*, 2004). Other helminth infections are caused by the ingestion of the eggs of *Ascaris* and *Taenia,* both infections responsible for exaggerating malnutrition and indirectly provoking vulnerability to other infections. Hookworms infect humans and animals by the larvae, not the eggs, which usually penetrates the skin and migrates to the intestine (Hotez *et al,.* 2005). Protozoa organisms are characterized by a considerable resistance in the environment and low infectious doses; *Entamoeba histolytica* causes the infection Amoebiasis, and *Cryptosporidium parvum* and *Giardia lamblia/intestinalis* are zoonotic agents associated with water-borne outbreaks (Schönning *et al.*, 2004; WHO, 2006; Nordin, 2010).

Pathogenic organisms can be transmitted by different routes integrated in the so-called F-diagram as it is referred in the literature; the pathogens excreted in faeces would be transmitted to the food or reach the face of the individuals by being in contact with fingers, flies, fluids or through the fields (agriculture). Apart from the faecal-oral route, possible infection through the skin should be also considered. Preventive barriers include personal hygiene methods such as washing hands and keeping the facilities clean, treatment of the sources and protection of the water prior disinfection (Schönning *et al.*, 2004; Nordin, 2010).

With regard to **heavy metals**, their concentration is generally low or very low in human excreta, since it reflects the amount present in consumed products (Jönsson *et al.*, 2004, Vinnerås 2007). Moreover, the concentration of heavy metals is usually lower than in chemical fertilizers and farmyard manure; often only one tenth as high in comparison with animal manure (Jönsson *et al.*, 2004, Winker *et al.*, 2009). The heavy metals found will only marginally be absorbed by plants until they are in a mobile phase and reach a threshold concentration in the soil (WHO, 2006b).

Organic micropollutants such as **pharmaceutical residues and hormones**, as well as substances originating from cleansing agents are considered one of the limitations of using blackwater as fertilizers (Winker *et al.*, 2009; Nordin, 2010). However, when compared to German farmyard manure, the antibiotics in human excreta constitute 1 % of that in manure (Winker *et al.* 2009) and hormones show the same trend. All mammals produce hormones that have been excreted in terrestrial environments during the course of evolution, thus the terrestrial microorganisms are adapted to these substances and are able to degrade them. Considering that most of the pharmaceutical compounds have their origin in nature, the diversity of the microbial community can degrade them and reduce the risks associated, especially in the active topsoil where the blackwater-derived fertilizers would be applied (Jönsson *et al.*, 2004). In ordinary wastewater treatment plants, these natural processes have been simulated and its efficiency verified. But due to the short retention time, many medical substances are not degraded before they are discharged into the water recipients. It is more likely that the terrestrial ecosystems are more used to mammal hormones exposure in comparison to the aquatic ones, and therefore the environment shows more resilience in the first case. Furthermore, the risk from pharmaceutical substances in agricultural systems is small in comparison to the amount in manure from domestic animals or to pesticides applied in the fields (Jönsson *et al.*, 2004). Yet some endocrine disruptors (17α-estradiol, estriol and testosterone) are not sensitive to photodegradation and thus may not degrade quickly and remain in the topsoil. More research is needed to determine the persistence and effects of pharmaceuticals and hormones in the environment (WHO, 2006a).

Another drawback that might influence the decision of whether using or not human excreta as fertilizer is the **social factor**. Regarding the use of sludge in agriculture there is a wide range of actors involved; the political system and public authorities, the water and wastewater sector, the farmers, the food industry, the consumers and the environmental organizations. The scientific information is appreciated and used by all the groups involved when discussing the issue, although they might use it and interpret it in different ways (Bengtsson and Tillman, 2004).

Generally speaking, farmers organizations and food industry are reluctant towards the use of sludge in agriculture due to the potential risk of containing unwanted substances such as heavy metals, organic pollutants or pathogens that could comprise the confidence of the consumers (Berglund, 2001; Tidåker *et al.*, 2006). Therefore, most of the Swedish companies prefer not to buy products produced in farms where sludge is used (Tidåker *et al.*, 2007).

Nevertheless, and despite the controversy, the use of sludge or waste material is still in the spotlight due to the necessity to fulfill the environmental goals about nutrients recycling and sustainability, the economic perspective and the simplicity of sludge handling in comparison to the costs of the possible alternatives (Bengtsson and Tillman, 2004). The Federation of Swedish Farmers (LRF) only accepts REVAQ certified sludge since 2008. REVAQ is a certification system developed by the Swedish Water and Wastewater Association that aims to reduce the flow of hazardous substances into sewage treatment plants and to create a sustainable and safe recycling of nutrients in the sludge to agriculture (Svenskt Vatten, 2012).

Urine sorting and blackwater sorting systems result in at site collected sewage fractions which are considered as alternatives to sludge for reuse of nutrients to arable land. Farmers are raising their interest in high quality fertilizers, preferably concentrated products since handling large volumes of water can be problematic. The Federation of Swedish Farmers has declared their support to source-separated material and the intention to work for an active circulation of plants nutrients from household wastewater (LRF, 2012).

3.3 Sanitation systems

In a sustainable society, the nutrients that are contained in the wastewater should be recycled (Vinnerås, 2007; Swedish EPA, 2009). Nowadays, the vast majority of households from urban areas in Sweden are connected to municipal sewer networks and treatment plants, whereas holidays homes in rural areas usually have their own stand-alone system for wastewater disposal which must follow the general recommendations established by the Swedish EPA regarding local installations in small scale. It is estimated that around 750,000 properties are not connected to municipal wastewater treatment plants in the whole country (Swedish EPA, 2009).

In general terms, **conventional Swedish wastewater treatment plants** usually combine several treatments (Figure 2) intended to remove solids, nutrients and microorganisms from the water phase, but not made to dispose of hazardous substances. The treatment normally starts with a mechanical removal of large solids by means of screening, sand trap and a pre-sedimentation process. Next step is the biological treatment where approximately 90 % of the organic matter is Swedish EPAremoved by microorganisms feeding on it and around 20 % of the nitrogen is consumed. The solid-phase is separated trough the activated sludge process. In plants designed for nitrogen removal, especially in large scale plants, nitrogen is removed during this biological phase at a rate of 50-70 % with a combination of aerobic/anoxic conditions. The following stage is the chemical treatment, where phosphorous is removed at a rate of 90-98 % by precipitating chemicals and further flocculation/sedimentation processes. At the end, in some plants the water is filtered in order to improve the degree of purification (Swedish EPA, 2009).

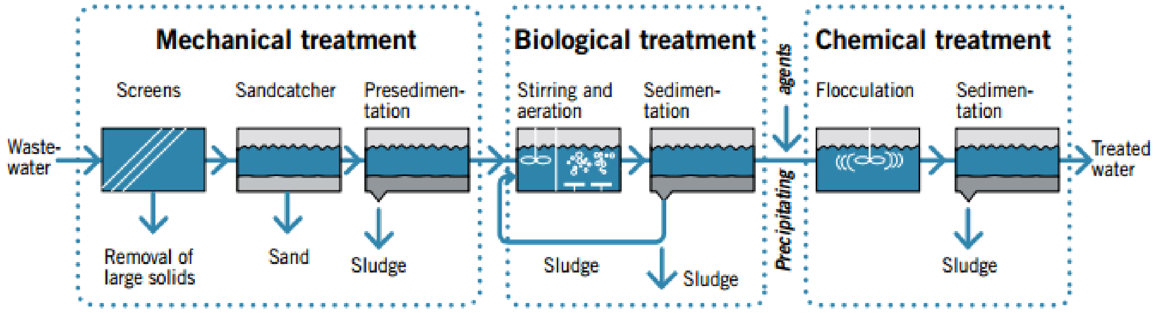

Figure 2. Conventional wastewater treatment methods in Sweden. Source: Swedish EPA (2009) and used with their permission.

The resulting sludge is collected and can be used as fertilizer on arable land. For this purpose, the sewage sludge should only contain low concentrations of hazardous substances such as heavy metals or organic pollutants, as well as pathogens. However, the better the treatment of the wastewater, the lower the quality of the resulting sludge due to the removal of contaminants from the water phase.

Along with the work for improvement in the quality of the sludge, there is scope for recycling nutrients from the wastewater by separating urine and toilet water in the households (Swedish EPA, 2009). Furthermore, **new sanitation systems** are starting to increase their importance for on-site wastewater streams and new wastewater products with promising characteristics. This can be explained by the inefficiency of the current wastewater treatment plants, water scarcity, prices of fertilizers, isolation of the households, delicate status of the water bodies and general environmental concern (Winker *et al.*, 2009).

Several options are available to treat human excreta from toilets, including primary and secondary treatments. All of them have advantages and drawbacks, and will have an influence on the nutrients content and availability for plants. For a safe re-use of human excreta and specially faeces, the elimination of pathogens is mandatory (Jönsson *et al.*, 2004).

The primary treatment of faecal matter is intended to decrease the risk of odours, flies and potential pathogens; desiccation using additives is the most common. A number of methods for secondary treatment have been tested and their efficiency proven to stabilise and sanitise the material, such as storage, composting, incineration, digestion and chemical sanitation (Jönsson *et al.*, 2004; Langergraber and Muellegger, 2005).

Chemical sanitation with urea is one method for secondary treatment that can be used to treat blackwater and consists of the addition of urea. This urea is degraded by the enzyme urease which is naturally found in faeces into ammonium and consequently the pH rises, to approximately pH 9 with addition of 1 % wet weight urea. As a result of ammonia in the form of NH_3, the pathogens are inactivated and the sanitation rate increases (Winker *et al.*, 2009). The proportion of urea needed for an adequate sanitation depends on the substrate to be treated, the storage time and temperature (Hjelmqvist *et al.*, 2012).

One of the advantages of this method is the possibility of estimating the speed of pathogen inactivation by measuring the pH and the concentration of ammonia. Another advantage is that there is no risk of re-contamination of the material as long as the ammonia is contained; this additional amount of ammonia in the toilet water also enhances the value of the product to be applied in the soil as fertilizer (Jönsson *et al.*, 2004; Winker *et al.*, 2009). Another plus is that this method does not need additional materials or energy to work, in comparison to other methods such as wet composting (Hjelmqvist *et al.*, 2012). A drawback is the fact that when the pH-value increases, the risk of nitrogen-loss when spreading increases, and therefore a suitable spreading technique is called for (Hjelmqvist *et al.*, 2012).

3.4 Legal aspects

The legislation in Sweden embraces the idea of sustainability and nutrient reuse and recycling and consequently there are a number of laws, regulations and guidelines that affect the implementation of source-separated systems (Johansson *et al.*, 2009).

The Environmental Code, dating from 1998, contains several options for the implementation of closed-loop oriented techniques in rural areas of the country. It includes a list of integral objectives, such as recycling, efficient use of natural resources, precautionary principle, polluter pays principle and the concept of 'Best Available Technology' (Kvarnström *et al.*, 2006; Johansson *et al.*, 2009). **The Wastewater Act** enhances the municipalities to supply their inhabitants with fresh water and build sewage systems; the **Planning and Building Act**, revised in 2005, gives the municipalities the capability to single-handedly decide on the planning and development of space and infrastructure at the local scale, and it could be used as a tool for strategic sanitation planning. After the revision, it has been proposed that enough space for source diversion of waste should be provided in all new built houses. Nevertheless, these regulations are hardly ever used to its full potential in Sweden, at least not in rural areas, regarding closed-loop approaches for wastewater systems (Johansson *et al.*, 2009).

In parallel to the Environmental Code, the **National Environmental Quality Objectives** were established in 1999. These are specific environmental objectives for achieving a sustainable development and, although they are not legally binding, they can be used to reinforce the existing regulations for waste management, nutrients recycling and closed-loop systems under progress. The following interim target was set: *"by 2015 at least 60 % of the phosphorus compounds in sewage are to be recycled for use on productive land, of which at least half should be used on arable land"* (Swedish EPA, 2005). It is unlikely that the target will be achieved by 2015, considering that only 25 *%* of the phosphorus was returned from sewage systems to agricultural land and 14 *%* to other productive land by 2010. The source-separating systems would contribute also to the achievement of the environmental objectives of "Zero eutrophication" and "Groundwater quality" (Hjelmqvist *et al.*, 2012).

The Swedish EPA has developed guidelines and a Handbook on small sewage systems (up to 25 people) for domestic wastewater. Regarding resources conservation and recycling, it is considered that the selected technology should enable the recycling of nutrients from sewage systems or other waste fractions; the requirements shall not be unreasonably costly in relation to the environmental benefit that they might be expected to give (Hjelmqvist *et al.*, 2012). The source-sorting systems are considered as a protective measure that reduces the discharge of nutrients into the environment and, in the case of blackwater systems, they can also improve the protection against infections and spread of diseases. Discharges of wastewater treatment are governed by two rules, which existed since before the Environmental Code came into force. The Regulations (SNFS 1994:7) concerning urban wastewater treatment regulated the collection, treatment, emission and control of wastewater to sewage treatment for more than 2000 people. The regulation contains the limit or target values for emissions of BOD 7, COD, and in some cases total nitrogen. The Regulations (SNFS 1990:14) on the control of emissions to water and water-recipients from plants for the treatment of urban wastewater is regulated by a closer inspection and sampling for plants that work for more than 200 people. These two regulations contains provisions to achieve the requirements for environmental and health protection, while for resource conservation and recycling of waste fractions there are no specific requirements (Hjelmqvist *et al.*, 2012).

At the European level, the **EU's Waste Directive** states that "waste policy should aim to reduce the use of resources and favour the practical application of the waste hierarchy" (Swedish EPA, 2012). Using the resources efficiently is basically about increasing social benefits without increasing the environmental impact while utilising the natural goods and services in the most appropriate and efficient way. For doing so, one way is to follow the EU's waste hierarchy, which consists of five steps to be applied as a prioritisation scheme for policy-making and legislation; these are prevention, preparing for reuse, recycling, other recovery (e.g. energy recovery) and disposal (e.g. landfill) (Swedish EPA, 2012).

The **Water Framework Directive** was adopted in the year 2000 by the European Parliament and Council, and incorporated into Swedish legislation in 2004 in order to establish a framework for the protection of groundwater, inland waters, transitional waters and coastal waters. The main goal is to achieve a “good water status” for all waters by 2015 and to promote a “long-term sustainable water management based on a high level of protection of the aquatic environment” (WFD; 2000/60/EC). The core environmental objectives are focused on preventing further deterioration of water resources, protect and enhance the status of water, promote a sustainable use of water, enhance protection and improvement of aquatic environments through concrete actions for the progressive reduction of discharges, and preventing and ensuring the reduction of groundwater pollution (WFD; 2000/60/EC).

3.5 Earlier environmental impact analyses of reuse systems

Numerous studies have been made based on LCA methodology and aiming to investigate the most favorable option when dealing with conventional and alternative wastewater systems.

Bengstsson *et al.* (1997) evaluated the environmental impact of different types of wastewater systems in three different municipalities in Sweden. The study included conventional wastewater systems and alternative technologies such as urine sorting, liquid composting of toilet and organic kitchen waste and sludge treatment (to be used locally or to be transported to mainland). It was concluded that the most favorable systems were the alternative technologies compared to the conventional wastewater systems. The most important factors in the evaluation of the environmental impact were the emissions of nutrients to water and the substitution of chemical fertilizers and consequent reduction of fossil fuels consumption and emissions to air. The substitutability of nutrients in the material compared to fertilizers was also analyzed in the study. The results showed that the large uncertainty in the substitutability values made it difficult to give generally applicable conclusions. The level of substitutability is influenced by factors such as regional climate, soil type, spreading conditions and type of precipitation chemical used. Nitrogen losses were found to be dependent on how the collection, transport, storage and spreading of urine and sludge was done. Similar results were found in several studies that compared a reference system (milled food waste mixed with wastewater and treated in a WWTP) to sludge utilization system and a blackwater system (Tidåker *et al.* 2006); and storage of urine from source-separated system, liquid composting of blackwater and chemical treatment (with urea) of sludge (Tidåker *et al.* 2007a). The technique and substitution of chemical fertilizers, the design of the facilities for collection and storage and the appropriate spreading technique were important factors to be considered in the blackwater systems, concerning energy use and global warming potential.

At the urban scale, Sweden is playing a proactive role in long term sustainable wastewater management, with several examples of implementation of wastewater systems that try to return the nutrient content of wastewater to agriculture. One example is Norra Djurgårdsstaden, one of Stockholm's eco-profiled areas, which has set high targets regarding wastewater management (Stockholms Stad, 2012). Some of the options being investigated are the use of urine separating systems or the use of vacuum-based toilet water systems as an alternative to urine separation. The project "Wastewater in ecocycles in the city", which aims to produce a large-scale pilot project that enhances wastewater systems in urban ecocycles, is being developed. Another example is Hammarby Sjöstad, a new city area of Stockholm that also has high environmental ambitions regarding waste management (Hellström *et al.*, 2008). Some of the environmental goals followed for water and wastewater are the reduction of hazardous substances in wastewater by 50 %, the reduction in water consumption by 50 % compared to average newly constructed areas in Sweden, and 95 % recycling of the phosphorous found in wastewater. Several systems for wastewater and organic waste management have been analyzed, such as conventional system complemented with sludge treatment for phosphorous recovery, different blackwater systems with and without urine diversion and food waste disposers, and local WWTP with nutrient recovery. The blackwater system with urine diversion stands out as an attractive alternative regarding energy consumption, recovery of fairly uncontaminated nutrients and emissions of eutrophicating substances.

In Swedish rural areas several projects related to blackwater systems are being currently developed and implemented, including different treatments that combine urea application and heat supply. The municipalities of Uddevalla, Södertalje, Örebro, Strängnäs and Västervik are only few examples of this initiative.

4. Part I: Sanitization

4.1 Background

The elimination of pathogens contained in human excreta is mandatory for a safe re-use of human excreta due to the potentially high concentration in the faeces. Previous studies concerning ammonia-based sanitization have shown optimistic results for the sanitation of human excreta (Vinnerås, 2007; Nordin *et al.*, 2009; Nordin, 2010; Magri *et al.* 2013).

Ammonia-based sanitization is based on the generation of a sufficiently high concentration of uncharged ammonia (NH_3) for pathogen inactivation. The ammonia content can be increased by increasing the temperature, the pH or the concentration of total ammonia nitrogen (TAN). One possibility is to add a solution of NH_3, while another alternative is to add NH_3 in the form of urea ($CO(NH_2)_2$) that will be enzymatically decomposed into carbonic acid (H_2CO_3) and NH_3 as shown in Equation 1 (Vinnerås, 2013):

$$CO(NH_2)_2 + 2\,H_2O \rightarrow H_2CO_3 + 2\,NH_3 \qquad \text{(Equation 1)}$$

Depending on the buffering capacity of the initial material and the concentration of urea added, the pH can increase up to approximately 9.2 (Vinnerås, 2013). In relation to the temperature, it has been reported by Nordin (2010) that in batches with faecal material with the same urea treatment "the incubation temperature did not result in significantly different pH values."

Microbial inactivation seems to be related to NH_3 concentrations and the inactivation rate has been found to increase with increased temperature (Nordin, 2010). For bacterial inactivation, the same author confirmed that the faeces treated with urea always resulted in faster inactivation rates in comparison to untreated faeces within the same batch, though variation in the inactivation rates were observed between different faecal batches undergoing the same urea treatment. *Enterobacteriaceae* group has been reported to be inactivated after urea addition at percentages above the 0.5 % (Vinnerås, 2007; Nordin, 2009; Magri *et al.*, 2013). Since previous studies have shown that *Enterococcus* persist longer than *Salmonella* spp. in urea treated material they could be used as an indicator of bacterial inactivation after urea treatment and their use would result, for gram-negative rods such as *Salmonella* and *E. coli* O157 (especially at low temperatures), in an overestimation of the risk associated with blackwater reuse (Nordin *et al.*, 2009).

Studies about inactivation mechanisms of entero viruses (poliovirus) have indicated that sensitivity to ammonia treatments is a general property of this group of viruses (Ward, 1978). While enteroviruses (single-stranded RNA viruses) show effective reduction, doubled-stranded viruses are relatively resistant to NH_3 concentrations (Albihn and Vinnerås, 2007). Bacteriophages are normally used as models for pathogenic viruses and their inactivation is generally slower compared to bacteria. It has been reported that bacteriophages are not affected by uncharged ammonia at low concentrations such as with 0.5 % urea added to faeces (wet mass) (Magri *et al.*, 2013). According to Nordin 2010, the inactivation time for *Salmonella* Typhimurium phage 28B (does not occur naturally in the environment) at 24 °C was reduced by 50 % to 82 days (after addition of 1 % urea) and by 75 % to 41 days (after addition of 2 % urea), compared to untreated faeces. Regarding their indicator value, current studies have shown that bacteriophages might be more persistent to temperature and ammonia based inactivation in comparison to animal or human viruses (Emmoth *et al.* 2011;

Bertrand *et al.* 2012; Fidjeland and Magri, manuscript). Therefore they are considered conservative indicators for virus inactivation studies, due to the higher persistence to ammonia sanitization. They are interesting to use as they are considerably easier to cultivate in contrast to human or animal viruses.

4.2 Aim for sanitization part

The aim of this part of the thesis was to evaluate the sanitization of blackwater collected from households and treated with urea 1 % in a farm located in the municipality of Uddevalla by monitoring the physicochemical and microbiological parameters

4.3 Materials and methods

Sampling site

Blackwater was treated in a tank previously used for storing manure (Figure 3). Dimensions correspond roughly to 22 meters in diameter and approximately 3 meters in height. The total volume of the tank was approximately 1000 m^3. The concrete tank was covered by a PVC roofing membrane but was not insulated.

The tank started to be filled with blackwater during the month of June. The amount of 5.4 tons of urea was added for the first time the 09.07.2013; by then the volume of blackwater inside the tank was 220 m^3. After that, there were several additions of blackwater, with the last addition the 13.09.2013. The second and last addition of urea was performed the 11.09.2013 (3.6 tons). The final volume of blackwater collected was 871.5 m^3.

The blackwater was pumped inside the tank from a small pre-tank located nearby. The urea was added from the top by one of the roof openings. No active mixing was performed, except from the indirect mixing due to the addition of the blackwater with pressure from the pump. The samples were taken in the same place where the urea was added.

Figure 3. Manure tank in the farm where the study took place. Photo: Brenda Vidal Estévez

Sampling

During the filling of the tank which took 16 weeks, three samples were collected at mid depth of the blackwater. When the tank was completely full, samples were taken from three different heights according to the location of the temperature sensors (one sample per level): bottom, middle and surface, leaving 0.05 meters from the bottom and the surface.

Three samples were taken each time with a syringe (60 ml) mounted on a pole or stick and then transferred into a 100 ml bottle, to be sent by post to the laboratory in Uppsala including two frozen ice packs to keep the samples cool.

Physico-chemical analysis

Temperature

The temperature was measured with four sensors type OW-TEMP-Bx-xxA temperature probe from EDS company (resolution ± 0. 07 °C), tied with a rope and distributed according to the same depths were the samples were taken; 5 cm above the tank bottom, middle (approximately 1m from the bottom), 5 cm below the blackwater level and outside the tank. The sensors were connected to a laptop installed outside the tank, and the temperature was measured every 15 minutes and could be followed online.

pH

The pH was measured using a radiometer electrode (PHM210 STANDAR pH METER, MeterLab. Copenhagen, Denmark), submerging the sensor in approximately 15 ml of sample after reaching room temperature.

Nitrogen content

The concentrations of total nitrogen (Tot-N) and total ammonium nitrogen (TAN) were measured photometrically by using, respectively, Nitrogen (total) Cell Test (concentration ranges of 10-150 mg/L N) and Ammonium Cell Test (concentration ranges of 4.0-80.0 mg/L NH4-N) from Merck KGaA, Darmstadt, Germany. The dilutions used were, generally, 10 dilution for the total nitrogen and 100 dilution for the ammonium nitrogen (for some samples dilutions up to 10000 were used).

Microbial analysis

One milliliter was taken from every sample and mixed with 9 ml buffered 0.9 % NaCl peptone water with 0.1 % surfactant Tween 80 (pH 7) and consecutively serial diluted to get concentrations suitable for enumeration. The colony-forming unit (cfu) was used to express the number of viable bacterial cells per milliliter.

Thermo tolerant coliform bacteria (TTC).

Dilutions to 10^{-3} were used. The TTC were plated in double layer of violet red bile (VRG) agar (Oxoid, AB, Sweden) by transferring 1ml of the sample into petri dishes and adding 10-15ml of melted VRG agar cooled to 45±1 °C. After solidifying, an additional layer was poured. The plates were incubated up-side down at 44±0.5 °C for 24±3 hours. Typical colonies of the VRG agar counted as positive are seed shaped, about 0.5 mm in diameter or larger, and have a pink precipitation area around them.

Enterococcus spp.

Dilutions to 10^{-4} were used. The amount of 0.1 ml of diluted samples was plated on the microbial medium Slanetzy Bartley Agar (SlaBa). The plates were incubated up-side down at 44 °C for 48 hours. Typical colonies of *Enterococcus faecalis* are maroon red with metal brilliance, but for other species of enterococci, the appearance can be pink to red.

Bacteriophages.

Filtered dilutions to 10^{-3} were used. A mixture of softagar, filtered and diluted sample, and host bacteria was poured onto plates with blood agar base agar (BAB). The host bacteria *Salmonella Typhimurium* WG49 was used to detect f-RNA phages, while the *Escherichia coli* 13706 was used to enumerate somatic coliphages. Both host bacterial solutions were cultured in nutrient broth during approximately 3 hours at 37 °C.

The BAB plates were incubated at 37 °C for 18 hours. Clear zones showing the lysis of the bacterial cells caused by the phages were counted as plaque forming units (pfu), a measure of the individual infectious virus particles.

Data analysis

A general evaluation was conducted for the microbial inactivation, using a linear regression analysis. The middle samples were used as reference values for the regression analysis. For those microorganisms that had not reached the detection limit by the time the last sample was analyzed (*Enterococcus spp*. and coliphages) the inactivation rate was calculated from the measurements after the last addition of urea and blackwater.

4.4 Results

Physico-chemical analysis

The outside air temperature showed diurnal variation, ranging between 16-2 ºC. Inside the tank, the temperature at the bottom was lower (approximately 2 degrees in average) than the temperature in the middle and surface, which held very similar temperature (between 16-18 ºC) until the second half of September. After that date, the temperatures from the middle and surface dropped considerably, being lower than the temperature from the bottom. The temperature at the surface tended to follow the trend of the air temperature outside with variation over the day, but to a lesser extent (Figure 4).

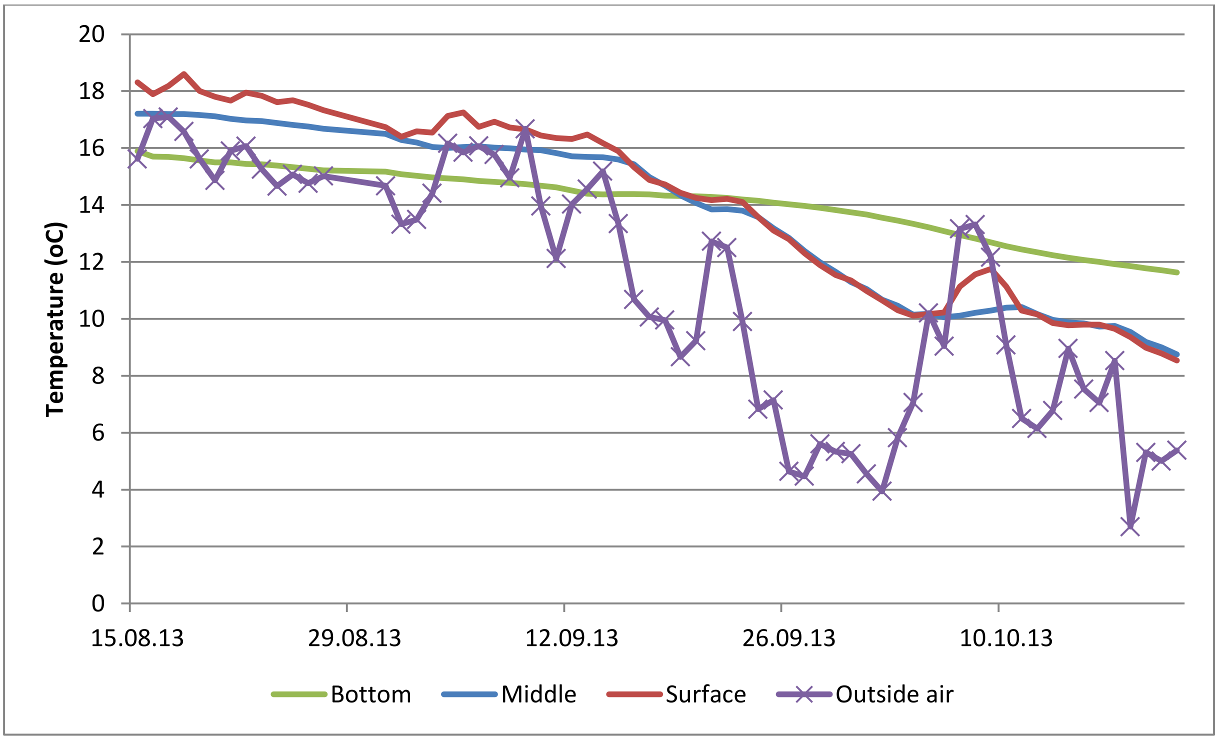

Figure 4. Average temperature measured at 15-min intervals at the bottom (0.05 m above the tank bottom), at the middle of the tank, and at the surface (0.05 m below blackwater level).

The pH in the middle part of the tank was relatively constant throughout the experiment, ranging between 8.6 and 8.8 units. In the bottom the values were considerably higher, up to 9.5 pH units. The values obtained from the middle were similar to those from the surface (Figure 5). After the second sample the pH decreased from 8.8 to 8.6 units. The last urea and blackwater addition was performed the 11th and 13th of September respectively. After the additions, the pH only increased after two measurements (the 30th of September) and remained constant until the last measurement.

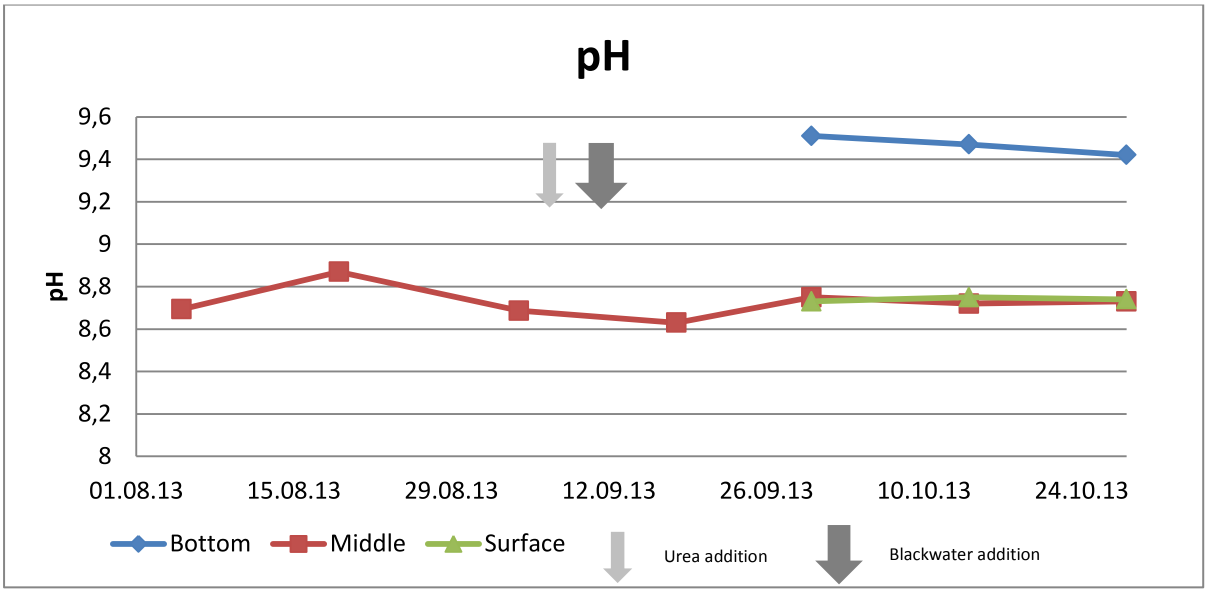

Figure 5. pH values obtained from the different sample points taken every two weeks. The first four points show an average value of the pH measured in the middle of the tank, while for the last three data sets the values were measured at three different levels (bottom, middle, surface).

The total ammonia nitrogen concentration varied between 0.8 and 1.8 grams per liter in the middle samples; similar results were obtained from the surface (Figure 6). On the contrary, the samples from the bottom showed very high concentration in all the cases and it was not possible to obtain reliable values of the exact concentration. In the last three samples from the bottom the ammonia nitrogen first decreased from 18 to 12 grams per liter, and then increased from 12 to 15 grams per liter.

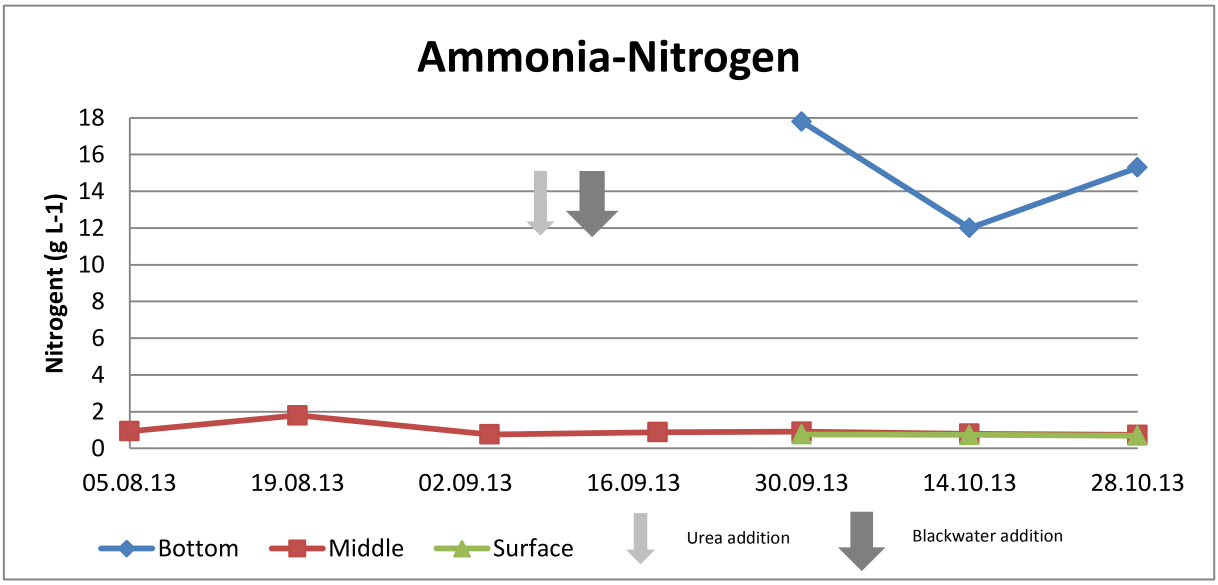

Figure 6. Average values of ammonia nitrogen measured every two weeks. The first four points show an average value obtained from three measurements (triplicate) in the middle of the tank, while for the last three data sets the values were measured at three different levels (bottom, middle, surface).

Regarding the total nitrogen, the concentrations of the first four samples varied within 0.8 and 1.2 grams per liter of blackwater (Figure 7). The values obtained from the bottom were also very high compared to the other two heights. In the samples analyzed the 14th of October the concentration of total nitrogen had increased in the middle and surface approximately one $\log_{10}$ and decreased in a lesser extend in the bottom (100 grams per liter approximately). The last sample showed a decreased of at least one $\log_{10}$ for the three heights.

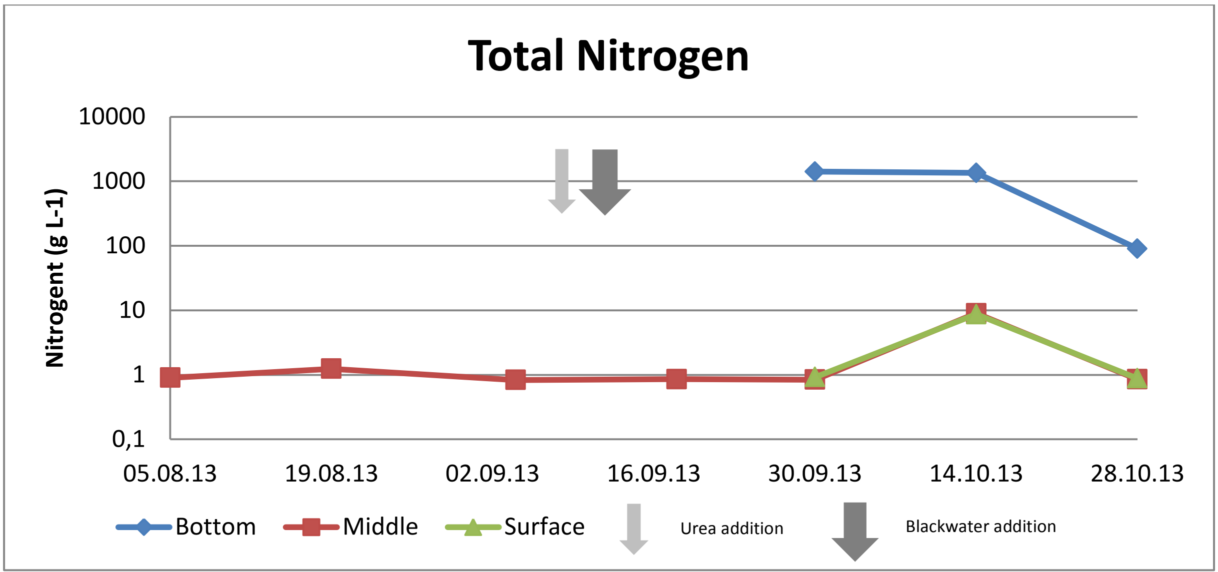

Figure 7. Average values of total nitrogen, measured every two weeks. The first four points show an average value obtained from three measurements (triplicate) in the middle of the tank, while for the last three data sets the values were measured at three different levels (bottom, middle, surface).

Microbial analysis

The thermotolerant coliforms (TTC) were non-detectable in almost all the samples, except for the second sample analyzed (19th of August), when a concentration of 2 $\log_{10}$ (CFU) g^{-1} was obtained (Figure 8a).

Enteroccus spp. was found in higher concentrations and showed to be more resistant to the treatment compared to TTC (Figure 8b). The concentration followed a descendent trend in general, with a sudden drop to non-detectable limits in the third sample (02.09.13). After this day, the concentration went back to earlier values and continued declining in the middle and surface but without reaching the detection limit. Enterococci were never detected in the bottom samples. It must be observed that there was addition of urea the 11.09.13 and blackwater the 13.09.13, which could explain the reappearance of enterococci after the previous inactivation. The regression analysis for the last four samples resulted in no significant decline ($P > 0.05$, $R^2 = 0.7$).

Of the two bacteriophages studied, f-RNA phages were the most sensitive and reached the detection limit in the fifth sample analyzed, following a clear inactivation tendency from 3 $\log_{10}$ to 0 CFU (Figure 8c). The regression analysis was conducted for the first five samples showing a significant reduction ($P < 0.05$, $R^2 = 0.9$). The presence of f-RNA phages was not detected in the last three samples, in none of the three heights. Coliphages were found in 1 $\log_{10}$ higher concentration compared to f-RNA phages (Figure 8d) and were less sensitive to the treatment. The concentration decreased from 4.3 $\log_{10}$ to 2.8 $\log_{10}$ in the middle samples. The regression analysis made for the last four middle samples showed a significant reduction of the coliphages ($P < 0.05$, $R^2 = 0.9$). The concentration of coliphages in the bottom increased from 1 $\log_{10}$ to 3 $\log_{10}$ in the three samples analyzed.

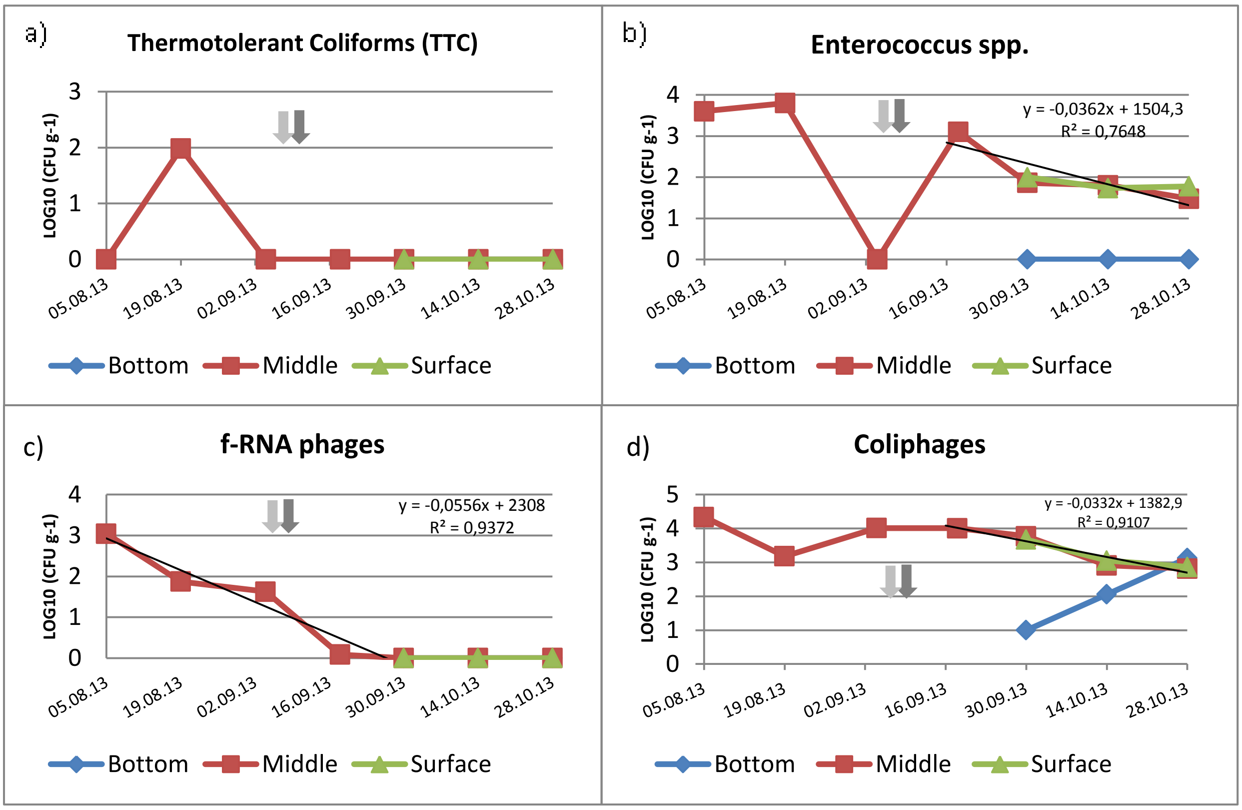

Figure 8. Measured concentration for a) thermotolerant coliforms, b) Enterococcus spp, c) coliphages and d) f-RNA phages, expressed in log10 CFU per gram. Each marker represents the sample analyzed the specific day. The light-grey arrow shows the last addition of urea (11.09.13) and the dark-grey arrow shows the last addition of blackwater (13.09.13).

4.5 Discussion

This study had as one of the objectives to investigate if mixing of the material after urea addition is necessary and is part of a project that will be run over a longer time replicating the treatment. Thus the present data alone cannot be used for an exhaustive quantitative analysis. The reason for this is that the amount of data collected was not big enough, with only 7 sets of samples and none of the last four samples were triplicate, and the conditions inside the tank during at least the first three samples were not the same due to the several additions of urea and blackwater.

The results from the microbiological analysis showed the relation between the pH, the concentration of nitrogen in the bottom and the concentration of indicator microorganisms at the same height; the higher the concentration of nitrogen in the bottom, the higher the pH and the lower the concentration of the organisms studied. In all cases the microorganisms decreased in time in all heights, except for the coliphages measured in the bottom (see Figure 8).

An increment in temperature increases the sanitization of the material both, by increasing the NH_3 formation and by enhancing the effect of the NH_3 present (Vinnerås *et al.*, 2011). Previous research about urea treatment of faeces (Nordin *et al.*, 2009) showed different treatment time requirements to achieve 6 $\log_{10}$ reduction (minimum pathogen reduction for unrestricted use recommended by WHO (2006a)) in several indicator organisms at different temperatures and urea concentrations. To achieve the target inactivation for enterococci group with a treatment of 1 % urea addition, the time required was 3 weeks at 34°C, 4 months at 24°C and 17 months at 14°C, which confirmed the influence of the temperature on the inactivation. In the present study the average temperature inside the tank was above 15°C only until the second half of September and between 15°C and 12°C until the end of the measurements (Figure 4). This might have had an effect on the inactivation rate of *Enterococcus* spp. and coliphages (TTC and f-RNA phages had achieved the detection limit by September), but to which extent the temperature did affect the inactivation rate was not possible to determine since no other scenarios with different temperatures were studied.

The fast inactivation of the thermotolerant coliforms (Figure 8a) was expected according to the literature; the increment from 0 to 2 LOG_{10} in the second sample could be explained by the heterogeneity of the blackwater or some extra addition of blackwater that was not reported. In the third sample the concentration was below the detection level again, as a result of the treatment. *Enterococcus* spp. also declined gradually in time (Figure 8b), but at a lower rate than the TTC. In the last three samples it decreased slightly from 1.9 LOG_{10} to 1.4 LOG_{10} in the middle level of the tank, and from 2 LOG_{10} to 1.7 LOG_{10} in the top level (Figure 8b). When looking at the second sample (19.08.2013), there is a peak in the concentration of ammonia nitrogen and total nitrogen that could explain why in the next sample (02.09.2013) the TTC and *Enterococcus* spp. dropped from 2 LOG_{10} and 3.8 LOG_{10} to zero in both cases, respectively, showing a delayed reaction to the urea addition. According to previous studies like from Vinnerås (2007), the initial reduction of microorganisms after adding urea can be slow due to a combination of factors such as the time needed for the enzymatic hydrolysis of urea to ammonia and the initial resistance of the microorganisms towards the presence of ammonia. The addition of blackwater the 13.09.13 could explain the reappearance of enterococci after the previous inactivation.

The bacteriophages followed slightly different patterns, but also showed a decreasing trend (Figure 8 c-d). As confirmed in previous research (Nordin, 2010; Fidjeland and Magri, manuscript 2013), the coliphages seemed to be more persistent than the f-RNA phages to the treatment. The regression analysis concluded a significant decrease in both cases, although the detection limit was reached only by the f-RNA phages. The concentration of coliphages increased consecutively 1 log_{10} in the three last measurements from the bottom; TTC bacteria had been already inactivated since the beginning of the study and therefore no growth of coliphages could happen. This increment can be explained by the attachment of the phages to the particles that tended to sediment in the bottom of the tank, as it could be observed that the concentration of coliphages in the upper layers has decreased in nearly the same extend. Bacteriophages are known to be absorbed and concentrate in wastewater biosolids, where they can persist longer than in the more liquid fraction (Sidhu and Toze, 2009).

Salmonella spp. is a common organism used as indicator of faecal contamination. In this study it has not been analyzed because *Salmonella* spp. has similar inactivation rates to thermotolerant coliforms as *Escherichia coli* and it is much more sensitive to the ammonia treatment than for instance *Enterococcus* spp. which has longer survival in relation to ammonia, as reported in the literature (Vinnerås *et al.*, 2003; Vinnerås, 2007; Nordin, 2010; Magri *et al.*, 2013). Therefore *Salmonella* spp. was expected to be present (but in lower concentrations compared to *E. coli* because not all persons are infected) and inactivated at the very beginning and no analysis was conducted.

The nitrogen concentration distribution can be explained by the addition of urea in the tank (Figures 6 and 7). The samples were taken in the same place where the urea had been added previously. The urea that was not diluted in the blackwater during the addition might have been deposited in the bottom of the tank, together with particles of higher density. All the measurements of total nitrogen and ammonia-nitrogen from the bottom had a very high content of nitrogen compared to the middle and surface, which could be an indication of the urea deposition. The values might not be very accurate due to the number of dilutions, but enough to show a concentration pattern. The presence of faecal material could influence the movement of the ammonia in the water, affecting the enzymatic hydrolysis of urea and the charged ammonia (NH_4) could be captured by the organic matter (Vinnerås *et al.*, 2009). The concentration of total nitrogen decreased in the bottom and increased in the middle and surface in the measurement from 14.10.13, indicating a possible diffusion of the nitrogen or movement of the water from the lower to the upper parts of the tank. A possible explanation for the movement of the blackwater could be the influence of the changes in temperature. In the middle of September the temperature swapped between the surface and bottom, being warmer at the surface in the first half of the month and then warmer at the bottom in the second half of the month (see Figure 4). This change could have created a gradient or stratification within the blackwater, causing a movement of the blackwater and the nitrogen content from the lower parts where the temperature was suddenly warmer, to the upper parts where the temperature was colder.

The concentrations of ammonia nitrogen and total nitrogen were relatively constant in the middle level, except for the second sample (see Figure 6 and 7). The higher concentration could be due to some extra addition of blackwater not reported which, after being mixed with the urea already present in the blackwater, could have resulted in higher levels of nitrogen in the middle.

The amount of 1 % urea addition was fixed since the planning phase of the current project and it has been discussed if a lower amount could be used, as long as it fulfills the inactivation requirements, to save energy and resources. Research has been carried out already with treatments that include different temperatures and urea concentrations (Vinnerås, 2007; Ottoson *et al.*, 2008; Nordin, 2010; Fidjeland *et al.*, 2013; Magri *et al.*, 2013). If less urea is added to the blackwater, the inactivation rate might decrease with decreasing temperatures which, in this specific study, is related to the weather conditions due to the lack of insulation in the tank. Therefore in case less urea is used, it is recommended to treat the blackwater during the summertime when temperatures above 15 °C can be ensured, or to install heating devises for the same purpose. Mixing is also recommended in order to achieve an even pathogens inactivation and distribution of the nitrogen within the blackwater, making possible to obtain a final product that is homogeneous and can be applied in the field. For a good performance after application, the nutrients distribution within the material should be even, so the plants can take up the nutrients under similar conditions. The urea application in the tank is suggested to be done from different spots in case there is no option for mixing, as well as the sampling for achieving more representative results.

No samples were taken from the blackwater before the urea was added, and therefore no quantitative comparison can be made between the concentration of the different parameters before and after the treatment.

4.6 Conclusions

All microorganisms declined over time after the last additions of urea and blackwater, except for the coliphages in the bottom samples. The higher pH and higher concentration of ammonia the faster inactivation, except for the coliphages.

The time required to sanitize the blackwater will depend on the microorganism considered. The TTC inactivated 2 $\log_{10}$ in approximately two weeks whereas the same reduction of enterococci required approximately three months.

Regarding the bacteriophages, f-RNA phages were more sensitive to the treatment than coliphages and inactivated approximately two $\log_{10}$ in two weeks after the last addition of urea and blackwater. The coliphages inactivated one $\log_{10}$ in two months. Further research about the indicator value of bacteriophages for virus inactivation in faecal material is needed. Based on the trend observed in this study and previous research, it can be concluded that urea (1%) can be used to sanitize blackwater but the treatment time has to be adjusted to the indicator organisms.

Mixing along with the addition of urea is recommended for achieving an even nitrogen distribution in the blackwater and uniform pathogens inactivation, resulting in a more homogeneous material after the treatment. Sampling could be done differently, in order to obtain more differentiated and representative measurements.

5. Part II: Systems Analysis

5.1 Background

The life cycle assessment (LCA) is one of several environmental techniques that can be used as a part of a more comprehensive decision-making process. It analyses and evaluates potential environmental impacts of products or services throughout their life cycle; *from cradle to grave*. The environmental aspects considered are the use of resources and the environmental consequences of the releases. The LCA methodology comprises four main phases: the goal and scope definition, including the definition of the functional unit, the definition of a system boundary and a descriptive flow chart of the different scenarios; the inventory phase, where all the information is collected from various sources and assumptions are defined; the impact assessment phase and the interpretation phase, where the output data and the environmental consequences are assessed and discussed.

5.2 Aim for System Analysis part

The main objective of this study was to analyze the environmental impact of the urea treatment of blackwater for three different scenarios, and its use in agriculture as fertilizer. The impact categories considered were the use of primary energy, use of electricity and global warming potential.

5.3 Methodology

Background and System descriptions

The town of Uddevalla, with approximately 31300 inhabitants (Statistics Sweden, 2010), is located on the west coast of Sweden, in the Västra Götaland County (Figure 9). The farm object of this study is situated approximately at 25 km north-west of the town and it produces grains (mainly oat) for horse feeding. In the surroundings of the farm there are nearly 1500 households, most of them summer houses, which currently have separated blackwater systems. From these 1500 houses, only 300-400 houses are assumed to be involved in this project where the blackwater is collected. This system analysis included three different scenarios:

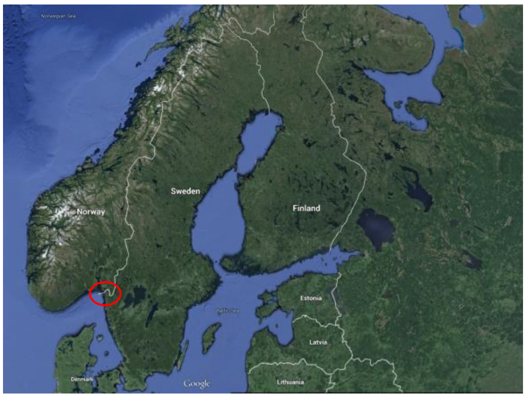

Figure 9. Location of the farm in Uddevalla, Sweden (left) and aerial view of the farm (right).

Scenario 1: Conventional wastewater treatment

It refers to the conventional system, where blackwater generated in the houses is collected and transported to a wastewater treatment plant in order to be treated. No recycling of nutrients is then achieved (Figure 10).

The wastewater treatment plant in Uddevalla is run by the municipality and receives the wastewater from the town itself and the peripheral areas. It provides service to 34 905 persons-equivalent (pe) counted for BOD (70 grams BOD/pe and day) (Miljörapport Skansverket, 2012). The distance from the houses to the treatment plant is assumed to be 25 km.

The wastewater treatment process consists of mechanical, biological and chemical treatment, including active sludge treatment with biological nitrogen removal and chemical phosphorus precipitation. Once treated, the water is discharged in the nearby Bäveån river from where it flows to the sea.

The sludge produced is digested anaerobically at 37 °C in two reactors, with a resident time of 17 to 18 days. The resultant gas is burnt in a boiler and it can also be flared if necessary. The heat produced is used mainly for the heating of the treatment plant and any excess heat is supplied to the local heating network. The digested sludge is dewatered, stored and transported to a landfill 5.5 km away where it is mixed with bottom ash from the heating plant and used for the final covering of the landfill. In the future, the sludge is expected to be used for land improvement and agricultural purposes. The heavy metals concentration is lower than the limit values required by the Sludge Directive 86/278/EEC and by Swedish legislation (European Commission, 2001). It is unknown if the sludge is REVAQ-certified.

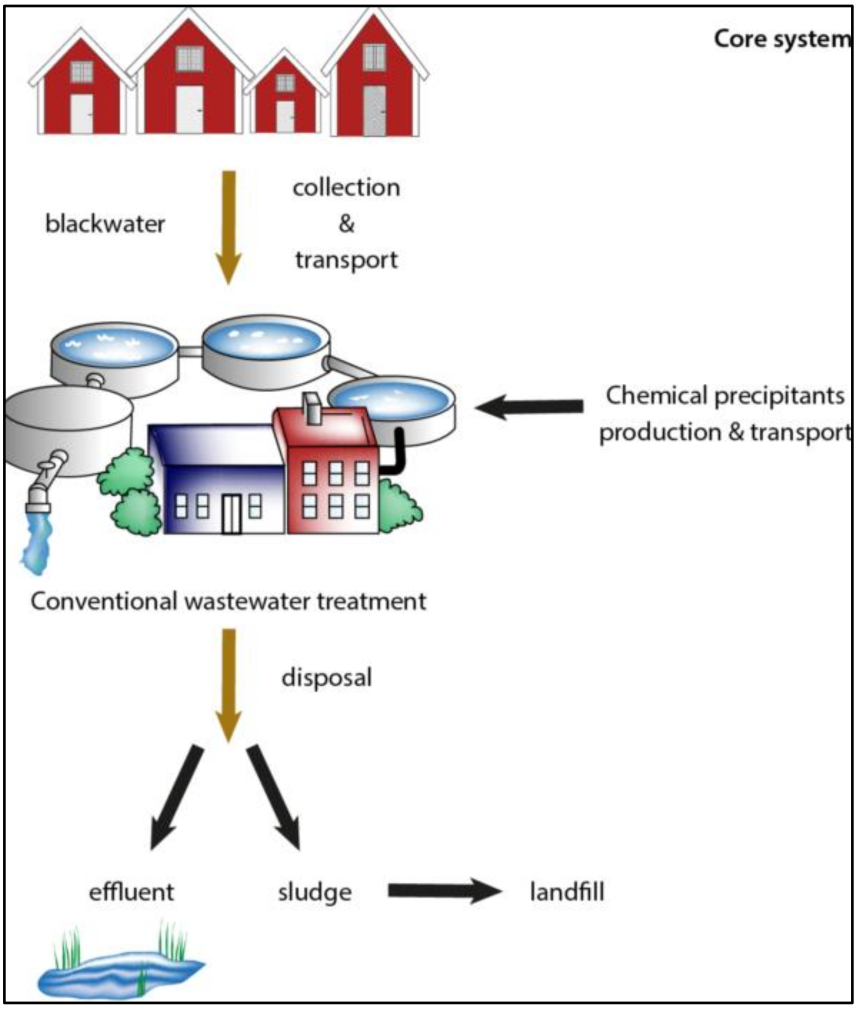

Figure 10. System boundaries scenario 1.

Scenario 2: Blackwater system - urea treatment.

The aim of this system was to recycle the nutrients contained in the toilet water that otherwise would be lost during the conventional treatment in the wastewater treatment plant. The households have already a separate-system with septic tanks of 3 m^3 volume and, therefore, it is assumed that neither installation of new toilets nor building of separated pipe-network is needed. No mixing with other organic fractions is presumed. The blackwater from the toilets including faeces, urine, toilet paper and flush water is collected separately from the greywater. From the 400 septic tanks, 100 are approximately 3 km far from the farm, while the remaining 300 tanks are approximately 8-10 km away from the farm. The average distance of 7 km has been assumed during the calculation process.

The blackwater is collected as often as it might be necessary and brought to the farm. The farm has an old manure concrete tank to store the incoming blackwater (previously emptied and cleaned), with a volume of 1000 m^3. No investments are needed for the construction or maintenance of the tank, with the exception of the roof. Approximately 455 m^2 of PVC roofing membrane were purchased and installed on the top of the tank by May 2013. The tank is not insulated.

The treatment consists in the addition of 1 % of urea right after the blackwater is discharged in the tank. The resulting material is spread in an oat field (Figure 11).

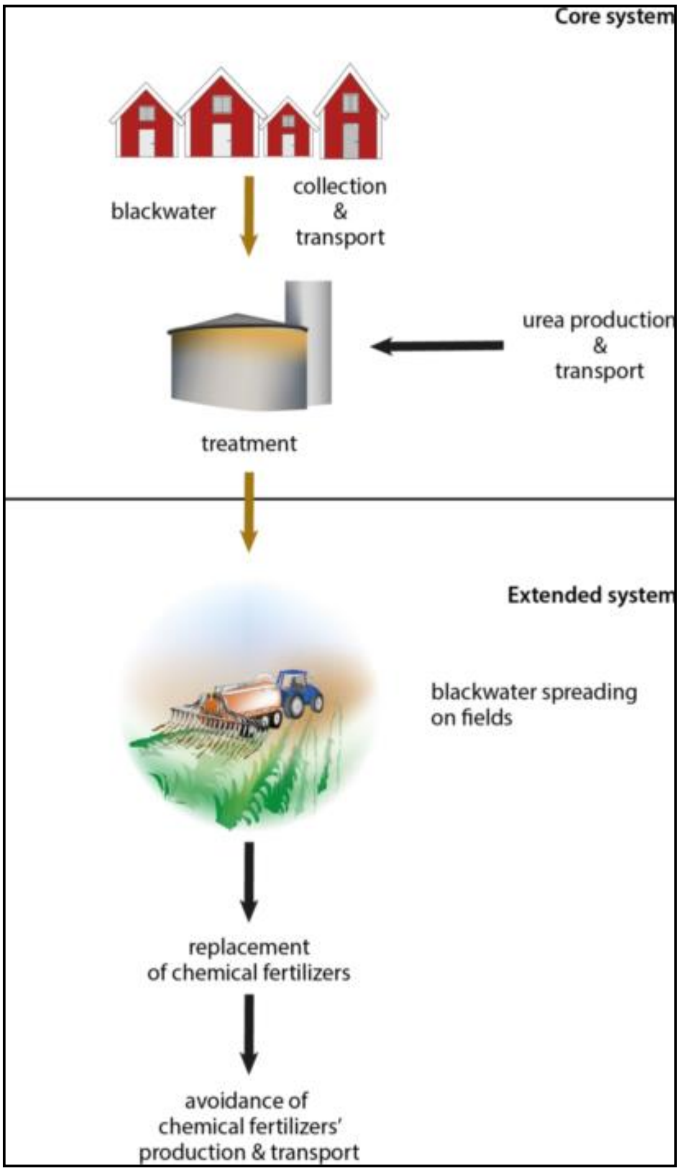

Figure 11. System boundaries scenario 2.

Scenario 3: Blackwater system (urea +heat)

The description of this system (Figure 12) is similar to the previous one. The difference in this case is the dose of urea to be applied; 0.5 % instead of 1 % and the application of heat to increase the temperature by 10 degrees Celsius. The energy required to produce heat was considered to be supplied from three different sources: solar energy, district heating and biofuel (wood pallets or chips).

The assumptions and data concerning households involved, transportation distances, investment in the storage tank and agricultural system were the same than in the previous scenario.

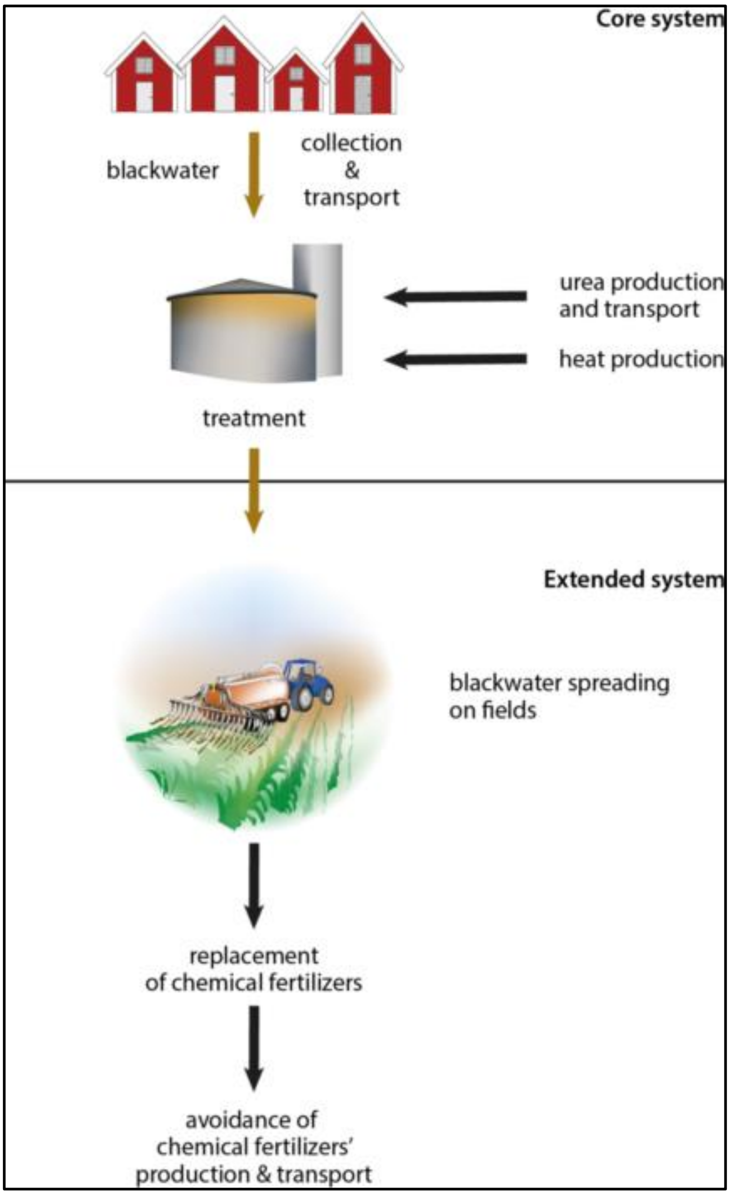

Figure 12. System boundaries scenario 3.

Functional unit

In order to make the results comparable, a functional unit (FU) was required so all the resources consumed and emissions can be related to it. In this case the functional unit was defined as: *cubic meter (m^3) of blackwater to be treated.*

Impact categories

The impact categories and activities used in the inventory were summarized in Table 1. The primary energy use refers to the total requirement of energy for all uses without being subjected to any transformation or conversion process. The use of electricity represents the amount of energy in form of electricity utilized for the production or use of the different activities. The global warming potential attempts to integrate the total impact that a specific action has on the climate, by relating the "*impact of emissions of a gas to that of emission of an equivalent mass of CO_2*" (IPCC, 2077). In this study, the time horizon of 100 years has been adopted, in order to include the integrating radiative forcing in the duration of the perturbation due to the emissions.

Table 1. Impact categories and activities included in the study.

Impact category	Activities
Primary energy use (MJ/FU)	Collection and transport of BW
	Infrastructure (storage)
	Fertilizers production
	Urea production
	WWTP functioning (including landfill and chemical precipitants)
	Field operations
	Heat production
Use of electricity (MJ/FU)	WWTP functioning (including landfill and chemical precipitants)
	Fertilizers production
	Urea production
	Heat production
Global warming potential (kg CO2-eq)	Collection and transport of BW
	WWTP functioning (including landfill and chemical precipitants)
	Infrastructure
	Field operations
	Field emissions
	Fertilizers production
	Urea production
	Heat production

General assumptions

The data corresponding to the **flow of nutrients** was summarized in Table 2, which shows the composition of wastewater fractions expressed as grams per day per household in the form of N-total, N-NH4, N-org, P-total for urine and faeces.

Table 2. Composition of blackwater, including urine and faeces separately in grams per person and day. Source: adapted from Jönsson *et al.* (2005).

	Urine	Faeces	Blackwater
N-total	11	1.5	12.5
$N\text{-}NH_4$	10.3	0.3	10.6
N-org	0.7	1.2	1.9
P-total	0.9	0.5	1.4

From the nutrients fractions, it was considered that 100% of the inorganic nitrogen ($N\text{-}NH_4$) and 10% of the organic nitrogen (N-org) were available for plants. For the phosphorous, 100% of it was considered to be plant available. The Swedish EPA (1985) estimated the reduction of nitrogen and phosphorous in a septic tank to 10-20% and 10% respectively. In this study the reduction of nitrogen in the septic (collection) tank was set to 15% and no reduction was considered for phosphorus. It was assumed that the remaining plant available nutrients replaced the fertilizers. The plant available nitrogen produced per household, 2.6 persons on average, was set to 8.7 kg/year, and for phosphorus 1.3 kg/year.

It was assumed that 77.1 liters of blackwater were generated per day per household presuming normal flushed toilets with a consumption of 4 liters per flush, 7 flushes a day per person. The households were supposed to consist of 2.6 people that spent 100% of the time at home. That is an estimation of 28.2 m^3 a year per household. Accounting for the 350 households, the total theoretical volume of blackwater that would be needed to be treated is 9855 m^3 per year.

During storage of the blackwater, ammonia losses might occur. Based on Karlsson and Rodhe (2002), it was assumed that the ammonia losses during the storage accounted for 1 % of the total nitrogen contained in the blackwater. Data for liquid manure stored in a roofed tank and filling up of the material from the bottom was used.

The **fertilizer products** considered for the LCA were calcium ammonium nitrate (CAN) and triple superphosphate (TSP). Data from these two fertilizers has been used for the calculation of NPK fertilizers.

When calculating the use of fossil fuel and GHG emissions due to transportation, it was assumed that the fertilizers were produced in the production site of YARA in Uusikaupunki, Finland. Based on NTMs methods and data, the amount of fossil fuel needed for transportation has been estimated considering the travelling distances of 1000 km by ship from the production site in Finland to the retailer's (Lantmännens) terminal in the harbor of Helsinborg, 350 km by truck combining the distances in Sweden and Finland.

Primary energy use and GWP was estimated based on the International Bentrup and Pallière (2008) considering the best available technology (BAT), whereas the electricity use was estimated according to Davis and Haglund (1999) due to the lack of available data regarding electricity consumption in the earlier source.

The **infrastructure** included in the LCA consisted solely of the installation of PVC roofing membrane from MPG Company to cover the storage tank with a diameter of 22 m. The use of energy and GHG emissions during the production phase was estimated based on ATHENA™ database and calculations (Franklin Associates, 2001). Transportation of the product was calculated from the production site of the company *Oy Scantarp Ab* in Kuopio, Finland, to Västra Frölunda, Sweden, where the company MPG is located, and from there to the farm in Uddevalla. Calculations regarding transportation were based on NTMs methods and data.

The **transportation** of the blackwater from the houses to the farm was estimated for 7 km, and 25 km from thee houses to the wastewater treatment plant in Uddevalla; calculations were based on NTMs methods and data. The distance from the storage tank in the farm to the field was disregarded.

Concerning **field operations**, according to the Swedish EPA regulations (The Sludge Agreement, "Slamöverenskommelsen"; Bengtsson *et al.*, 1997), the sludge spread per hectare in agricultural land may not contain more than 22 kg of phosphorus and 150 kg of ammonia-nitrogen. Considering the flows of nutrients calculated, the concentrations of phosphorus and ammonia-nitrogen per hectare were below the limits established by the Swedish EPA. Based on data from the Swedish Board of Agriculture, Jordbruksverket (SJV), the recommended fertilizer spreading rate for oat production was assumed to be 80 kg of nitrogen per hectare, for an expected yield of 5 tons of oat per hectare (Albertsson & Blomquist, 2009). In terms of functional unit, in the BW scenario with 1% urea the area fertilized per m^3 of BW would be 0.061 ha and 0.0323 ha for scenario with 0.5% urea, fulfilling the nutrient-requirements by the crop. For the BW scenario with 0.5% urea, the spreading rate was higher because more blackwater would be needed to cover the nitrogen demand of the crop; 30.9 m^3/ha and an extension of 32.3 hectares.

There are plenty of techniques for spreading different types of manure in the fields and hence the amount of energy and consequently the emissions do vary among them. According to the Swedish Institute of Agricultural and Environmental Engineering (JTI, 2002), the amount of fossil fuel needed for spreading artificial fertilizer accounted for a Tractor Valtra 6600 is 17,3 MJ/ha (1.05 MJ per m^3 of BW), while for urine or liquid manure the value would be 69,3 MJ/ha (4.2 and 2.2 MJ per m^3 of BW for 1 and 0.5% urea addition respectively). For this LCA, the blackwater was considered as liquid manure. Emissions due to field operations were calculated according to the same source.

Greenhouse gases **emissions from the fields** after application of fertilizers are dominated by the emissions from nitrogen compounds. The contribution from phosphorous-based compounds is relatively small and, consequently for NPK products, the GHG emission factors are applied only to the nitrogen component. Nitrogen in the soil is subject to several microbial conversion processes in both forms, organic and inorganic. Some of these processes may produce N_2O emissions. Nitrification and denitrification are the two main nitrogen conversion routes followed by the soil microbiology that emit N_2O. Indirect emissions of N_2O occur also due to volatilization of NH_3 (Bentrup and Pallière, 2008).

Some studies (Bouwman *et al.* 2002) show an inverse relation between the share of nitrate in the fertilizer and the N_2O emissions and therefore the N_2O emissions from fertilizers decrease with the increment of the share in nitrate contained in it. It seems reasonable since the use of fertilizers based on urea or ammonium implies (almost always) both types of transformation losses (nitrification and denitrification), while the application of nitrate implies only the risk of denitrification loss (Bouwman *et al.* 2002, Bentrup and Pallière, 2008). The N_2O emissions from applied N in the soil were calculated based on the emission factor 0.8% of N applied for mineral fertilizers, according to Swedish EPA (2003) and 1% for the blackwater as a default IPCC factor (IPCC, 2006). Indirect emissions due to ammonia losses after the material spreading were included. Based on data from the Swedish Board of Agriculture (SJV), the volatilization of ammonia was estimated at 11% of the nitrogen applied, considering trailing hoses technique for spreading (Karlsson & Rodhe, 2002).

For the calculations related to the **use of urea**, it was assumed that the urea was produced in the production site of YARA in Brunsbüttel (Järpemo, 2013), Germany; shipped to Helsinborg (325 km by ship) where the retailer Lantmännen is located and from there to the farm in Uddevalla (320 km by truck). In the first scenario the dose of urea applied to the blackwater was 1%, which means 10 kg per m^3 of blackwater (considering the volume of the tank is 1000 m^3); and 0.5% for the second scenario, which means 5 kg per m^3 of blackwater. The production of urea is always connected to an ammonia plant due to the CO_2 consumption. Nevertheless, the captured CO_2 during the production is released when the urea is hydrolyzed shortly after application in the field, being this amount equivalent to the amount fixed during the production process (Jenssen and Kongshaug, 2003). Energy consumption and GHG emissions from urea production were calculated based on the data (best available technology) from Bentrup and Pallière (2008) and the electricity use was estimated according to Davis and Haglund (1999).

The technical data regarding the **wastewater treatment plant** in Uddevalla was obtained from the latest environmental report (Miljörapport 2012 Skansverket) and can be found in Appendix 5 in more detail. For the calculations of the different impact categories, the values from the functioning of the plant itself, together with the use of precipitants and final disposal of the sludge (landfill) were included.

To relate the values obtained from the WWTP to the functional unit of this case study, the biological oxygen demand (BOD) was considered as a good allocation factor. Default values from URWARE model were used (Jönsson *et al.*, 2005) to estimate the amount of BOD_7 resulting from the blackwater system model (Table 3).

Table 3. Biological oxygen demand (BOD) in blackwater, expressed in grams per person per day. Source: adapted from Jönsson *et al.* (2005).

	Urine	Faeces + toilet paper	Blackwater
BOD_7 in g/pe, day	5.0	34.1	39.1

The total BOD_7 produced by 350 households in the blackwater scenario was 12987 kg BOD/year. In terms of functional unit, the value was 1.32 kg of BOD_7 per m^3 of blackwater. For the calculation of the energy consumption, the energy required for biological treatment and for the nitrogen removal was considered. The factor of 6 kWh per kg of nitrogen removed was assumed according to the literature (Swedish EPA, 2009).

The use of primary energy and electricity associated with **landfilling of the sludge** was calculated based on the data from SimaPro 7.3 Educational (2009). The total primary energy use was 0.39 MJ/kg sludge. The proportion of BOD_7 from the wastewater per kg of sludge treated was used to calculate the amount of energy consumed per functional unit, considering the BOD_7 in the blackwater.

The **chemical precipitants** used in the WWTP, 103 ton/year of iron chloride (PIX-111) and 153 ton/year of polyaluminium chloride (PAX-XL60), were assumed to be produced by Kemira Kemi AB located in Helsingborg, 300 km from Uddevalla. The data for the LCA was obtained from the Swedish Environmental Institute (IVL Swedish Environmental Research Institute Ltd, 2003 a and b).

The **emissions from the WWTP** included emissions from the functioning of the plant itself, together with the use of precipitants and landfilling of the sludge. Emissions from flared gas were not considered due to the small volume produced and lack of detailed information.

Nitrous oxide emissions (N_2O) can occur as direct emissions from treatment plants or from indirect emissions from wastewater after discharge of effluent into aquatic environments (IPCC, 2006).

According to Westling (2011), the suggested value for emitted N_2O from wastewater treatment plants with biological nitrogen removal would be 1.6% of reduced N in WWTP. In the WWTP, 136 tons of N was reduced per year which multiplied by 1.6% results in 2.2 ton/year of N_2O (equal to 648 ton/year CO_2-eq).

Concerning the emissions of nitrogen from the effluent, the following equation proposed by the IPCC (2006) was used:

$$N_2O \text{ Emissions} = N_{EFLUENT} * EF_{EFLUENT} * 44/28 \qquad \text{(Equation 2)}$$

Where:

N_2O emissions = N_2O emissions in inventory year, kg N_2O/yr.
$N_{EFLUENT}$ = nitrogen in the effluent discharged to aquatic environments, kg N_2O/yr.
$EF_{EFLUENT}$ = emission factor for N_2O emissions from discharged to wastewater, kg N_2O-N/kg N.
The factor 44/28 is the conversion of kg N_2O-N into kg N_2O.

In the WWTP from Uddevalla, approximately 63 tons of N was emitted in the effluent last year which following the equation resulted in 0.5 ton/year of N_2O (equal to 149 ton/year CO_2-eq). The emission factor ($EF_{EFLUENT}$) used was 0.005 as default value for domestic wastewater nitrogen effluent (IPCC, 2006).

The landfill emissions were calculated based on the SimaPro 7.3 Educational software (calculations for 1 kg disposal of refinery sludge to sanitary landfill). The value of 0.65 kg CO_2-eq per kg of sludge was considered.

The production of heat for the third scenario was calculated for three different possible sources of heating energy; solar heating, biofuels (pellets) and conventional district heating. Calculations for the use of primary energy and GHG emissions were conducted according to Miljöfaktaboken (2011) (Table 4). It was considered that one kilocalorie is needed to increase the temperature of one kilogram of water 1 °C, and the density of the blackwater was assumed to be the same as water, 1000 kg/m^3. Therefore, 41.84 MJ of energy were required to increase the temperature of one m^3 of blackwater 10 °C. Heat losses were assumed to be 10% in the three cases. Electricity use related to the solar heating was assumed to be equal to the energy required for the pump (to pump the fluid through the panel) and it was set to 6% of the total energy produced. Solar radiation was not included.

Table 4. Primary energy use and GWP for the three alternatives to produce heat; solar heat, biofuels and district heating. Source: Miljöfaktaboken (2011).

Source of heat	Primary energy use (MJ/MJ heat)	GWP (kg CO_2eq/MJ heat)
Solar heat	0.22	0.01
Biofuels	1.18	0.004
District heating	0.79	0.02

5.4 Results

The **primary energy use** (Figure 13) was 83.6 MJ/FU for the WWTP system, 59.6 MJ for the blackwater system with 1% urea treatment and 65.8 MJ for the blackwater system with 0.5% urea (average for different heat sources). The energy saved due to the production of fertilizers was 172.6 MJ for the BW system (1% urea) and 92.7 MJ for the BW system (0.5% urea); the negative values in Table 5 refer to the saving in energy use. Together with the urea production, they represent the main flows of primary energy use in the analysis. The WWTP system required more primary energy use for the collection and transport of the wastewater than the other two systems, due to the longer distances between the households and the plant in Uddevalla (25 km) than between the households and the farm where the blackwater was treated (7 km).

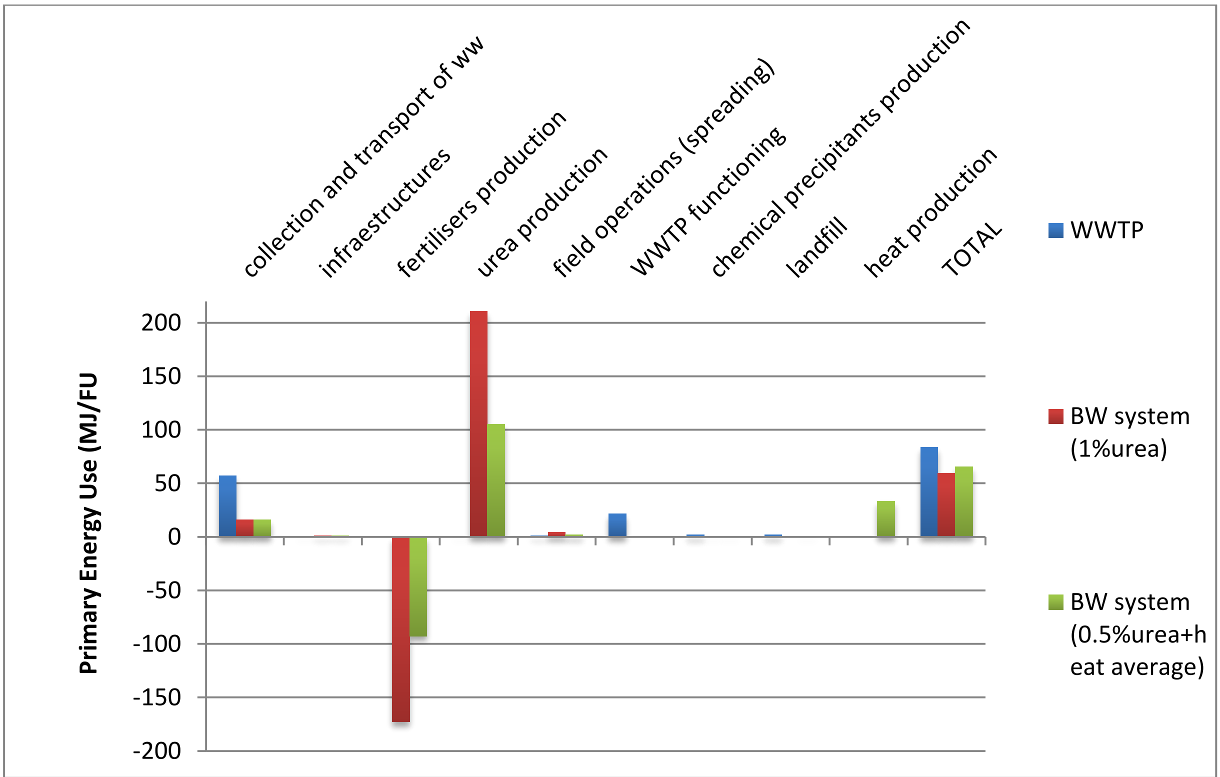

Figure 13. Use of primary energy in the three systems, in MJ per functional unit. The third system is presented as an average of the three possible alternatives for heat production, therefore is not a real option.

The primary energy used for the infrastructures, field operations, chemical precipitants production, and landfill is outranked by the energy used in the other activities. The field operations for the WWTP system required one quarter as much primary energy use as each of the two BW systems (Table 5), but the production of chemical precipitants, the energy required for operating the plant and the landfill activity count only for the WWTP system.

Table 5. Primary energy use (MJ/FU), calculated results.

Activities	Scenarios					
	WWTP	BW system (1% urea)	BW system (0.5%urea+heat average)	BW system (0.5%urea+heat from solar energy)	BW system (0.5%urea+hea t from biofuel)	BW system (0.5%urea+heat from district heat)
Collection and transport of ww	57	15.96	15.96			
Infraestructures		1.25	1.25			
Fertilizers production		- 172.64	- 92.66			
Urea production		210.85	105.42			
Field operations (spreading)	1.05	4.21	2.24			
WWTP functioning	21.46					
Chemical precipitants production	1.88					
Landfill	2.24					
Heat production			33.57	10.06	54.31	36.36
TOTAL	83.64	59.63	65.79	42.27	86.52	68.57

The three different heat sources required different primary energy amounts (Figure 14), although in the previous Figure 13 the value presented for the third system (BW 0.5% urea + heat) was an average approximation. In this order, solar heat used least primary energy for heat production (10 MJ) followed by district heat (36 MJ), and the biofuels was the source of heat with highest requirement of primary energy (54 MJ).

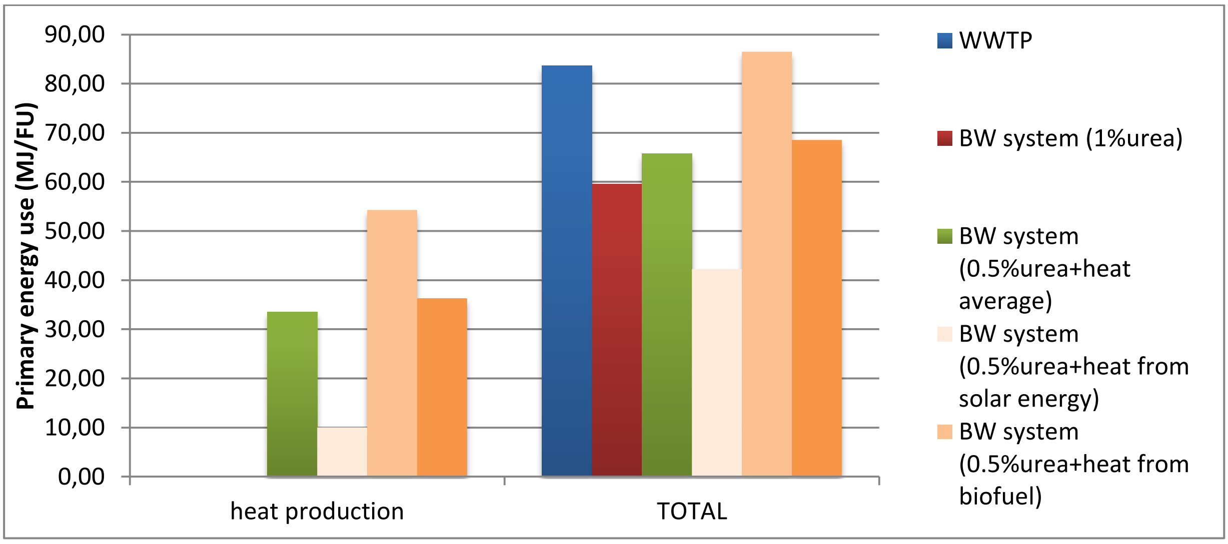

Figure 104. Use of primary energy in the three systems, in MJ per functional unit. The third system (BW 0.5% urea+heat) is shown according to the three possible options considered for heat production: solar heat, biofuels and district heat.

Overall, the results showed that the best option regarding primary energy use per m^3 of blackwater treated is the BW with 0.5% urea and solar heat scenario (42 MJ), followed by the BW with 1% urea scenario (60 MJ). The worst case scenarios were the BW system with 0.5% urea and heat from biofuels (86 MJ) and the WWTP with 84 MJ of energy consumed per functional unit.

The **electricity use** was lowest for the BW system 0.5 % urea with 2.4 MJ (Figure 15), since the electricity consumed for the production of urea was counteracted by the electricity saved during the production of mineral fertilizers. The electricity consumed for heat production was assumed to be the same for the three different alternatives regardless heat source. The BW system 1 % urea obtained similar results, 2.6 MJ per functional unit. The WWTP system used 17.2 MJ, from which 92 % was due to the normal functioning of the plant; specifically for the BOD and nitrogen removal.

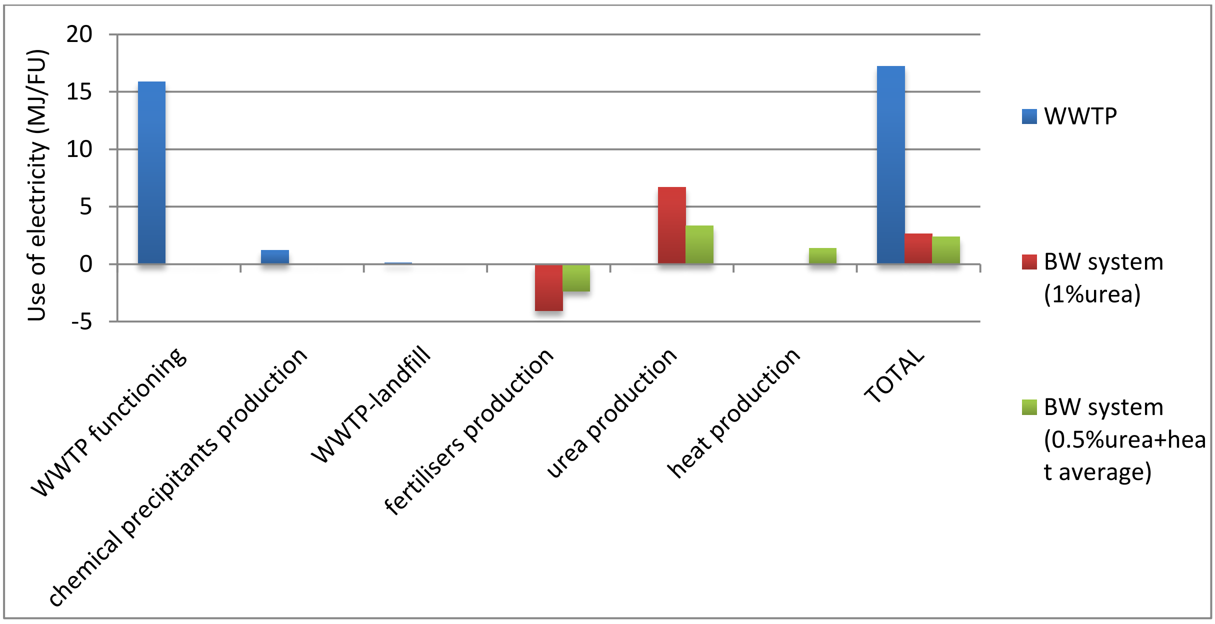

Figure 115. Use of electricity the three systems, in MJ per functional unit.

The **global warming potential** expressed as CO_2-equivalents (Figure 16) was in total 6.05 kg for the WWTP per m^3 of blackwater, -3.7 kg for the BW system (1% urea) and -0.7 kg for the BW system (0.5 % urea, heat average). The negative values indicate that emissions were avoided in the BW systems, because the replacement of fertilizers saved energy and electricity used in the production and transport. The reduced need for chemical fertilizers was therefore a determining factor for the total result. The largest emissions were due to the fertilizers production, followed by the emissions due to urea production. Regarding heat production in the third system, the biofuels had lower emissions (0.2 kg CO_2-eq) than the solar heat (0.6 kg CO_2–eq) and the district heat (1.1 kg CO_2–eq). The global warming potential accounted to the first scenario (WWTP) was mainly due to the collection and transport of the blackwater and the functioning of the treatment plant.

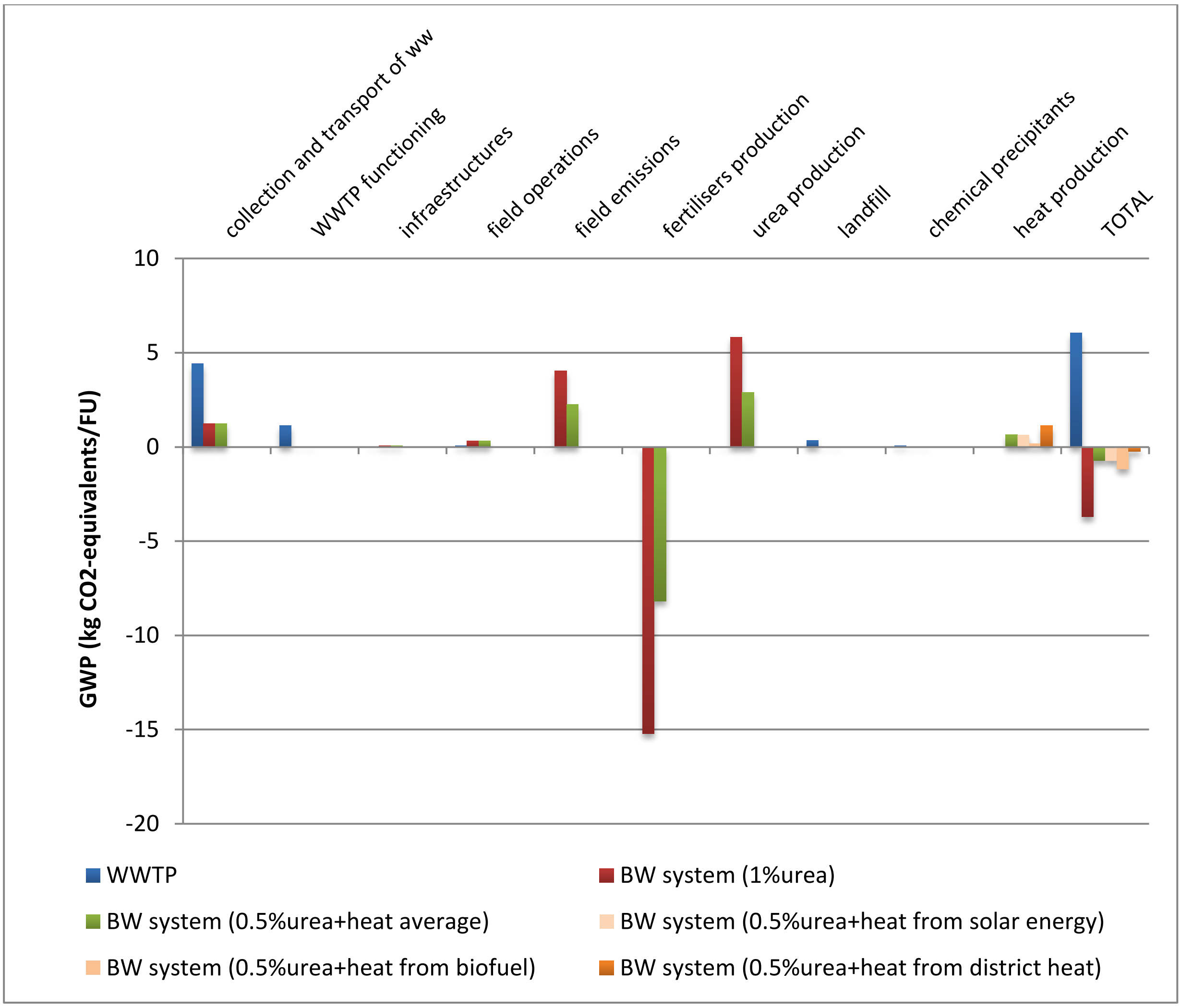

Figure 12. Global warming potential in kg CO_2-eq per functional unit.

Field emissions (Figure 17) were calculated according to the emissions from the field in the blackwater scenarios (positive values) and the emissions from the field if chemical fertilizers would be used instead of blackwater which represents the WWTP scenario (negative values). These two values represented the gross field emissions. The difference between the BW scenarios and the WWTP scenario represented the net field emissions. Emissions during blackwater spreading were assumed to be higher (1 % of applied N) than emissions from chemical fertilizers application (0.8 % of N applied) and that is the reason why the net emissions have positive values.

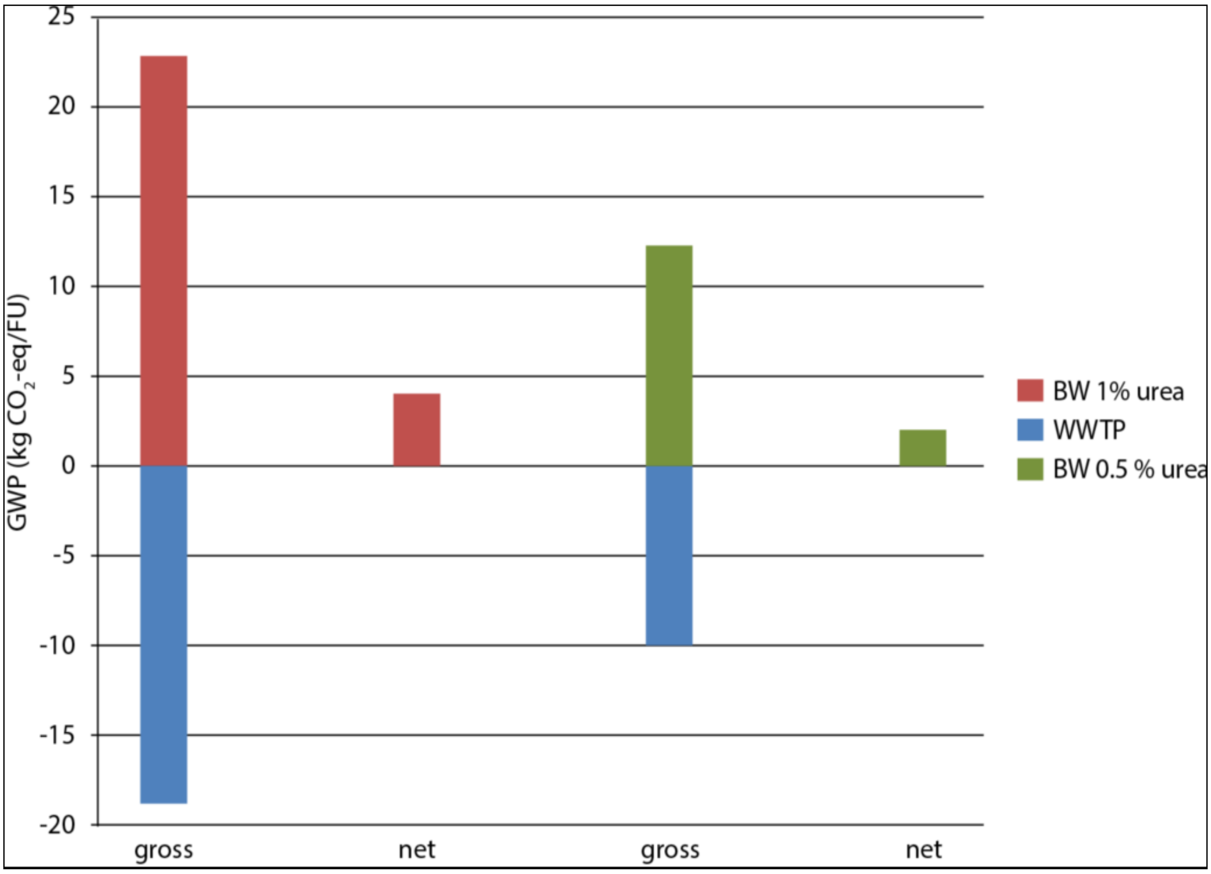

Figure 17. Field emissions expressed as gross emissions and net emissions.

5.5 Sensitivity analysis

The design of a blackwater system can be conducted in different ways and therefore, the assumptions made during the analysis process do influence the final results and their interpretations.

Use of low-flush or vacuum toilets

The calculation of the nutrients flow and blackwater collected from the households was based on conventional toilets that used an average of four litters per flush. If low-flush toilets or vacuum toilets would be used for the analysis, the volume of blackwater collected per household would decrease significantly and consequently the concentration of nutrients would be higher per unit of blackwater. Based on Jenssen *et al.*, (2004) it was assumed that the use of toilets based on vacuum or gravity would use 0.5 litres per flush, instead of 4 litres. These toilets would produce 5.2 litres of blackwater per person per day (17% of the normal one), which means 5 m^3 of blackwater per household per year. The energy consumption was assumed to be 4 kWh/pp/year based on data from Jets™ group (Sanitary Systems).

The primary energy use, the electricity and the GWP were affected when use of low flush toilets was assumed (Table 6).

Table 6. Original total values compared to the values assuming use of low flush toilets (*), for the three impact categories and the three scenarios.

Activities	Scenarios					
	WWTP	WWTP*	BW system (1% urea)	BW system (1% urea)*	BW system (0.5%urea+heat average)	BW system (0.5%urea+heat average)*
Primary energy use (MJ/FU)	83.64	215.48	59.63	11.11	65.79	17.27
Electricity use (MJ/FU)	17.23	106.50	2.65	7.28	2.39	7.02
GWP (kg CO_2–eq/FU)	6.05	13.62	-3.71	-5.99	-0.72	-3.07

The primary energy use and electricity use increased considerably in the first scenario (WWTP) because the blackwater to be treated in the treatment plant would have higher BOD_7 per m^3 of incoming blackwater, and also a higher amount of nitrogen to be removed. For the blackwater scenarios the primary energy use was considerably lower due to the fact that larger amount of fertilizers could be substituted by blackwater since the concentration of N was higher per unit of volume. The GWP was higher for the first scenario (WWTP) due to the emissions from nitrogen removal. The emissions from the field, however, increased approximately 10% in the BW scenarios.

Transportation distance

The results might be also affected by the distance the blackwater has to be transported, whether is to a farm or to a treatment plant. Different distances were calculated for the blackwater system to observe the variations in primary energy use and emissions (Table 7), and compared to the values obtained for the first scenario (WWTP) where a distance of 25 km was assumed.

According to the primary energy use, a distance up to 18 km of transportation would still favor the on-site blackwater alternative to the treatment plant; a larger distance would be detrimental and there would be almost no difference compared to the primary energy use in the WWTP scenario. However, regarding GWP, a distance up to 60 km would still favor the BW system.

Table 7. Comparison between BW 1% scenario considering different transportation distances and the WWTP scenario as a reference.

Distance to the farm (BW scenario)	Primary energy use	GWP (kg CO_2-equivalents)
5 km	53.54	-4.07
10 km	64.94	-3.18
15 km	76.34	-2.30
20 km	87.74	-1.41
60 km	178.9	5.67
WWTP scenario (25 km)	83.64	6.05

5.6 Discussion

Previous research has shown that separation systems offer significant potentials for a sustainable society. Generally, these separation systems have higher rate of nutrients recycling and generally lower emissions to water ecosystems (Bengtsson *et al.*, 1997; Tidåker *et al.*, 2006; Remy, 2010). System analysis based on Swedish scenarios have shown that blackwater systems could be an optimal option if the construction of storage facilities is minimized, the collection and storage facilities are well-designed and optimal spreading technique are used in the field (Tidåker *et al.*, 2006). Furthermore, fertilizers based on human excreta have higher quality compared to sewage sludge or mineral fertilizer in terms of lower concentration of heavy metals (Remy, 2010).

In this study, primary energy use was higher for the first scenario (Table 5), conventional wastewater treatment, due to the higher amount of energy consumed in form of fossil fuels (diesel) for the collection and transportation of the blackwater from the households to the treatment plant. This energy accounted for nearly 70% of the total energy consumption. It was assumed that the blackwater needed to be collected and transported 25 km to the treatment plant, in contrast to the blackwater scenario, where only 7 km to the farm were assumed. The proximity was an important factor and it is one of the strong points of the on-site waste treatments (see Table 7). However, the urea production required the highest amount of primary energy use compared to the other activities analyzed in this study. The production phase of urea requires nearly three times as much energy as the production of other chemical fertilizers (e.g. CAN or TSP) in terms of MJ per kilogram of product; 20 MJ against 7 MJ (Brentrup & Pallière, 2008). The difference is less per kg of nitrogen as urea contains 46% nitrogen, while CAN contains 26.5% nitrogen. Nevertheless, the additional nitrogen input from the urea to the blackwater enhances the value of the material as a fertilizer. It is however extremely important both that the nitrogen in the blackwater is not lost and that the farmer knows how much it is, so that use of other fertilizers is actually decreased.

The sanitization effect of the material is an advantage of the blackwater systems compared to the conventional wastewater treatment plants, where the water discharged into the recipient waters still contains pathogens. The majority of the treatment plants are designed to eliminate mainly organic matter and phosphorus. Only 50 to 60% of the pathogenic bacteria are removed during the primary sedimentation and 90 to 99% in case of activated sludge treatment (Hoogenboezem, 2007).

The solar heat production had lower primary energy use compared to the other two options (Figure 14), biofuels and heating district, and represents a feasible option in the near future in Swedish municipalities like Strängnäs where the concept has been already launched. It does seem suitable to get much energy during the months of spring, but the main constrain would be the economic factor because an investment in solar heat devices would be needed.

The electricity use was highest for the conventional wastewater treatment scenario (Figure 15), due to the own consumption of electricity during the operational phase; approximately eight times higher than the other two systems. Based on the Swedish EPA (2009), it was assumed that approximately 70% of the electricity was used for the BOD removal, while the remaining 30% corresponded to nitrogen removal from the total amount of electricity used; supposing that the electricity used for heating is nearly disregarded due to the internal production of heat. The heat production for the third scenario also required certain amount of electricity, in the case of solar heating this consumption is needed for pumping the fluid through the panel (6% of the total energy produced was assumed) and the same value was adopted for the case of the biofuels and district heat. If electrical energy was to be used in a hypothetical system also to heat the blackwater, the amount of electricity required for the same conditions (heat up 1 m^3 of blackwater in 10 ºC) would be approximately 46 MJ (12 kWh). If this would have been the case, the results of this study would change considerable, the third scenario (BW 0.5% urea + heat) would have the highest primary energy use due to the higher electricity consumption.

The global warming potential was higher for the first scenario (Figure 16), conventional wastewater treatment, and the emissions from the collection and transport contributed most to the total number. The BW scenarios resulted in negative numbers, which is translated into avoided GHG emissions due basically to the replacement of the chemical fertilizers by blackwater. The main contributor to the emissions was the production and transportation of the fertilizers. The nitrogen-based fertilizer CAN had the largest emissions (13.62 kg CO_2-eq per m^3 of blackwater) compared to the phosphorous-based TSP (0.16 kg CO_2-eq per m^3 of blackwater). Comparing the production phase of chemical fertilizers and urea, urea production has only about one third as large CO_2-eq emissions as the fertilizer production, which can be seen as an advantage of the blackwater scenarios. Heat production contributed to a lesser extent to the total amount of emissions; the biofuels contributed least and the district heat the most.

Recent studies have shown that the addition of urea to sludge could decrease the emissions of methane and nitrous oxides due to the effect that urea has on the processes involved during the methanogenesis and nitrogen cycle (Jönsson, H., 2013). In that case the GWP results might be slightly different. Further research is needed to determine to what extend it influences the nutrient cycles.

In this specific case study, no new infrastructures were needed because there were already source-separated systems installed in the houses. But this might not be usually the case; when installation of pipes and tanks is needed, the energy requirements would increase and consequently the emissions of greenhouse gases.

The assumptions for the WWTP scenario were specific for the case study in Uddevalla and included the landfilling of the sludge produced in the plant. This activity contributed with greenhouse gases emissions and energy consumption. The sludge could be used for soil amendment instead; then the energy consumption and emissions would differ from the values obtained in this study. The emissions from the WWTP scenario might have been higher since emissions from flared biogas and from possible methane leakage from the anaerobic digestion were not included.

As an overall summary, the blackwater system with 0.5% urea and solar heating was the best option regarding primary energy; the electricity use was the same for the three sources of heat and the emissions were nearly the same for the solar heating and the biofuels. The next best option would be to use blackwater treated with 1% urea. All the impact categories showed the conventional wastewater treatment plant as the least favorable option.

5.7 Conclusions

Review of literature showed that on-site wastewater separation systems with closed collection tanks and nutrient reuse have several environmental benefits and can be a practical management of the wastewater produced by scattered houses in sensitive areas. Minimizing the volumes of blackwater and making sure that the nitrogen in the blackwater replaces chemical fertilizer are crucial for the environmental benefits of the wastewater reuse systems.

In this study, systems including nutrient reuse from the blackwater were a better option compared to the conventional wastewater treatment for all the studied impact categories. Less urea should be used (0.5% instead of 1%), to decrease the environmental impacts in terms of energy consumption and GHG emissions. Between the three different sources of heat, solar heating was found to be the best option for heat production as an overall.

The conventional wastewater treatment scenario required largest amount of energy and electricity and had higher GWP than the blackwater scenarios, making it the least favorable option. The collection and transportation of the blackwater and the functioning of the plant itself had the main impact on the results.

The use of low-flush toilets (0.5 liters per flush) would reduce the primary energy use and the GWP in the blackwater scenarios compared to the use of normal toilets (4 liters per flush), because less volume of blackwater would be needed to be transported and larger amount of fertilizers could be substituted by blackwater if same volume considered. However, the electricity use would increase. A reduction of the transportation distance was found to be of major importance because it contributes considerably to the primary energy use and greenhouse gas emissions; the lower the distance between the production site and the treatment site, the lower the use of primary energy and GWP. Therefore, for scattered houses in rural areas (like in the present study), treating the blackwater on site is a better option than transporting it to a conventional wastewater treatment plant.

6. General conclusion

This study aimed to link up sustainable sanitation, wastewater management and agriculture, based on the idea that wastewater and blackwater in particular should not be considered as a waste but as a resource. The results from this study support that nutrient recycling from wastewater systems separating blackwater is part of a viable way towards sustainability by solving problems which need urgent solutions such as the lack of quality of freshwater, the spreading of diseases and the environmental deterioration.

Urea treatment of blackwater is one option among many others for sanitizing blackwater, and it has proved to be efficient from a hygienic and environmental point of view. Further research is needed on how to optimize reuse systems working under different conditions and adapted to the local circumstances, like for instance the possibility of using less urea and using solar energy for heat production.

The results from the environmental system analyses indicate that decentralized treatment is a feasible option for blackwater management, especially in rural areas where the households are scattered and often not connected to a municipal sewage network. There are many possibilities regarding the amount of urea to be used and the possibility of applying heat for sanitization purposes. The optimal combination of all the different factors must be studied in every case. The use of low-flush toilets shows great advantages by reducing the volume of wastewater to be treated and increasing the nutrient concentration in the end fertilizer product.

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9. Appendix

Appendix 1. Calculated results.

Table 1. Electricity use in MJ per FU.

Activities	WWTP	BW system (1%urea)	BW system (0.5%urea+heat average)
WWTP functioning	15.89	0	0
chemical precipitants production	1.21	0	0
WWTP-landfill	0.12	0	0
fertilizers production		-4.07	-2.35
urea production	0	6.72	3.36
heat production			1.38
TOTAL	17.23	2.65	2.39

Table 2. Global warming potential, in kg CO_2-equivalents.

Activities	WWTP	BW system (1%urea)	BW system (0.5%urea+heat average)	BW system (0.5%urea+heat from solar energy)	BW system (0.5%urea+heat from biofuel)	BW system (0.5%urea+heat from district heat)
Collection and transport of ww	4.43	1.24	1.24			
WWTP functioning	1.13	0.00	0.00			
Infrastructures	0.00	0.075	0.075			
Field operations	0.08	0.32	0.32			
Field emissions	0	4.05	2.26			
Fertilizers production	0	-15.21	-8.18			
Urea production	0	5.81	2.91			
Landfill	0.35	0	0			
Chemical precipitants	0.08	0	0.00			
Heat production			0.66	0.64	0.20	1.13
TOTAL	6.05	-3.71	-0.72	-0.74	-1.18	-0.25

Appendix 2. Fertilizers substitution.

Table 3. (Davis and Haglund, 1999) and primary energy consumption (Brentrup, F. & Pallière, C., 2008) for the production of fertilizer Calcium ammonium nitrate (CAN) 26,50 % N in MJ.

	For the whole fertilizer (MJ/kg)	For the nutrient (N) (MJ/kg)	Per FU (1% urea)	Per FU (0.5% urea)
Electricity	0.2	0.8	3.7	1.8
Primary energy	8.3	31.4	154.1	81.9

-Calculations of energy required for producing the nutrient were based on the fraction of nitrogen present in the fertilizer (26.50 %).
-Calculations of energy required per functional unit (FU) were based on the amount of nutrient present in one m^3 of blackwater (see *General Assumptions*).

Table 4. Greenhouse gases emissions from BAT production of CAN, in kg CO_2-eq /kg of product at plant gate (Brentrup, F. & Pallière, C., 2008).

GHG	Emissions
CO_2	0.5
N_2O	0.22
other	0.03
TOTAL	0.75

Table 5. Electricity (Davis and Haglund, 1999) and primary energy consumption (Brentrup, F. & Pallière, C., 2008) for the production of fertilizer Triple superphosphate TSP (48 % P_2O_5) in MJ.

	For the whole fertilizer (MJ/kg)	For the nutrient (P) (MJ/kg)	Per FU (1% urea)
Electricity	1.76	8.5	0.40
Primary energy	6.39	31.0	1.46

-Calculations of energy required for producing the nutrient were based on the fraction of Phosphorous present in the fertilizer (48 % of P_2O_5) and the proportional molecular weight of P in the molecule ($P_{atomic\ mass}$ = 31 u).
-Calculations of energy required per functional unit were based on the amount of nutrient present in one m^3 of blackwater (see *General Assumptions*).

Table 6. Greenhouse gases emissions from production of TSP, in kg CO_2-eq /kg of product at plant gate (Brentrup, F. & Pallière, C., 2008).

GHG	Emissions
CO_2	0.34
N_2O	0
other	0.01
TOTAL	0.35

Table 6. Distance in kilometers and fossil fuel consumption in MJ considered for the transportation of NPK fertilizers, as a block product built from nitrogen and phosphorous based compounds. Production site of YARA (fertilizers producer): Uusikaupunki, Finland.

	Distance (km)	MJ per tkm	MJ per tone	MJ per kg of product
By truck	350	2.3	798	
By ship	1000	0.2	200	
TOTAL			998.0	0.9

Kg of product per m^3 of blackwater (1 % urea): 19 kg/m^3

Kg of product per m^3 of blackwater (0.5 % urea): 10 kg/m^3

Appendix 3. Use of urea.

Table 7. Electricity (Davis and Haglund, 1999) and primary energy consumption (Brentrup, F. & Pallière, C., 2008) for the production of urea 46 % N in MJ.

	MJ per kg of product (MJ/kg)	MJ per FU (urea 1%)	MJ per FU (urea 0.5%)
Electricity	0.7	6.7	3.4
Primary energy	20.3	202.9	101.4

Table 8. Transportation of the urea calculated from the production site of YARA in Brunsbüttel, Germany to the harbor in Helsinborg where the retailer Lantmännen is located and from there to the farm in Uddevalla.

	Distance (km)	MJ per tkm	MJ per tone	MJ per FU (1%urea)	MJ per FU (0.5% urea)
By truck	320	2.28	730		
By ship	325	0.2	65		
TOTAL			795	8	4

Table 9. Greenhouse gases emissions from production of urea, in kg CO_2-eq /kg of product at plant gate (Brentrup, F. & Pallière, C., 2008).

GHG	Emissions
CO_2	0.45
N_2O	0
other	0.07
TOTAL	0.52

Appendix 4. Emissions due to transportation:

Table 10. Emissions from truck, medium truck for the road transport ((MDV) Medium lorry/truck Dieseldriven medium truck<18 ton, Euro 3) and general cargo 10000 Dwt ship for the overseas transport .

Emissions	Medium truck (road)	General cargo ship (sea)	
	kg per tkm CO_2-eq	kg per tkm CO_2-eq	convertion to CO_2-eq (IPCC, 2007)
Carbon dioxide. CO_2 (total)	0.18	0.0147	1
Carbon dioxide. CO_2 (fossil)	0.18	0.0147	1
Nitric oxides. NO_X	0.0014	0.0004	-
Hydrocarbons. HC	0	0	-
Methane. CH_4	0	0	25
Carbon monoxide. CO	0	0	-
Particles. PM	0	0	-
Sulphur dioxide. SO_2	0	0.0002	-

Table 11. Distances in kilometers and GWP for all the products included in the system analysis.

Activity	Means of transport	Distance in km	GWP (kg of CO_2-eq per ton of product)	GWP per FU (kg of CO_2-eq per m of BW)
BW from households to farm	Truck	7	1.24	1.24
BW from households to WWTP	Truck	25	4.43	4.43
Fertilizers transport	Truck	350	61.95	1.15*/0.63**
Fertilizers transport	Ship	1000	14.73	0.27*/0.15**
Urea transport	Truck	320	56.64	0.57*/0.28**
Urea transport	Ship	325	4.79	0.05*/0.02**
Chemical precipitants transport	Truck	240	42.48	0.02
Investments transport	Truck	970	171.69	0.005
Investments transport	Ship	300	4.42	0.00

Note: distances have been estimated according to Google maps, guessed best possible way to transport goods.
*Urea 1 %
**Urea 0.5 %

Appendix 5. Wastewater treatment plant.

Table 12. General information from the wastewater **treatment plant in Uddevalla (Miljörapport 2012 Skansverket).**

Concept	Amount	
People-served water supply	33 966 p	
People served by WWTP	34 905 p	
Inflow of wastewater	7 589 188 m^3/year	
Outflow of wastewater	7 531 629 m^3/year	
Sludge produced	43 512 m^3/year	
Sludge disposal: landfill	4 074 tons sludge/year Dry matter means 28.7 *%* TS content 1169 tons/year	
Chemicals consumption		
Ferrous precipitation chemical	*PIX-111*	103 tons/year
Polyaluminum chloride	*PAX-XL60*	153 tons/year
Polymers	*Superfloc C493 & A110*	9.0 & 0.9 tons/year

Table 13. Mean values in tons per year of the BOD, COD, P-tot and N-tot from both the incoming water and the effluent water (Miljörapport 2012 Skansverket).

	Incoming water	Effluent	Reduction (%)
BOD 7	933	27	97
COD-Cr	3103	276	91
P-tot	31	1,8	94
N-tot	199	63	68

SLU
Institutionen för energi och teknik
Box 7032
750 07 UPPSALA
Tel. 018-67 10 00
pdf.fil: www.slu.se/energiochteknik

SLU
Department of Energy and Technology
P. O. Box 7032
SE-750 07 UPPSALA
SWEDEN
Phone +46 18 671000