VA-teknik Södra

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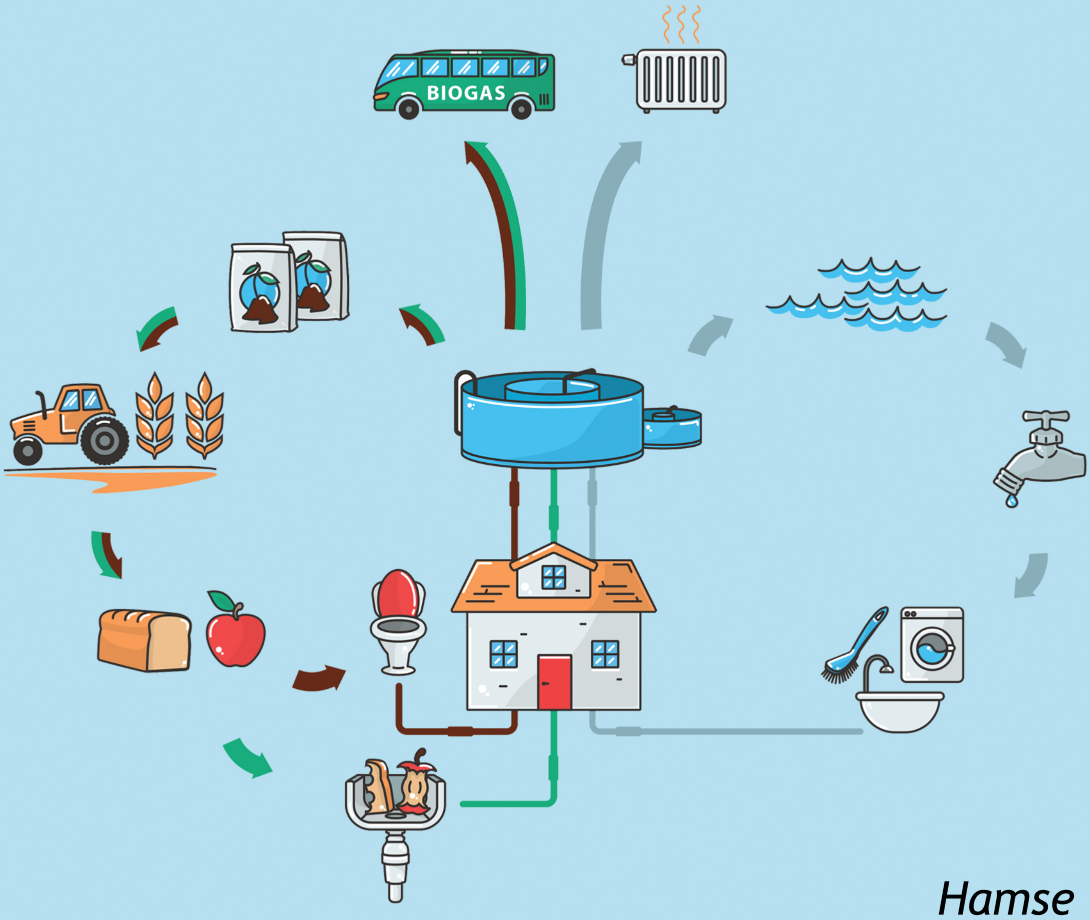
Can source separation increase sustainability of sanitation management?

Impact on nutrient recovery, climate change and eutrophication of two sanitation systems for a hypothetical urban area in Southern Sweden using Life Cycle Assessment



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FOREWORD

The study was performed between the fall of 2015 and the fall of 2016. The background to the study was the ongoing plan to implement source separation systems in the H+ area of the city Helsingborg, Sweden. All work was performed by Hamse Kjerstadius (Lund University, Sweden), Anna Bernstad Saraiva Schott (SAGE, Brazil) and Johanna Spångberg (Swedish Agricultural University, Sweden). The project had a reference group consisting of Håkan Jönsson (Swedish Agricultural University), Marinette Hagman (NSVA), David Gustavsson (Sweden Water Research), José-Ignacio Ramirez (SWECO), Åsa Davidsson (LTH) and Susanne Tumlin (Gryaab). The authors would like to thank the reference group for their work and input to the study. Furthermore the authors would like to thank all companies who supplied data for the study; Jets Sweden, Desah, Ekobalans, NSR and NSVA. The study was partly financed by Åke and Greta Lissheds stiftelse. Front page image was produced by Friends and family and is published with permission of NSVA.

SUMMARY

The Swedish environmental protection agency (SEPA) has been working on more stringent laws in regards to handling of wastewater sludge since the 1990's. The most recent proposal includes goals for nutrient recovery, of phosphorus and nitrogen, from wastewater. Similarly the Swedish Generation goals aims for reduced emissions of greenhouse gases. Swedish conventional management of domestic wastewater and food waste are already contributing to these goals, however source separation systems have been mentioned as potentially more sustainable in regards to climate change and nutrient recovery. Such claims mainly stem from international literature and this study aimed to investigate the sustainability of source separation system relative to conventional management in a Swedish context.

The study used an attributional life cycle assessment (LCA) to investigate two systems for sanitation management in a hypothetical urban area in Southern Sweden. The study included the management chain from household collection, transport, treatment and finally return of nutrients to farmland or disposal of the residual end-product. Results were calculated for nutrient recovery (phosphorus and nitrogen), climate impact and marine and freshwater eutrophication. In order to make the results more generally comparable only domestic wastewater and food waste was considered; thus excluding stormwater, industrial wastewater and other wastes. Of the two systems, the conventional system included collection of food waste by truck, digestion for biogas and return of the entire wet fraction to farmland as biofertilizer. For domestic wastewater, the conventional system consisted of a single wastewater pipe for toilet wastewater (blackwater) and other domestic wastewater (greywater). The combined stream was treated at a central wastewater treatment plant with primary and secondary (activated sludge biological nitrogen removal) treatment with subsequent anaerobic digestion of the produced sludge. The dewatered sludge was used as soil producer (57%) or returned to farmland (43%), thereby replacing mineral fertilizer. For the source separation system three pipes were considered from each household. One for food waste (collected with food waste disposer), one for separated blackwater (collected with vacuum toilet) and one for the remaining greywater fraction. These streams were treated individually at a wastewater treatment plant, with increased biogas production from anaerobic digestion and nutrient recovery by struvite precipitation and ammonia stripping. The final effluent was polished with tertiary precipitation in order to reach discharge standards. The produced nutrient fractions (struvite and ammonium sulphate) were returned to farmland while the produced sludge was returned to farmland (43%) or used for soil production (57%).

The results showed that the source separation system increased nutrient recovery (0.30-0.38 kg P capita^{-1} year^{-1} and 3.10-3.28 kg N capita^{-1} year^{-1}) and decreased climate impact (21-56 kg CO_2-ekv capita^{-1} year^{-1}). Nutrient recovery was increased by the use of struvite and ammonium sulphate. Climate impact was decreased mainly by the increased biogas production, increased nutrient recovery and less emissions of nitrous oxide from wastewater treatment. For marine eutrophication the systems had an equal impact, dominated by discharge of nitrogen via the effluent from the wastewater treatment plants. For freshwater eutrophication the source separation had an increased impact, mainly due to the extensive use of chemicals for ammonia stripper. In conclusions the study showed that source separation systems could potentially be used to increase nutrient recovery from urban areas to farmland while decreasing climate impact.

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1. Introduction

Sanitation systems for food waste and wastewater are covered by the Swedish Environmental law (SFS, 1998). The Swedish environmental protection agency (SEPA) has been working on more stringent laws in regards to handling of wastewater sludge since the 1990's. The most recent proposal (SEPA, 2013) relates to the Swedish Generational goal, which states that next generation should receive a society in which the largest environmental problems have been solved, without causing increased environmental problems outside the Swedish boundaries (SEPA, 2016c). To solve these problems, 16 national environmental quality objectives have been created, out of which several are linked to the management of sanitation systems for food waste and wastewater systems (SEPA, 2016d). The goals for no eutrophication and decreased climate change are of special relevance, due to the current return of nutrients from sanitation systems to farmland and the impact thereof. Today, residues from food waste management can be returned to farmland as certified biofertilizer (SWMA, 2016) and there also exists a certification system for return of sewage sludge (SWWA, 2016). The return of nutrients from sanitation systems will help to levitate climate change due to decreased need of mineral fertilizer (Brentrup and Pallière, 2008; IFA, 2009). In their latest proposal on more stringent laws for handling of wastewater sludge the Swedish Environmental protection agency includes suggested targets for return of nutrients from wastewater to farmland. The suggested targets are return of 40% of the phosphorus and 10% of the nitrogen from wastewater to farmland (SEPA, 2013). However, some municipal water utilities may have trouble to meet these targets due to the focus on removal, rather than recovery, of nutrients from wastewater. In conventional wastewater treatment phosphorus is mainly removed via the sludge phase. Nitrogen is mainly removed via activated sludge biological nitrogen removal while a minor fraction ends up in the sludge phase. Recycling of the nutrient to farmland via the sludge fraction is practiced in Sweden, but on a national average only 25% of the produced sludge is returned (Statistics Sweden, 2016b). Thus, reaching the proposed targets of recycling of nutrients might be difficult with today's conventional system.

It has been suggested that source separation systems could be an alternative to conventional wastewater management (Hillenbrand, 2009; Meinzinger, 2010; Otterpohl et al., 2003). In source separation systems, toilet wastewater (blackwater), household wastewater (greywater) and food waste is separated from other urban waste and wastewater flows. Separated streams could be treated differently at a wastewater treatment plant in order to increase biogas production and nutrient recovery (Kjerstadius et al., 2015). Some pilot areas with source separation systems are already implemented in northern Europe, and several more are currently being planned for implementation (Skambraks et al., unpublished).

Increased biogas production and nutrient recovery with source separation systems could potentially decrease environmental impact of wastewater treatment and aid the work to reach the proposed national environmental goal of phosphorus and nitrogen recovery for Sweden (SEPA, 2013). The impact of such management would also affect the Swedish Environmental quality objectives for climate change and eutrophication. Although initial work on the potential for nutrient recovery from source separation systems exist (Kjerstadius et al., 2015) there is a lack of up to date research linking this potential to the environmental impact for climate change and eutrophication in a Swedish context. This research gap thus needs to be filled in order to answer if source separation system can help to reach the suggested targets and the Swedish national environmental goals. Life Cycle Assessment (LCA), looking at cradle-to-grave impact of sanitation systems, could be utilized to help fill the research gap. LCA provides a methodology to estimate the environmental impact for selected impact categories. Applied to sanitation systems the LCA method would provide results useful for to municipal water utilities and policy makers who plan city infrastructure in a long term perspective.

1.1. Aim & Goal

The project's aim was to identify environmental advantages or disadvantages of the studied sanitations systems; focusing on impact on climate change, eutrophication and potentials for nutrient recovery to farmland. The goal was to obtain conclusions in regards to the potential for nutrient recovery and what parameters are more important in order to decrease environmental impact for the studied impact categories.

1.2. General Method

The aim and goal was reached by applying an attributional life cycle assessment (LCA) covering the main parts of the studied systems (collection, transport, treatment, energy & nutrient recovery and sludge disposal). The results are presented using an environmental impact assessment with selected impact categories relevant for the aim and goal.

1.3. Delimitations

The study did not aim to give a conclusive answer in regards to which of the studied system that has the lowest overall environmental impact. This cannot be done since only some of the parameters with potential environmental impact are included in the present study. One example of a parameter that would have to be included for a more definite answer would arguably be the fate of micro-pollutants and their eco-toxicological effect on humans and the environment.

2. Methods

The study considered a hypothetical development of a green-field urban area in an existing city in southern Sweden. In this development area all infrastructure for food waste and wastewater management is considered to be built from scratch. It is thus an option to construct either the same infrastructure for food waste and wastewater management as in the rest of the city (a conventional system) or to construct a separate and different system for the new development area (a source separation system). Thus, it is assumed that a conventional system would have the benefits being connected to a large scale implementation (for 120 000 capita) while the source separation system would be built only for the development area (12 000 capita). The rationale here is that a change of wastewater systems from the conventional system to source separation system would take place gradually in urban renovation or green-field areas and thus be built in smaller segments at a time. Since wastewater sewage nets have a long life span (>50 years), it is not reasonable to believe that the entire sewage system could be replaced in a larger city at the same time (for example replacing conventional system with source separation system). Thus, a conventional system for food waste and wastewater management would presently always benefit of the larger scale of such systems. To represent this, the study compares two systems of different scales. One system represents what today is conceived as a conventional system in Sweden, while the other is a source separation system. The data for the conventional system is mainly for a city of 120 000 capita while the data for the source separation system is for a separate urban catchment of 12 000 capita with a completely decentralized wastewater treatment plant. In both cases, a densely populated urban area with multi-residential housing (80 apartments in each building) was assumed. The scale of the study is important to keep in mind since all results will be given per capita and annum. In conclusion, the study thus presents the conventional system with some benefits of scale; which is seemingly realistic due to the small scale installations of source separation systems today.

A graphical description of the study is presented in Figure 1. Each system included infrastructure for collection, transport, treatment and nutrient recovery, as well as spreading on farmland or use of sludge as soil improver. As shown in Figure 1, the study only included domestic food waste and wastewater. The decision to exclude stormwater and industrial wastewater was on similar studies (Thibodeau, 2014; Remy 2010) who excluded these fractions due to the local variation of these streams. It is worth noticing that these excluded wastewaters will have an impact on infrastructure of sewers and treatment plants (Remy, 2010) as well as energy and chemical usage. Lastly, any food waste remaining in the mixed waste of the households was not included in the study. Only food waste which was source separated from other household wastes was considered in the study, similar to the study by Thibodeau (2014).

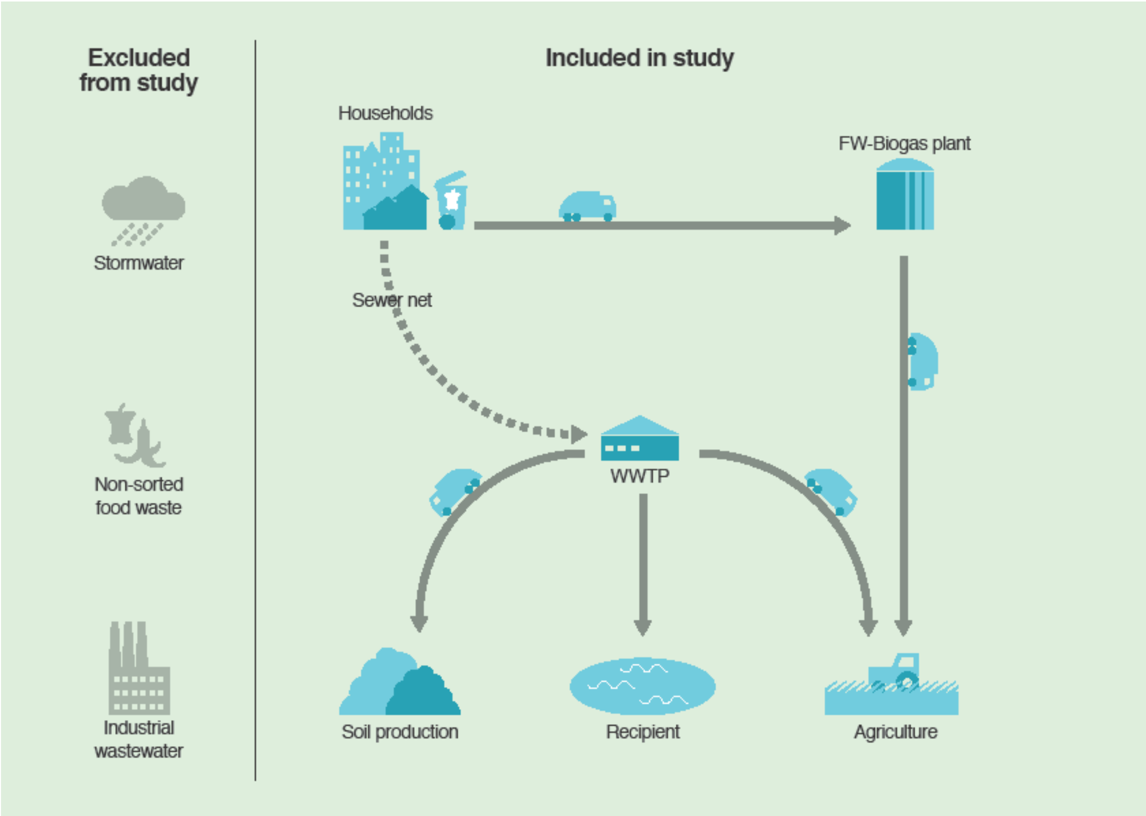


Figure 1. Overall schematic of considered processes and not included waste and wastewaters. Image with permission of NSVA.

2.1. System description – Conventional system

2.1.1. General description – Conventional system

In the conventional system (Figure 2), food waste is sorted at household level, using paper bags. Separated food waste is collected on a bi-weekly basis by garbage trucks and treated at the food waste treatment plant (FW-Biogas plant) were it is being digested to biogas after which the liquid digestate is returned to farmland. Blackwater and greywater is collected in a gravity sewer and treated at the wastewater treatment plant before the treated water is released in to the ocean (recipient). Produced sludge at the wastewater treatment plant is returned to agricultural farmland or used for soil production. Biogas produced at the FW-Biogas plant and the wastewater treatment plant is upgraded to vehicle fuel and used in city buses.

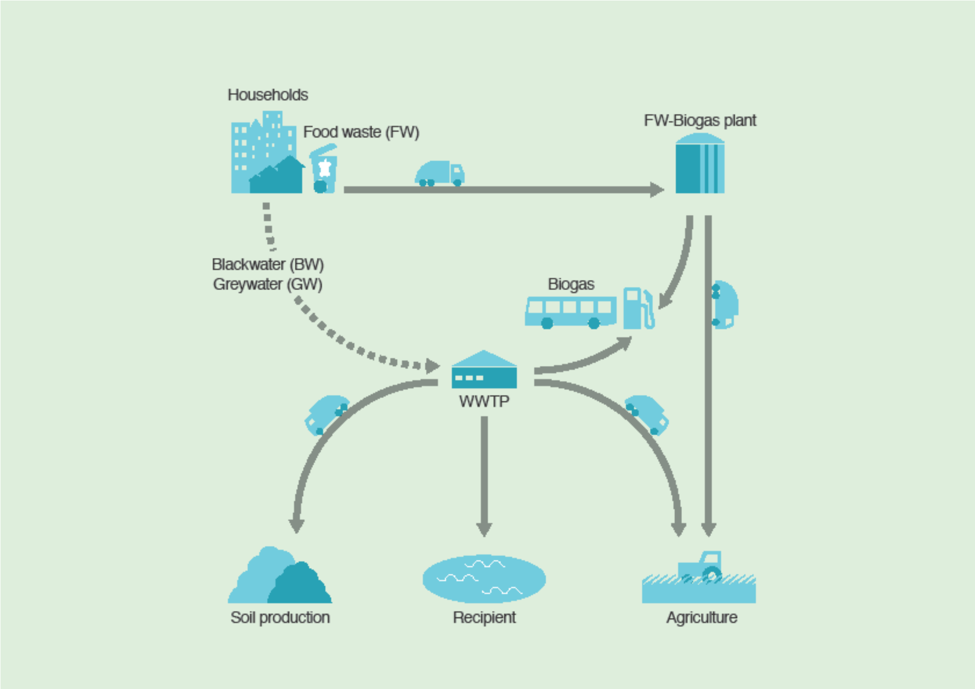


Figure 2. Simplified Schematic of the conventional system. Image with permission of NSVA.

2.1.2. Food waste management – conventional system

The amount of generated food waste was based on literature average data (Jönsson et al., 2005). The Food waste is separated from other waste in the household using paper bags (Figure 3). It is assumed that 50% of the generated food waste was sorted in to the paper bags. The production of paper bags assumed to be done using sulfate pulp, which accounts for the majority of the Swedish pulp production according to Skogsindustrierna (2014), was modelled after adjustment of electricity and heat in order to represent Swedish conditions. Transports of paper bags from production plant to final use was not included in the study. Such transports have proven relevant in the previous studies by Chiew et al. (2015), but as the main reason was inefficient distribution of paper bags, this was not considered relevant for the present study. The amount of paper bags was based on an assumed change of bag every fifth day in each household.

Figure 3. Paper bag with plastic vessel for separate collection of food waste. Published with permission from VA SYD.

Production of plastic for vessels (Figure 3) used for food waste collection in households was based on average European processes, as it is assumed that they were produced outside of Sweden. Transformation of plastic to vessels and transports of vessels from production plant to final use were not included. The amount of vessels used per functional unit was based on an assumed change of vessels in households every fifth year. End-of-life treatment of vessels was not included in the study. Production and end-of-life treatment of waste bins used for separately collected food waste were not considered, as the extra amount of bins needed for separately collected food waste is offset by a decreased need for residual waste. Segregated food waste is collected in a 2-compartment truck. The truck collects food waste and residual waste at the same time, but in different compartments with a total capacity of 6 tons. The fraction of separately collected food waste was estimated to 16% by mass (Bissmont, 2014), and thus 16% of emissions and energy use from collection was allocated to food waste. The transport distance was estimated to 20 km (both ways), and the energy use to 8.2 kWh km^{-1} based on Rehnlund (2010). Although biogas is used in the conventional system in Helsingborg, diesel was the assumed fuel in the study, in order to make results more generic, as use of biogas in collection vehicles is very rare in other parts of the world. The amount of energy used per ton was assumed to be the same as when biogas was used, i.e. 8.2 kWh km^{-1}. A process for collection of municipal solid waste available in ecoinvent v.3.0 was used, using the energy use per km presented above. The process includes provision of fuel, maintenance and use of road. After collection, separately collected food waste is pre-treated by screw-press. The amount of steel used in the screw press was calculated based on data from SWMA (2013) representing the pre-treatment plant at Sysav, Malmö, assumedly sufficient for management of food waste from 120 000 persons. The lifetime was assumed to 25 years. The electricity use in the screw press was calculated as an average from previous assessments of electricity use per ton treated waste in Malmö and Gothenburg (SWMA, 2013). Based on a previous assessment of the mechanical pre-treatment plant in Helsingborg, the amount of dry mass separated as residue was set to 37% by mass.

Residues from pre-treatment are incinerated with energy recovery. Data for infrastructure for incineration plant was collected from Brogaard and Christensen (2013). Auxiliary materials and energy use was based on data from Sysav (2015). Emissions from incineration of residue was modelled using EASETECH, assuming 9.28 kg paper bags per ton incoming food waste (based on the assumed change of paper bag every fifth day, 2.1 persons per apartment and a generation of 190 g food waste capita^{-1} out of which 50% was source separated). Energy recovery from incineration of food waste residue was based on Truedsson (2010), presenting a lower heating value in this fraction of 1 243 kWh ton residue^{-1}. Electricity recovery was assumed to 15% and thermal energy recovery to 85% of the total energy recovery based on Sysav (2015).

Infrastructure needed for the biogas reactor used for food waste was based on data from Remy (2010) (recalculated from a volume of 600 m^3 to 940 m^3). Pumps, plastics and non-alloy steel were assumed to have a lifetime of 15 years while concrete was given a lifetime of 30 years.

Digested food waste is assumed certified as biofertilizer according to the Swedish certification system SPCR 120 (SWMA, 2016). The produced biofertilizer, entire liquid fraction, was transported (20 km) in trucks for six months storage in covered concrete basins. For transport of biofertilizer the lorry (>35 ton, EURO 5) was assumed to have an empty return. Transport and concrete basins were included (for basin material and transport of the material). Following storage the biofertilizer was transported to farmland (30 km) and spread with agriculture spreading equipment for liquid fertilizers with vacuum tanker on farmland. Emissions at storage were ammonia emissions of 1% of total nitrogen, as for liquid manure (Karlsson and Rodhe, 2002), direct nitrous oxide emissions 0.24% of total nitrogen and methane emissions 16.28 gCH_4-C/kgVS (Rodhe et al., 2013). Emissions at spreading, except for the actual spreading operation, were ammonia emissions of 15% of ammonium content of the fertilizer and direct nitrous oxide emissions 0.10% of total nitrogen (Rodhe et al., 2013). As ammonia emissions are depending on pH this was adjusted for according to Chiew et al. (2015) which also was a study on digested food waste. For both storage and spreading indirect nitrous oxide emissions of 1% was included (IPCC, 2006). Avoided spreading of the mineral fertilizers by broadcaster and direct emissions of 1% of total nitrogen (IPCC, 2006) was included. Spread biofertilizer was assumed to replace mineral fertilizers triple super phosphate (TSP) for P and ammonium nitrate (AN) for N. The P of the biofertilizer was assumed to have a plant availability of 100% (Bernstad and la Cour Jansen, 2011) and the N of 70% (Delin et al., 2012). Also carbon storage was included and the average of 3.9% of total carbon added to soil was used based on Linzner and Mostbauer (2005) for digestate. For references and data used see Table A2-6 in Appendix A2.

2.1.3. Sewers – conventional system

Physical installations for collection of household wastewater, such as toilets and sinks, was not considered in the study. Household piping for combined collection of blackwater and greywater was based on Remy (2010) calculated for multiresidential housing. The total length of household piping was 5.8 m capita^{-1}, divided among collection, down pipes and base pipes according to Remy (2010).

The sewer net was considered to be low pressure sewer (LPS) and both infrastructure, 200mm Polypropylene (PP) pipes, and excavation (2m^3 m^{-1} sewer net) was considered. Infrastructure for all pumps in the LPS were based on (Remy, 2010) while electricity demand for LPS-pumps (0.1 kWh m^{-3}) was assumed from general pump curve data and supplied by the water utility of the city of Helsingborg (Dahl, 2015). Number of pumps (0.17 pumps km^{-1} sewer) and sewer lengths (4.9 m capita^{-1}) was based on empirical data from the city of Helsingborg and supplied by the city´s water utility NSVA (Dahl, 2013). Manholes and service stations were not included since they were assumed to be similar in number in both systems and therefore to have similar environmental impact for both systems.

2.1.4. Wastewater treatment plant – conventional system

The conventional wastewater treatment plant was based on Kjerstadius et al. (2015). This included primary sedimentation for organics phosphorus removal with subsequent biological nitrogen removal (BNR) using activated sludge and sedimentation. A sand filter was assumed for polishing before the treated water was discharged in to the ocean. The removed sludge was digested in an anaerobic digester (37 °C, 20 days hydraulic retention time) in a continuous stirred-tank reactor. A layout of the treatment plant is presented in Figure 4 and for a detailed description the reader is referred to Kjerstadius et al. (2015). It is worth mentioning that the selection of treatment processes at the treatment plant are based on the assumption of a large (120 000 capita) treatment plant.

Calculations for the wastewater treatment plant included infrastructure, excavation, operation (electricity, heat, chemicals), atmospheric emissions (CH_4 and N_2O), heat recovery from the effluent and emissions to the recipient ocean water body (direct discharge of nitrogen and phosphorus and gaseous emissions of N_2O from effluent nitrogen). Environmental impact from manpower was disregarded.

Infrastructure was based on a comprehensive review of German treatment plants by (Remy, 2010) stating an average value per treated volume of wastewater. The heat and electricity (49 kWh_{heat} and 57 $kWh_{electricity}$ $capita^{-1}$ $year^{-1}$ respectively) was calculated using a calculation tool presented by Remy (2010) together with the mass balances in the present study (Appendix A1) in order to calculate removal down to the assumed Swedish discharge standard (10mgN/L and 0.5mgP/L). Chemical usage was based on Öresundsverket WWTP in Helsingborg (NSVA, 2014).

Emissions during operation included methane (from anaerobic degradation in sewers and slip during production of biogas from the anaerobic digester) and nitrous oxide (from BNR in the activated sludge system). The emissions of methane, nitrous oxide and chemicals were based upon the carbon foot print tool for wastewater treatment plants presented by (Tumlin et al., 2014) which has previously been published by (Gustavsson & Tumlin, 2013). This included N_2O-emissions as 0.01 kg N_2O-N kg N $denitrified^{-1}$ in biological nutrient removal (Foley et al., 2010). Methane emissions were 0.1 g CH_4 kWh $methane^{-1}$ in the influent wastewater (Göthe, 2013), and emissions from effluent wastewater to the receiving ocean recipient were 0.002 kg N_2O-N kg N^{-1} (Foley et al., 2008).

Chemical usage was based on the Öresundsverket wastewater treatment plant in Helsingborg (NSVA, 2014). However, since stormwater was excluded from the present study some additional post-precipitation was assumed, as described by Lindquist et al. (2003). No chemicals were added as external carbon source for the activated sludge or for pre-precipitation, as described by Lindquist et al. (2003). Thus, chemical need was limited to chemicals for post-precipitation, prohibiting scum-formation in the anaerobic digester and polymers added in sludge centrifugation, details being found in Appendix A2.

Heat recovery from the combined wastewater was assumed to be performed with heat pumps on the wastewater treatment plant effluent. The temperature of the mixed BW and GW stream was calculated to 23 °C and heat losses in sewer net and at the treatment plant were assumed to 4 °C in total. A temperature lift to 50 °C district heating was assumed, resulting in a coefficient of performance (COP) of 3.9, based on (Hellborg Lapajne, 2016). It should be noted that this is much higher than the current COP of the actual heat pump at the wastewater treatment plant in Helsingborg, which has a COP of 2.9-3.2 according to Baaring (2015). This is an effect of the present study not considering stormwater.

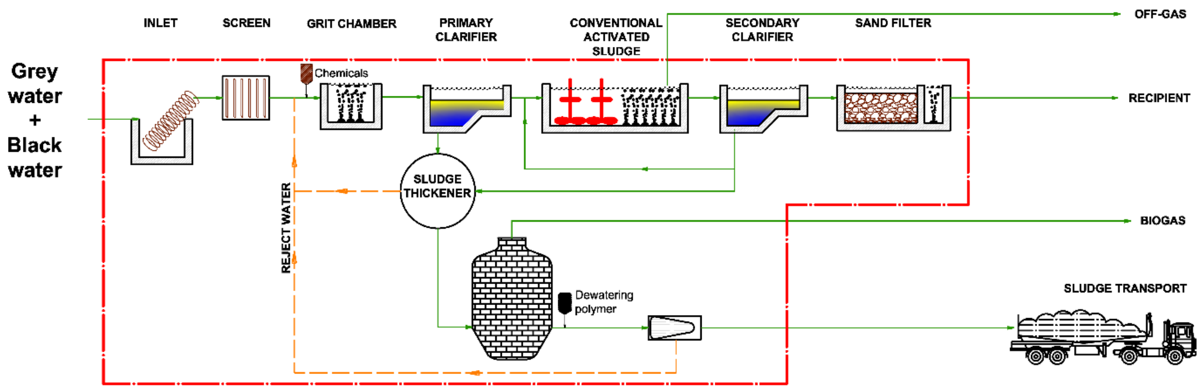


Figure 4 – Lay out of wastewater treatment plant for the conventional system. Image based on Kjerstadius et al. (2015).

2.1.5. Biogas upgrading and usage – conventional system

Produced biogas was considered to be upgraded both at the food waste AD-plant and the conventional wastewater treatment plant respectively. Upgrading was assumed to be done with water scrubbers, as this is a common technology in Sweden (Bauer et al., 2013). Upgraded biogas was spiked with propane in order to match the energy content of the natural gas grid in South Western Sweden (Bauer et al., 2013) and used in city buses, substituting diesel as fuel. Production of buses was assumed to be similar for both systems and thus excluded from the study.

Infrastructure for a biogas water scrubber upgrading facility was originally calculated according to (Starr et al., 2012) but was replaced by an approximate estimation of the gas upgrading unit at Öresundsverket WWTP in Helsingborg, since the estimation according to (Starr et al., 2012) seemed to underestimate the need of stainless steel (the calculated amount of steel is 80 times higher than the amount calculated according to Starr).

Emissions of methane during biogas production was estimated to 0.27 % of the produced biogas Tumlin et al. (2014) and 1% of the upgraded methane during water scrubbing (Bauer et al., 2013).

Upgraded biogas was assumedly used as substitute of diesel in buses. Substitution was based on the ecoinvent process “Transport, regular bus CH”, with the unit “personkm”. A value of 0.024986 kg diesel person-km^{-1} was used, not considering potential differences in energy use for diesel compared to methane. Emissions during usage in city buses include methane and nitrous oxide during combustion in engine (Fruergaard and Astrup, 2011).

It should be noted that, for the sake of simplicity in comparison, all used emission factors from upgrading and usage of biogas were the same as used in the carbon foot print tool for wastewater treatment plants presented by Tumlin et al. (2014) with the exception of biogas slip during digestion for which the 1% slip reported by Bauer et al. (2013) was used.

2.1.6. Sludge & nutrient recovery management – conventional system

Digested and dewatered sludge from wastewater treatment plants is transported 20 km by truck to storage facilities where the sludge is stored on a covered concrete foundation for 6 months. Lifetime of the concrete foundation was assumed to be 50 years and the material and the transportation of the material was included (see Table A2-6 in appendix A2). Emissions at storage were ammonia emissions of 10% of total nitrogen, as for semi-solid manure (Karlsson and Rodhe, 2002), direct nitrous oxide emissions 197.6 mg N_2O per m^3 and hour and methane emissions 123.3 mg CH_4 per m^3 and hour (Flodman, 2002; revised calculations). For sludge storage indirect nitrous oxide emissions of 1% was also included (IPCC, 2006). Following storage, 43% of the sludge was assumed to go to farmland and 57% to production of constructed soil. The 43% sludge return to agriculture is average data for the Scania region in southern Sweden (Statistics Sweden, 2016b) and is far more than the Swedish national average of 25% (Statistics Sweden, 2016b; SEPA 2013).

For sludge going to agriculture (43%) a transportation of 30 km from sludge storage to agriculture was assumed based on local conditions in the city of Helsingborg. For agricultural application calculations included spreading operation (with equipment for spreading solid manure), carbon sequestration and avoided production of replaced mineral fertilizers. Carbon sequestration was calculated in the same manner as for food waste biofertilizer. Of the nitrogen content in sludge used in agriculture 50% was assumed to be plant available replacing mineral nitrogen fertilizer (Delin et al., 2012). Of the phosphorus in sludge, 70% was assumed to be plant available (Hospido et al., 2005; Peter & Rowley, 2009). Emissions at spreading, except for the actual spreading operation, were ammonia emissions of 27% of ammonium content of the fertilizer, as average of solid and liquid manure (Karlsson and Rodhe, 2002) and direct nitrous oxide emissions as 1% of total nitrogen (IPCC, 2006). For data used and references see Table A2-6 in Appendix A2. Cadmium content of the avoided phosphorus fertilizer was assumed to be 5.0 grams per ton phosphorus (Statistics Sweden, 2015).

The sludge fraction going to production of constructed soil (57%) was, after storage, transported by truck 100 km to the constructed soil production site. The storage for this fraction was assumed to be negligible as production of soil improver is continuously taking place year around. At the constructed soil production site the sludge was composted together with other materials. Emissions of nitrogen from composting was assumed to be 30% of the total nitrogen content in the sludge (Vogt et al., 2002); of which 66% where in form of ammonium nitrogen (Boucher et al., 1999) and 2% in the form of nitrous oxide (Kirkeby et al., 2005). Furthermore, methane emissions from composting were assumed to be 0.75% of the total carbon in the sludge (Kirkeby et al., 2005).

The constructed soil was assumed to be used for i.e. golf courses etc., as this was the most common use for this kind of soil in Sweden (Statistics Sweden, 2016b). Spreading was not included as it was assumed that any replaced soil would have been spread in the same manner. As the compost was assumed to be stable and the soil covered after use, no further emissions were included. The constructed soil was assumed to not replace any other soil nor nutrient as the handling of sludge in this manner was assumed as a mean of getting rid of the sludge rather than a product having any actual value.

For both sludge fractions transport with lorry (>35 ton, EURO 5) was included (ecoinvent database 3.0). For transport of sludge the lorry was assumed to have an empty return. Infrastructure for spreading (tractors and other vehicles) was not included as they were assumed to be the same for both systems.

2.2. System description – Source separation system

2.2.1. General description – Source separation system

In the source separation system (Figure 5) there are three pipes leaving each household. Food waste is sorted at household level by the use of food waste disposers (installed in kitchen sinks) and transported in low pressure sewers (LPS) to the wastewater treatment plant. Blackwater is collected separately using vacuum toilets and transported in vacuum sewers to the wastewater treatment plant. Greywater is transported in a separate low pressure sewer (LPS) to the wastewater treatment plant. At the wastewater treatment plant, collected food waste and blackwater is treated directly in an upflow anaerobic digester, UASB-ST digester, as described by Kujawa-Roeleveld et al. (2006). Nutrient recovery is performed on the digestate effluent by struvite precipitation and ammonia stripping (to produce ammonium sulphate). Greywater is treated separately in a high loaded activated sludge unit from which the produced sludge is treated in the biogas digester and the effluent water is treated (using post-precipitation) to discharge limits before discharge in the ocean (recipient). Produced sludge at the wastewater treatment plant is returned to agricultural farmland or used for soil production. Produced struvite and ammonium sulphate is returned to farmland.

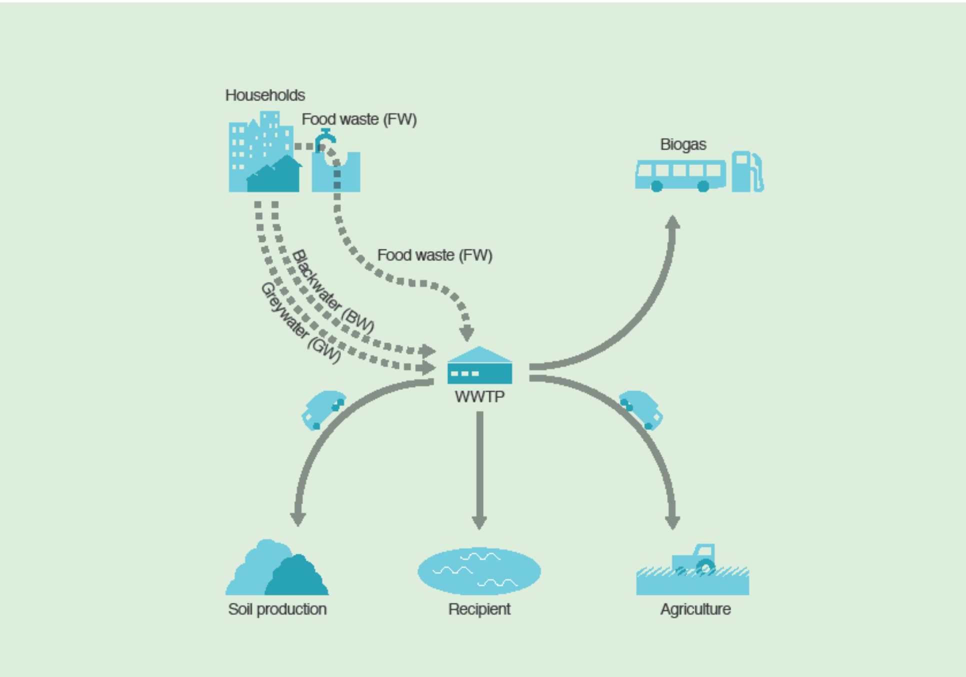


Figure 5. Schematic of the source separation system. Image with permission of NSVA.

2.2.2. Food waste management – Source separation system

Household collection of food waste included a food waste disposer (FWD) and household piping system. Production of food waste disposers was calculated as the amount of material per disposer over 50 years, assuming that each disposer has a lifetime of 15 years, divided by the number of habitants per household. The study includes only provision of the raw materials used in the product, based on data from Annerhall (2010), while further manufacturing, transport and end-of-life treatment of disposers was not included. For the piping connected to the food waste disposer the piping dimensions and lengths were calculated using the model multi-residential houses presented in the dissertation of Remy (2010). This includes smaller piping systems in each apartment as well as down pipes and base pipes to connect to the sewer network outside the property. The total length of household piping used for food waste was 3.2m capita^{-1}, divided among collection, down pipes and base pipes according to Remy (2010).

2.2.3. Sewer net – Source separation system

Physical installations for collection of household wastewater, such as toilets and sinks, was not considered in the study. Household piping for separate collection of blackwater and greywater was based on Remy (2010) calculated for multiresidential housing. The total length of household piping was 7.8m capita^{-1}, divided among collection, down pipes and base pipes according to Remy (2010).

The source separation system has 3 sewers networks (one each for FW, BW and GW). GW and FW sewers each include LPS-pumps while the BW sewer include vacuum pumps. The sewer lengths of each source separated sewer was assumed to 3.7 m capita^{-1}. The assumption was based on the rationale that source separation systems has local treatment plants (for each 12 000 capita) and thus shorter sewer network systems than the conventional system. The assumption that source separation system have local treatment plants follows the general assumption for the study that source separation systems will gradually be implemented in a city. The assumed length was calculated from the Floor Space Index of the H+ area in the city of Helsingborg (FSI of 1.5 compared to 1.0 for the rest of the city) thus becoming (4.9 m capita^{-1}*1.0/1.5 = 3.7 m capita^{-1}). The LPS sewer net for GW was considered to be 200mm polypropylene (PP) pipes while the LPS sewer net for FW and the BW net was considered to be 100mm polyethylene (PE) pipes. Since the triplicate sewer net was assumed to be constructed at the same time, an excavation of 2m^3 m^{-1} sewer net was only considered for the GW sewer net.

The needed number of pumps for LPS-sewers for GW and FW sewers was assumed the same as for mixed household wastewater in the conventional system (0.17 pumps km sewer^{-1}). The BW sewer was assumed to have double the amount of vacuum pumps (100% redundancy) calculated from supplier data of pump capacity (Markstedt, 2015). Electricity demand of vacuum generators (5 500 kWh pump^{-1} year^{-1}) was based on supplier data (Markstedt, 2015). The BW sewer was in addition assumed to have the same amount of LPS-pumps as the GW and FW sewers due to need of pumping along the sewer net. Infrastructure for all pumps were based on Remy (2010) while electricity demand for LPS-pumps (0.1 kWh m^{-3}) was the same as in the conventional system.

Manholes and service stations were not included, since they were assumed to be similar in number in both systems and therefore to have similar environmental impact for both systems.

2.2.4. Wastewater treatment plant (WWTP) – Source separation system

The source separation wastewater treatment plants were assumed to be dimensioned for 12 000 capita each (a factor 10 smaller than the conventional wastewater treatment plant). The assumption that the source separation system has smaller treatment plants follows the general assumption for the study that source separation systems will gradually be implemented in a city. The configuration of the treatment plant (lay out in Figure 6) is based on Kjerstadius et al. (2015), who in turn used a pilot area in the Netherlands as reference (Wiersma and Elzinga, 2014). The treatment includes an activated sludge treatment with BNR for GW (Wiersma, 2013) from which excess sludge is forwarded to an upflow anaerobic sludge blanket septic tank (UASB-ST) digester. Incoming food waste and blackwater is forwarded directly to the UASB-ST digester which is operated at 25 °C and a hydraulic retention time of 30 days based on de Graaff et al. (2010). Digester effluent is treated for phosphorus recovery in a struvite precipitation chamber (STOWA, 2014; Wiersma and Elzinga, 2014) and for nitrogen recovery in an ammonia stripper (Jiang et al., 2014; Sagberg and Grundnes Berg, 2000). The effluent of the nutrient recovery processes is led as reject water in to the activated sludge treatment; the effluent from which is treated with post-precipitation according to (Lindquist, 2003) in order to meet the discharge demands, details are given in Appendix A1 and A2.

Calculations for the wastewater treatment plant included infrastructure, excavation, operation (electricity, heat, and chemicals), atmospheric emissions (CH_4 and N_2O), and heat recovery from the effluent and the same emissions to the recipient ocean water body as for the conventional system.

Infrastructure was based on a previous life cycle assessment of the construction of decentralized treatment plant in Sneek, Netherlands (Witteveenbos, 2014). The heat and electricity was calculated using empirical values from the same treatment plant (Meulman, 2015a; Meulman, 2015b; de Graaf & Van Hell, 2014) and mass balances by Kjerstadius et al. (2015) amended to include post precipitation (Lindquist, 2003) in order to meet the assumed Swedish discharge standard (10mgN/L and 0.5mgP/L). The final mass balances are presented in Appendix A1. According to Wiersma & Elzinga (2014), the amount of particles in the digester effluent of the UASB-ST is low (<800 mg COD/L). Thus, struvite precipitation was assumed possible directly on the digester effluent. Infrastructure, electricity and chemical usage ($MgCl_2$) for the specific process in a 12 000 capita unit was calculated by a supplier (Thelin, 2015). Ammonium stripping was assumed to occur at 65 °C using NaOH for pH increase and sulphuric acid for precipitation of nitrogen as ammonium sulphate. Infrastructure, electricity and chemical usage (NaOH and H_2SO_4) of the ammonia stripper for the specific process for a 12 000 capita unit was calculated by a supplier (Thelin, 2015). Overall, the massbalances for the source separation system are updated from Kjerstadius et al. (2015) according to Appendix A1. The data for struvite precipitation and ammonium stripping is given in Appendix A2.

Emissions during operation of the treatment plant were calculated using the same factors as for the conventional system for methane (from anaerobic degradation in sewers and slip during production of biogas from the anaerobic digester) and nitrous oxide (from BNR in the activated sludge system and effluent nitrogen in the recipient). No gaseous emissions were considered from struvite precipitation (Rodriguez-Garcia et al., 2014). No gaseous emissions were considered from ammonia stripping based on similar assumptions in literature (Jiang et al., 2014; Paccanelli et al., 2015; Sagberg and Grundnes Berg, 2000; Ten Hoeve et al., 2014; Vázquez-Rowe et al., 2015).

Emissions from chemical production were gathered from ecoinvent (2013) and global processes were used, indicating global average transport distances rather than actual ones. The exception was climate impact where the impact factor for post-precipitation was collected from Homa & Hoffmann (2014) and impact factor for NaOH was collected from Dahlgren et al. (2015) in order to simulate realistic Swedish conditions.

Heat recovery from the combined wastewater was assumed to be performed with heat pumps on the wastewater treatment plant effluent. The temperature of the mixed BW and GW stream was calculated to 23 °C and heat losses in sewer net and at the treatment plant were assumed to 4 °C in total. A temperature lift to 50 °C district heating was assumed, resulting in a coefficient of performance (COP) of 3.9, based on Hellborg Lapajne (2016). It should be noted that this is much higher than the current COP of the actual heat pump at the wastewater treatment plant in Helsingborg, which has a COP of 2.9-3.2 according to Baaring (2015). This is an effect of the present study not considering stormwater.

Heat recovery from the greywater effluent at the local treatment plant was assumed to be performed using a heat pump. Values for calculation was taken from Hellborg Lapajne (2016), based on calculations for the H+ area. Total extracted heat from greywater was 660 kWh cap^{-1} $year^{-1}$ to be compared with the assumed maximum of 800 kWh cap^{-1} year^{-1} by Larsen (2015) and a calculated COP of 4.7 due to the high temperature of the greywater and the small assumed losses of temperature due to the short distances in the sewer net. The temperature lift was assumed to be done to 50 °C district heating (Hellborg Lapajne, 2016).

Due the amount of heat needed for anaerobic digestion (performed at 25 °C according to de Graaff et al. (2010)) and ammonia stripping (performed at 65 °C according to Thelin, 2015) a heat exchanger was also assumed to be used after the ammonia stripper. The efficiency was calculated by the supplier of ammonia stripper (Thelin, 2015) and was calculated to reduce the heat required for the ammonia stripper from 16 to 11 kWh m^{-3}.

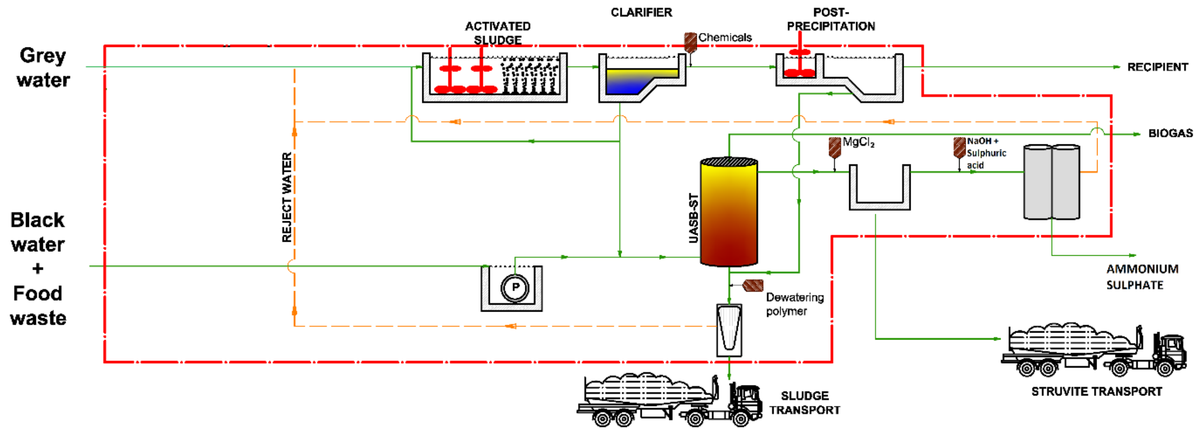


Figure 6 – Lay out of a wastewater treatment plant for the source separation system. Image based on Kjerstadius et al. (2015).

2.2.5. Biogas production, upgrading & usage – Source separation system

Produced biogas was considered to be upgraded at the wastewater treatment plant. Calculations were done in the same manner as for the conventional system (section 2.1.5). This included upgrading using water scrubbers and use of upgraded biogas in city buses, substituting diesel as fuel. Production of buses was assumed to be similar for both systems and thus excluded from the study.

2.2.6. Sludge & nutrient recovery management – Source separation system

In the source separation system three streams of solids are produced from the wastewater treatment plant; dewatered sludge from the anaerobic digester, precipitated struvite and ammonium sulphate. Of these streams, sludge was assumed handled in the same way as for the conventional system, i.e. with 43% going to agriculture and 57% being used for soil improver production. The sludge fraction was calculated in the same way as for the conventional system, including six month storage after which it was transported to agriculture (30km) or constructed soil (100km). Emissions during storage and spreading were the same as for sludge in the conventional system. It is worth noticing that the sludge produced in the source separation system does not contain the same amounts of nutrients or heavy metals as the sludge in the conventional system, as presented by the mass balances in Appendix A1.

The produced struvite and ammonium sulphate was assumed stored for six months, following the same spreading cycles in agriculture as for sludge, though no special container was needed for the storage. It was assumed that all of the produced struvite and ammonium sulphate was used in agriculture. The phosphorus of the struvite was assumed to have the same P availability as the phosphorus of the mineral fertilizer (Johnston & Richards, 2003), and thus the phosphorus content of the struvite replaced 100% of the P content of the mineral fertilizer. The nitrogen recovered as ammonia sulphate was assumed to replace 100% of the nitrogen of the mineral fertilizer. As the two fertilizers were mineral and thus inert they were assumed to cause no emissions at storage. Spreading and emissions from spreading was included for both recovered fertilizers and the replaced mineral fertilizers. Spreading equipment was the same as for mineral fertilizer and did thus not cause any impact on the result. The same was for emissions at spreading from soil as the nitrous oxide emissions would be the same for both fertilizers. No other emissions were assumed to take place at spreading. For data and references used see Table A2-6 in Appendix.

For transport of sludge the lorry (>35 ton, EURO 5) was assumed to have an empty return. For transport of dry fertilizers as struvite and ammonia sulphate the lorry was assumed to transport other goods on return and thus no return included. Cadmium content of the avoided phosphorus mineral fertilizer was assumed to be 5.0 grams per ton phosphorus (Statistics Sweden, 2015).

2.3. Main differences between the systems

In order to make comparison between the systems easier for the reader, the main differences for calculation purposes are highlighted in bullet-point below.

- The conventional system has larger benefits of scale (120 000 cap) while the source separation system is calculated only for 12 000 cap.
- Food waste is collected separately in the conventional system. In the source separation system, food waste is collected and treated together with wastewater.
- The conventional system uses only one large sewer pipe for wastewater management. The Source separation systems uses two smaller sewer pipes (one each for BW and FW) and one larger pipe (for GW).
- The conventional system uses one large central (120 000 cap) wastewater treatment plant with large areas for sedimentation basins (i.e. much more concrete for the wastewater treatment plant). The source separation system uses a smaller (12 000 cap) and more compact treatment plant (that uses more steel than the conventional system).
- Wastewater management in the conventional system is focused on removal of organic material and nutrients from wastewater. The source separation system is focused on increased recovery of biogas and nutrients from the wastewater.
- The conventional wastewater treatment plant uses enhanced biological nitrogen removal (i.e nitrification-denitrification), which potentially releases some nitrous oxide in to the atmosphere) for a larger fraction of the wastewater. In the source separation system, biological nitrogen removal is only done on a smaller fraction of the wastewater, while a majority of the nitrogen is recovered through the ammonia stripper.
- The source separation system uses struvite precipitation at the wastewater treatment plant, which increases the recovery of phosphorus to farmland.

2.4. LCA methodology

2.4.1. Type of LCA

An attributional LCI-modelling approach was used. The choice was based on recommendations provided in the ILCD Handbook (EC, 2010), as the study is aimed as decision support, but no large-scale consequences on processes in the background system are expected from the decisions. In addition, this approach was seen as relevant as the study considers comparison of two different systems that are to be completely newly built. Lastly, the chosen approach makes it easier to compare results with previous studies where this approach has been used, such as Remy (2010).

2.4.2. Scope and functional unit

The scope is management and recovery of energy and nutrients from household wastewaters (blackwater, greywater and food waste). To meet the scope the functional unit (FU) was selected as management of 1 person equivalent (P.E.) yearly load of food waste (FW), blackwater (BW) and greywater (GW).

$$FU = \frac{Management\ of\ 1\ P.E.load\ of\ FW, BW\ and\ GW}{year}$$

It is here implied that treatment is defined as collection, treatment and disposal according to Swedish law. Furthermore, discharge limits for wastewater treatments plants are assumed to 10mgN/L and 0.5mgP/L. It is noteworthy that the definition of the functional unit is similar to the study of Remy (2010) who used the term “provision of the primary functions” rather than the used word management in the present study.

For calculation purposes the FU corresponds to a daily mass of solids, organic material, phosphorus and nitrogen according to Jönsson et al. (2005).

2.4.3. Study boundary

The boundary for the life cycle assessment is given in Figure 7. Each system included infrastructure for collection, transport, treatment and nutrient recovery, as well as spreading of sludge on farmland or using sludge for soil improver. Emissions to water or air were considered from several processes (striped clouds in Figure 7). In general, all stationary infrastructure was included while no infrastructure for transports was considered. Management services (such as needed personnel) were not included and end-of-life treatment of infrastructures was not included since it had been shown to be negligible in a similar study (Hillenbrand, 2009). Details of included and excluded processes are given in Appendix A2.

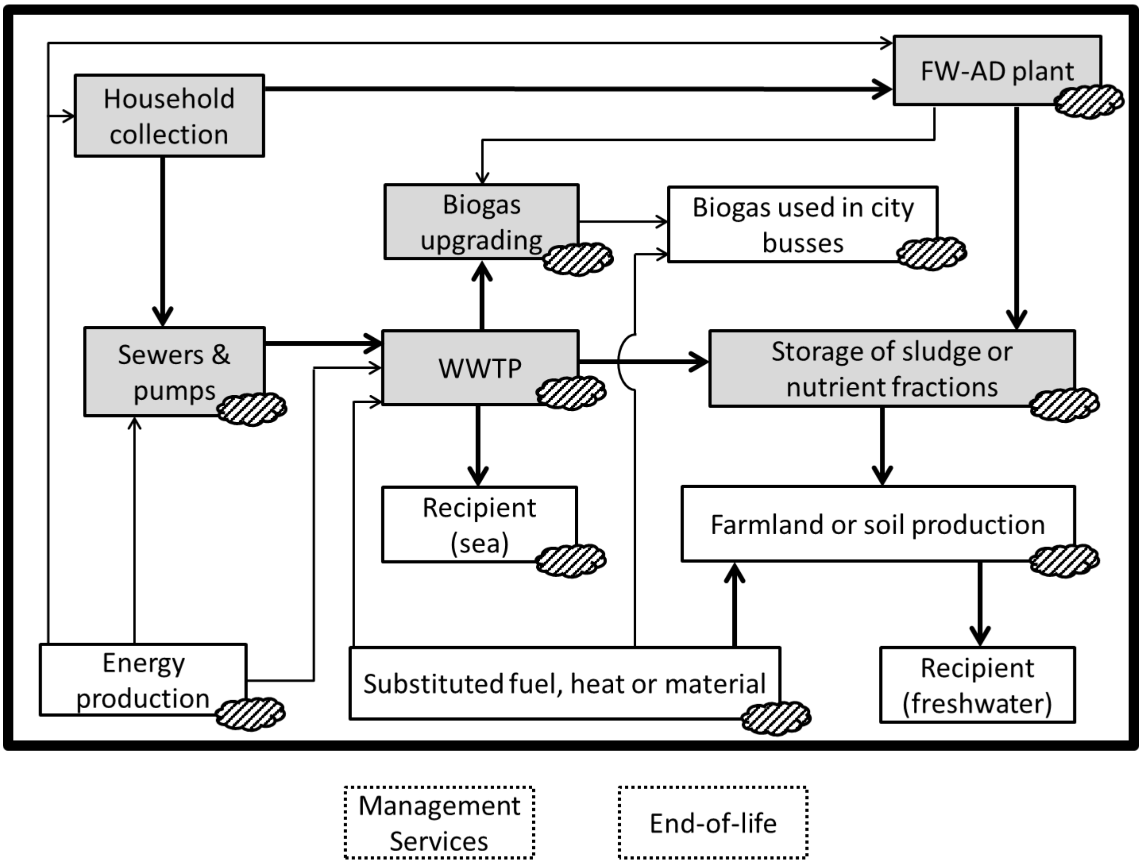


Figure 7. System boundary. Grey boxes indicate that infrastructure was included. Thick arrows indicate transport of materials. Striped clouds indicate emissions to air or water. Dashed boxes were not included in LCA. FW-AD plant is the food waste anaerobic digestion plant.

2.4.4. Handling of multifunctional processes

System expansion was used to investigate the potential environmental benefits related to use of biogas and nutrients recovered from waste fractions in the system.

Produced biogas was assumed to be used in the transport sector, substituting diesel in local buses on an energy content basis. Provision and use of substituted diesel is considered in the LCA.

Nutrients in digestate from anaerobic digestion of food waste and sludge which is applied on farmland were assumed to substitute production and use of mineral fertilizers on a plant available nutrient content basis.

Sludge which is not applied on farmland was not assumed to substitute production of other construction soils as it was seen as a mean to get rid of this fraction and not having a direct value.

Heat recovered from wastewater through the use of a heat pump was assumed to replace district heating produced from biofuels.

Electricity and heat produced in incineration of reject from pretreatment of food waste at the FW AD-plant was assumed to substitute average electricity and heat in the Swedish systems.

2.4.5. Time span of study

The time span for the study was 50 years, being the longest technical life span of some of the infrastructure components included in the study as well as being used in similar studies (Thibodeau, 2014; Witteveenbos, 2014). Thus, impacts from infrastructure will be evenly divided over the time span. The rationale behind this choice is to ease comparison to other LCA-studies. However, the real impact of the systems studied will of course not be evenly distributed since infrastructure will be built/replaced in specific periods of time.

2.4.6. Selection of environmental impact categories

The lifecycle impact assessment (LCIA) methodology ReCiPe was chosen for the classification and characterization steps in present study. The methodology was created by Dutch universities and consultants, and relies primarily on European background data for deposition and fate modeling (ReCiPe, 2016). The ReCiPe LCIA methodology contains 18 midpoint environmental impact categories; five related to human health, ten to ecosystem quality and three to resource use. Three impact categories were considered in the study; climate change, freshwater and marine eutrophication. The rational being to focus the study on issues assumed most important for the wastewater and food waste management due to several reasons. Firstly, climate change is a heavily debated topic and the production of biogas from food waste and wastewater, being central in the present study, has been justified from the perspective of reducing climate change impacts, both on national and regional level (Region Skåne, 2016; SEPA, 2016a). Additionally, nutrient recovery, being another central component to the study, has a potential to drastically effect climate change mainly due to the energy intensive Haber-Bosch process, used in production of nitrogen fertilizers, but also to some extent from production of phosphate (Jenssen & Kongshaug, 2003; IFA, 2009). Thus, the effect on climate change was decided an important impact category. Secondly, aquatic eutrophication (marine and freshwater) is a crucial issue in a Swedish context due to the eutrophication of streams and lakes as well as the sensitive Baltic Sea (SEPA, 2016b). Lastly, the above impact categories relate to the return of nutrients from sanitation systems to farmland, a practice that reduces the need of mineral fertilizer and the impact on climate change and eutrophication. To link the impact potential to the return of nutrients two impact categories regarding potential for nutrient return to farmland was included in the study. This decision was made due to the suggested legal demands for nutrient recovery (40% of P and 10% of N) from wastewater in Sweden (SEPA, 2013). The selected impact categories are presented in Table 1.

It should be noted that several other impact categories from the ReCiPe LCIA methodology could have relevance for the systems investigated. However, these were excluded both due to aim being focused on the SEPA (2013) legislation proposal as well as lack of appropriate data for the studied systems. For example, the fate of micro-pollutants could have been of interest for the study, some of which being mentioned in the proposal for a new EU Water Framework Directive (EU, 2013). However, as the input data did not include mass balances for more than seven heavy metals and no data for other potentially human and eco-toxic substances, any related impact categories were excluded from the assessment. Similar exclusion due to lack of appropriate data was also made in a similar study by Remy (2010).

Table 1. Impact categories considered in study.

Climate change	Freshwater eutrophication	Marine eutrophication	Return of nitrogen to farmland	Return of phosphorus to farmland
kg CO_2 eq	kg P eq	kg N eq	kg N	kg P

2.4.7. General data and assumptions

The LCI modeling was made in Simapro v.8 with use of ecoinvent database v.3.0 (ecoinvent, 2013). All processes were modelled using average European data (RER) if available in the ecoinvent database (ecoinvent, 2013). If not, global average data (GLO) was used. Datasets were adjusted to Swedish conditions when relevant (details in Appendix A2). For provision of electricity, average Swedish electricity, available in the ecoinvent v.3 database (low voltage) was used.

Data on environmental impacts from average Swedish district heating were based on Gode et al. (2011) and modelled in Simapro. In the case of peat, wood chip, and bio-oils, no processes for generation of heat were found in ecoinvent. In these cases, a process for combustion of wood pellets was adjusted to include provision of these fuels, while emissions were assumed to be the same as in the case of combustion of pellets – except for the case of peat, where emissions of fossil carbon dioxide were added to the emissions, based on Gode et al. (2011).

2.4.8. Specific data collection

Data was mainly collected from published literature. This included data over conventional food waste and wastewater management in Sweden or data over source separation systems from pilot areas in Europe. In addition, some data was collected from the municipal water utility of the city of Helsingborg (sewer net length and needed number of pumps as well as some data from the municipal wastewater treatment plant) or from suppliers of products assumed in the study (vacuum sewers, struvite precipitator and ammonia stripper). All data used are provided in Appendix A2.

Mass balances for the systems were based on (Kjerstadius et al., 2015) with some amendments. The source separation system was amended to include BNR in the activated sludge system according to data from an existing system (Wiersma & Elzinga, 2014; Wiersma, 2013). The effect of this on the mass balance was that effluent N was decreased from 20% to 15% of total incoming and the amount of NH_3 strip was decreased from 73% to 68%. The methane production is also slightly decreased to constituting roughly an increased potential of 60% compared to conventional system instead of 70% as reported by (Kjerstadius et al., 2015). The conventional system was amended with post-precipitation and increased BNR in order to meet the discharge demands (10 mgN L^{-1} and 0.5mgP L^{-1}), the latter being necessary since stormwater was not included in the present study. The addition of storm water would otherwise have diluted the concentration of nutrients in the wastewater and make excess treatment un-necessary. The full effect on the mass balances for both systems compared to Kjerstadius et al. (2015) are clearly stated in Appendix A1. As examples, the mass balances for total solids (TS), phosphorus and nitrogen used in the study are presented in Figures 8 and 9.

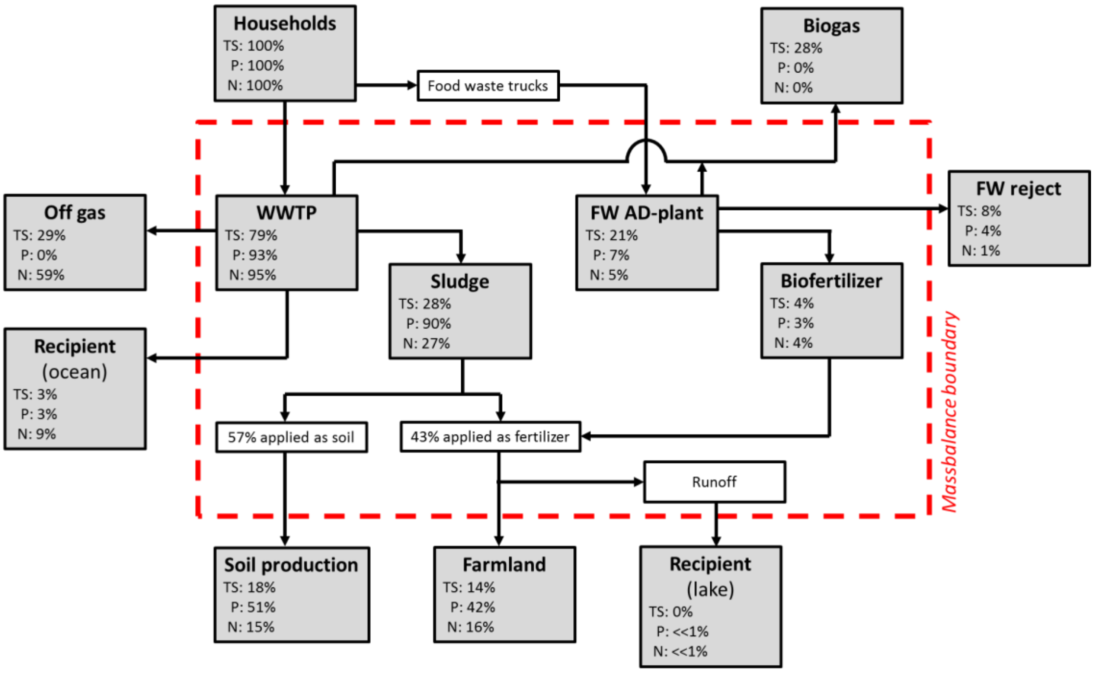


Figure 8 – Mass balances for total solids (TS), phosphorous (P) and nitrogen (N) over the conventional system.

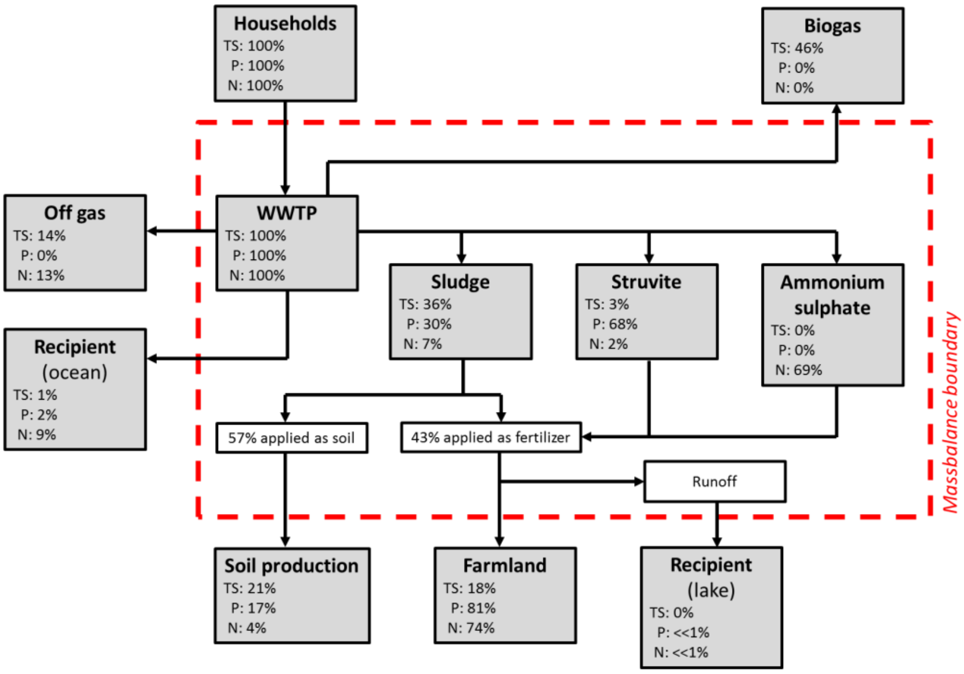


Figure 9 – Mass balances for total solids (TS), phosphorous (P) and nitrogen (N) over the source separation system.

In addition, albeit not being used in any of the impact categories, the mass balances for heavy metals in Kjerstadius et al. (2015), were compared to more recent data in Yoshida et al. (2015). Out of mass balances for seven heavy metals, three (Pb, Cd, Zn) showed good comparability between the studies while the mass balances for the remaining heavy metals (Cu, Cr, Hg, Ni) were upgraded according to Yoshida et al. (2015). Mass balances over heavy metals are given in tables A1-3 and A1-5 in Appendix A2.

3. Results & Discussion

The life cycle assessment was performed of the conventional system and source separation system in regards to three impact categories as well as return of nutrients to farmland.

3.1. Impact on climate change

As seen in Figure 10, the impact of both the source separation system and the conventional system on climate change are slightly negative, with the source separation system having the largest decrease in impact on climate change.

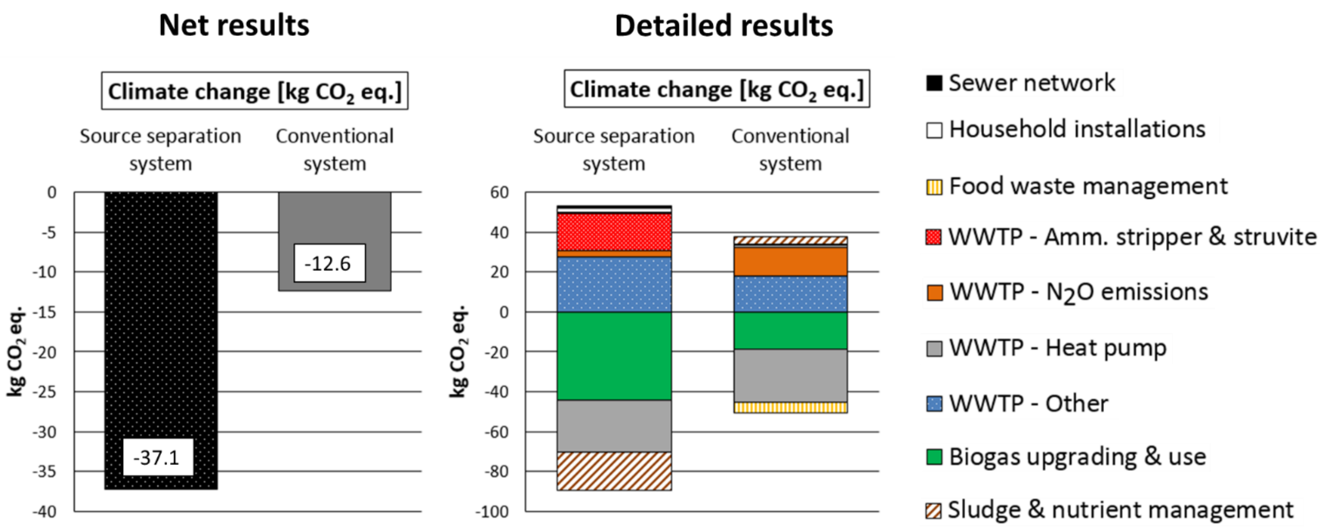


Figure 10 – Results for impact on climate change. Results given per functional unit (FU).

For the conventional system the main contributors to climate change stem from operations at the wastewater treatment plant; the largest being emissions of N_2O from biological nitrogen removal (WWTP – N_2O emissions constituting 15 kg CO_2-eq. capita^{-1} year^{-1}). Other emissions from the wastewater treatment plant (WWTP - Other) were 18 kg CO_2-eq. capita^{-1} year^{-1}. The largest contributors to WWTP-Other was heat and electricity usage (7 kg CO_2-eq. capita^{-1} year^{-1}), infrastructure (7 kg CO_2-eq. capita^{-1} year^{-1}) and emissions of methane from wastewater and nitrous oxide from the WWTP effluent (together 5 kg CO_2-eq. capita^{-1} year^{-1}). The impact of chemicals used for sludge dewatering and post precipitation is almost negligible (1 kg CO_2-eq. capita^{-1} year^{-1}). In total the wastewater treatment plant processes (WWTP – Other and WWTP – N_2O emissions) constitute 33 kg CO_2-eq. capita^{-1} year^{-1}, not including the heat pump or the biogas upgrading and usage. The heat pump (WWTP – Heat pump) and biogas upgrading and usage (biogas upgrading & use) both constitute large sinks to the climate change impact category. The heat pump constitute the largest sink (-26 kg CO_2-eq. capita^{-1} year^{-1}) for the conventional system, a large decrease compared to heat pumps for conventional treatment plants as reported by Gustavsson & Tumlin (2013). The large sink of the heat pump is an effect of the chosen system boundaries, which excludes stormwater handling. This is analyzed in further details in section 4.1 below. The other large climate change sink is upgrading and usage of biogas from the food waste plant and wastewater treatment plant. Here, the large beneficial impact (-19 kg CO_2-eq. capita^{-1} year^{-1}) on decreasing climate change stem from replacing diesel as vehicle fuel in city buses, a common application for biogas in Southern Sweden (Biogasportalen, 2015). The Biogas upgrading & use section in Figure 10 includes assumed slips of methane in biogas upgrading and addition of propane to reach gas grid energy standards as stated in Appendix A2. Avoided emissions of greenhouse gases (GHG) also stem from food waste management where the reject from pretreatment of collected food waste, using screw press, is incinerated and assumed to replace electricity and district heating; an effect that gives the food waste management a slight negative impact on climate change. A slight net contribution to climate change, stemming from sludge & nutrient management, was also seen in the conventional system. This fraction includes transport of sludge for either return to agriculture (43 % by mass) or for use as constructed soil (57 % by mass). Sludge returned to agriculture cause emissions of nitrous oxide and methane during storage and spreading, but also replaces mineral fertilizer and cause carbon sequestration. The nitrous oxide emissions dominated the impact of these emissions, about 97%, and dominated the total impact from sludge & nutrient management. Sludge used as constructed soil is composted (which results in emissions of GHG, mainly from methane emissions) and cause carbon sequestration only when applying. The net effect of these processes is a slight contribution (5 kg CO_2-eq. capita^{-1} year^{-1}) to climate change. Lastly, it should be noted that household installations (piping) and sewer net have a negligible (2 kg CO_2-eq. capita^{-1} year^{-1}) impact on climate change compared to the other sources of impact for the conventional system.

Also for the source separation system, large contributors to climate change stem from the wastewater treatment. Wastewater treatment, excluding nutrient recovery by the ammonia stripper and precipitation of struvite, constitute 27 kg CO_2-eq. capita^{-1} year^{-1}. The main sources for the impact are infrastructure, due to the increased need of steel (Witeveen Bos, 2014), (7 kg CO_2-eq. capita^{-1} year^{-1}) and emissions of methane from wastewater and nitrous oxide from the WWTP effluent (together 9 kg CO_2-eq. capita^{-1} year^{-1}). Slip of methane from biogas production (4 kg CO_2-eq. capita^{-1} year^{-1}) are larger than for the conventional system due to the increased biogas production. Other contributors are the heat & electricity use (8 kg CO_2-eq. capita^{-1} year^{-1}) and infrastructure (7 kg CO_2-eq. capita^{-1} year^{-1}). The effect of chemical usage due to post precipitation is rather small (1 kg CO_2-eq. capita^{-1} year^{-1}). A large contribution, presented separately in Figure 10, is that of ammonia stripping and struvite precipitation (20 kg CO_2-eq. capita^{-1} year^{-1}). Almost all of this impact (16 kg CO_2-eq. capita^{-1} year^{-1}) stem from the use of heat and chemicals (sodium hydroxide, citric acid and sulphuric acid) in the ammonia stripper. The high impact can be related to the relatively dilute concentrations of nitrogen (1 000 mg NH_4-N L^{-1}) in the wastewater which still demands high temperature (65 °C) and chemical usage. The high impact of the ammonia stripper does not take into account the benefit of the recovered nitrogen which is presented in the sludge & nutrient management fraction of the diagram in Figure 10. It can be seen that this fraction constitute a rather large decrease (-21 kg CO_2-eq. capita^{-1} year^{-1}) of the climate change, compared to the positive impact of these processes in the conventional system. The large decrease is in fact almost solely due to the increased return of nitrogen from the ammonium stripper (-22 kg CO_2-eq. capita^{-1} year^{-1}), the returned struvite constituting only a smaller (-1 kg CO_2-eq. capita^{-1} year^{-1}). Sludge & nutrient management also has minor contribution (2 kg CO_2-eq. capita^{-1} year^{-1}) to global warming from sludge transports and storage (mainly due to the handling of sludge). Another noticeable difference between the source separation system and the conventional system is the much smaller impact of nitrous oxide emissions from the source separation wastewater treatment (WWTP - N_2O emissions); the latter being only 3 kg CO_2-eq. capita^{-1} year^{-1} compared to the 15 kg CO_2-eq. capita^{-1} year^{-1} in the conventional system. The reason being the decreased nitrogen removal through activated sludge system in the source separation system were only greywater and reject water from the anaerobic digestion is treated in biological nitrogen removal. The large contribution to climate change due to nitrous oxide emissions from wastewater handing has previously been reported as a major contributor to the overall impact of wastewater and sludge management (Gustavsson & Tumlin, 2013) as well as for studies over systems similar to the present study (Remy, 2010; Hillenbrand, 2009) and will be discussed in detail in section 4.2. Two processes that causes decrease in climate change are, similar to the conventional system, the heat pump and the biogas usage. The heat pump causes a large decrease (-26 kg CO_2-eq. capita^{-1} year^{-1}) due to potential for extracting heat from the source separated greywater. The effect of the heat is further discussed in section 4.1. The usage of biogas to replace fossil vehicle fuel causes the largest decrease in climate change (-44 kg CO_2-eq. capita^{-1} year^{-1}) which is larger than for the biogas produced by the conventional system due to larger amount of biogas being produced in the source separation system. The increased biogas production (60 % higher than the conventional system) is mainly due to less organic material being treated in activated sludge treatment in the source separation system, where blackwater and food waste is directly treated in the anaerobic digester. In addition, some of the increase is explained by avoiding some losses of food waste that for the conventional system occurs in the pretreatment of food waste by screw press. Both of these effects presented in earlier references on which the mass balances in the present study are based (Kjerstadius et al., 2015, 2012). Lastly, it should be noted that household installations (piping) and sewer net have a very small (3 kg CO_2-eq. capita^{-1} year^{-1}) impact on climate change compared to the other sources of impact for the source separation system.

3.1.1. Comparison to other studies

When comparing the results for climate change to other studies (Table 2) it is clear that the result in the present study (-13 to -37 kg CO_2-eq. capita^{-1} year^{-1}) are in the lower range of reported values (ranging -22 to 315 kg CO_2-eq. capita^{-1} year^{-1}). Albeit being studies on source separation systems the reason for the range in results can be explained by difference in studied systems, system boundaries and in-data. For example, the high impact of systems in other studies is to some extent due to using indata for more fossil intensive electricity production compared to Swedish data; for comparative reasons, the average European electricity mix (0.415 kg CO_2-eq. kWh^{-1}) (Elforsk, 2008) can be compared to the Swedish electricity mix (0.067 kg CO_2-eq. kWh^{-1}) (ecoinvent, 2013) used in the present study. In fact, Hillenbrand (2009) stated that the impact on climate change was constituted out of electricity use to roughly 40% for the conventional system and 65% for the source separation system, which alone would correspond to roughly 100 kg CO_2-eq. cap^{-1} year^{-1} (conventional system) and 190 kg CO_2-eq. cap^{-1} year^{-1} (source separation system). The relationship between energy usage and climate impact can also be seen when comparing net energy demand and climate impact (Table 2) where studies reporting high energy demand also report high impact on climate change and vice versa. The present work stands out as having a high recovery of heat (negative impact) due to the usage of heat pumps, the impact of which is further discussed in section 4.1. Another reason for the difference compared to other studies could also be emission factors for emissions of methane and nitrous oxide from storage and spreading of sludge, which proved to have a large impact in this study as seen in Figure 10, since the emissions factors depend on many variables such as storage time, soil type and spreading technique (Spånberg et al., 2014; Thibodeau, 2014). Finally, the production of biogas from food waste in both systems as well as the use of a Swedish electricity in the present study decreases the impact on climate change compared to wastewater treatment alone, as seen when compared to the values of Gustavsson & Tumlin (2013). In conclusion, there is a large variation in reported values for climate impact and the present study states relatively very low values for climate impact for both the conventional and the source separation system.

Table 2. Comparison of annual climate impact of sanitation systems. Results given in units of person equivalents (P.E.) or per capita.

System	Climate impact [kg CO_2-eq. cap^{-1} $year^{-1}$]	Net energy demand [kWh cap^{-1} $year^{-1}$]	Comment	Reference
Conventional	140 per P.E.	333[1] / 160[3]	SCTS pilot area, Germany	Remy (2010)
Source separation	85 per P.E.	250[1] / 220[3]		
Conventional	244 per capita	860[1]	DEUS 21 pilot area, Germany	Hillenbrand (2009)
Source separation	315 per capita	1 500[1]		
Conventional	32-40 per capita	88[2]	Noorderhoek pilot area, Netherlands	Witeveen Bos (2014) & STOWA (2014)
Source separation	-22 per capita	-184[2]		
Conventional	52.8 per capita	219[2]	Hypothetical area, Quebec, Canada	Thibodeau (2014)
Source separation	65.3 per capita	274[2]		
Conventional	32 per capita	122[2]	Hypothetical area, Sweden. Food waste and heat pump not included.	Spångberg et al. (2014)
Source separation	21 per capita	33-37[2]		
Swedish WWTP´s	7-108 (average 46) per P.E.	-	WWTP operation and sludge return. Used European electricity mix. Food waste not included.	Gustavsson & Tumlin (2013)
Conventional	-13 per capita	$165_{electr.}$[3] $-393_{thermal}$[3]	Hypothetical area, southern Sweden.	Present study.
Source separation	-37 per capita	$119_{electr.}$[3] $-281_{thermal}$[3]		

1) calculated from cumulative energy demand. 2) calculated from primary energy use. 3) only includes energy for operation.

3.1.2. Normalizing impact on climate change

In order to put the results for impact on climate change in to perspective the results were normalized against selected other impacts. The normalization was done against selected relevant impacts and is presented in Table 3 together with the normalized impact of the studied systems given as percentage of the reference value. It is clear from Table 3 that the emissions from the studied systems are low in comparison to current per capita emissions in Sweden (SEPA, 2016e). Additionally, the emissions from the studied systems are low compared to the suggested planetary boundary (Nykvist et al., 2013) which has been suggested as a limit for safe operating space oh humanity. Lastly, comparing the studied systems against the reported emissions from wastewater treatment and biological treatment of solid waste (here assumed representative of food waste) it can be seen that source separation systems potentially can have a large effect on reducing emissions. It should also be noted that the conventional system in the present study have a much smaller impact than the reported average. This is likely mainly due to the positive effect of the heat pump when excluding stormwater from the study, as discussed in section 4.1. In summary, the impact of management of wastewater and food waste on climate change is very small compared to other emissions in Sweden.

Table 3 – Normalization of impact of climate change against selected literature values. Results for the conventional system (-13 kg CO_2-eq. cap^{-1} $year^{-1}$) and the source separation system (-37 kg CO_2-eq. cap^{-1} $year^{-1}$) are normalized as percentage of the reference values.

	Tonnes $year^{-1}$	**kg CO_2-eq. cap^{-1} $year^{-1}$**	**Reference**	**Conventional system**	**Source separation system**
2014 gross positive emissions	99.5*10^6	10 250[1]	SEPA (2016)	~0%	~0%
2014 net emissions	54.4*10^6	5 600[1]	SEPA (2016)	~0%	~0%
Suggested planetary boundary	18*10^6	2 000[2]	Nykvist et al. (2013)	~0%	-2%
2014 emissions for treating wastewater and biological treatment of solid waste[3]	365*10^3	37.6[1]	SEPA (2016)	-34%	-198%

1) Assuming a population of 9.7 M in Sweden. 2) Assuming a population of 9.0 M in Sweden. 3) Out of which 236 tonnes are for wastewater treatment and 129 tonnes for biological treatment of waste (assumed to be representative for food waste).

3.2. Impact on freshwater eutrophication

The impact on freshwater eutrophication is shown in Figure 11. The net results show that the impact from the source separation system is 70% higher than for the conventional, mainly due to the extraction of nutrients (WWTP – amm. stripper & struvite) at the wastewater treatment plant in this system. Specifically, almost all of the impact of the nutrient recovery is due to the usage of sodium hydroxide and sulphuric acid in the ammonia stripper.

Apart from this excess impact, both systems have quite similar sources of impact except for the sludge & nutrient management and food waste management being a larger source to freshwater eutrophication for the conventional scenario. The major part of the contribution to freshwater eutrophication from food waste management in the conventional system is related to production of paper bags for separate collection of food waste, or more specifically, provision of biomass for paper production. The reason for the greater impact from sludge & nutrient management in the conventional system is due to the larger amount of nutrients being returned to agricultural soil with organic fertilizers as sludge and digestate which cause relatively larger risk of freshwater eutrophication, both at storage and spreading, than nutrients in a more inert form as mineral fertilizers, struvite and ammonia sulphate. Freshwater eutrophication was calculated using characterization factors of ReCiPe which was slightly larger for mineral fertilizer than for manure added to agricultural soil. But as phosphorus added to soil with the organic substrates sludge and digestate has a lower plant-availability than the mineral fertilizers replaced, these fractions added in total larger amounts of phosphorus which in total caused a larger result on freshwater eutrophication. In this study leakage of phosphorus was set to about 5% of P added to agricultural soil which was based on values used in ReCiPe (2016). This is a relatively high percentage as the average is around 3% when considering average use of phosphorus per hectare and average leakage of P in Sweden (SMED, 2016; Statistics Sweden, 2014). Leakage was thus not underestimated in the present study.

There are few other LCA studies on similar systems, e.g. with ammonia stripping and struvite production for wastewater treatment, and many studies also include eutrophication as a common result for both emissions of nitrogen and phosphorus, using .e.g. PO_4^{3-}-equivalents. In general one can see the results of a higher chemicals use and energy input as in the WWTP treatment in Spångberg et al. (2014) which mainly had impact on climate changebut also to some extent on eutrophication. Also in that study nutrient management had the greatest impact on eutrophication. This can also be seen in Willén et al., (*not published*) where sludge was treated with urea in one scenario, this chemical use had an impact mainly on climate change and to a very small extent on eutrophication, which results were dominated of the impact from management of the sludge.

Only a minor impact on freshwater eutrophication is due to wastewater treatment plants. This might seems contradictory giving the discharge of phosphorus in the effluents from wastewater treatment plants. However, since the treatment plants in the present study are assumed to discharge their effluents in the ocean (marine water body) this phosphorus have no effect on the freshwater eutrophication potential according to the Recipe method (Recipe, 2016). The impact from wastewater treatment plants in the present study is caused mainly by use of electricity in the treatment processes, which is higher for the conventional system. The contribution to freshwater eutrophication from the Swedish electricity mix is mainly caused by P-emissions from provision of biofuels, resulting in an overall contribution to freshwater eutrophication of $4.3*10^{-5}$ kg P-eq. per kWh electricity. Additionally, the use of chemicals for nutrient recovery in the source separation scenario (WWTP – Amm. Stripper & struvite) cause a large impact on freshwater eutrophication. This is mainly due to the use of NaOH and H_2SO_4 used in the ammonium stripper which, even though both chemicals are assumed produced in Sweden, cause a large impact due to the energy use in the production (NaOH) or emissions in the entire production chain (H_2SO_4). The intensive use of chemicals for the ammonium stripper is due to the dilute digester effluent (estimated to 1000 mgNH_4-N L^{-1} based on real life data in Wiersma & Elzinga (2014)) and some organic material effluent (estimated to 800 mg COD L^{-1} based on real life data in Wiersma & Elzinga (2014)) that increase use of NaOH. Since the calculation on chemical use (Thelin, 2015) was precautionary calculated of NaOH the use of the NaOH might be exaggerated; none the less it stands to be concluded that the use of an ammonium stripper for nutrient recovery will have large impact in freshwater eutrophication. For optimized source separation systems there is thus a need for a different technology for nutrient recovery. In conclusion, the source separation causes greater freshwater eutrophication than the conventional system due to the use of chemicals in the ammonium stripper.

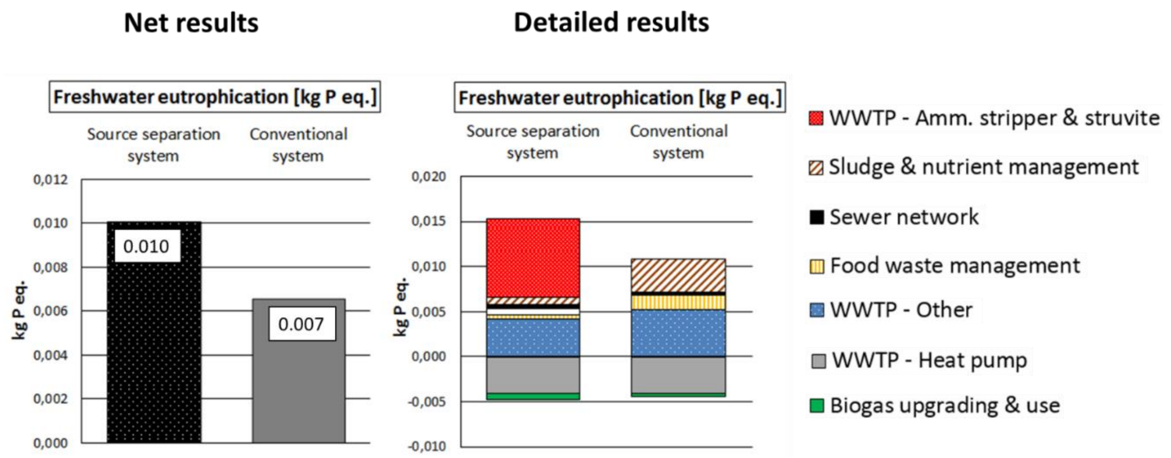


Figure 11 – Results for impact on freshwater eutrophication. Results given per functional unit (FU).

3.2.1. Comparison to other studies

To put the results in relation to other studies they are presented in Table 4 together with selected studies comparing systems similar to the ones investigated in the present study. It is clear from the table that the results from the present study are lower than any reported values. This is likely due to the present study assuming discharge of wastewater treatment plant effluent on to the ocean (causing only marine eutrophication) while the other studies in Table 4 considers discharge in to freshwater bodies. This reasoning is also supported by the fact that direct discharge of nitrous compounds and phosphorus from wastewater treatment plant is the main source of eutrophication potential in the studies were detailed results can be found (Meinzinger, 2010; Remy, 2010; Hillenbrand, 2009). Additionally, in at least one case higher discharges limits for the wastewater treatment plants was assumed for the other studies. The study by Remy (2010) assumed discharge limits of 2 mg P L^{-1} (compared to 0.5mg P L^{-1} in the present study) and 18 mg N L^{-1} (compared to 10mg N L^{-1} in the present study), the increased discharge of nutrients due to higher limits also increases the freshwater eutrophication. Contrarily to the high impact from WWTP effluents in other studies, the impact on freshwater eutrophication in the present study is mainly due to electricity use for wastewater treatment plants and for chemical production (WWTP – amm.stripper & struvite) as well as from emissions from sludge storage (sludge & nutrient management) as explained above. If direct discharge of phosphorus from the WWTP effluent should have been added to the results in the present study the impact would have been 0.023 kg P cap^{-1} $year^{-1}$ for the source separation system and 0.031 kg P cap^{-1} $year^{-1}$ for the conventional system which would have been in the range of values reported in the cited studies. Thus, the difference between the results in the present study compared to the other studies can be explained by the present study assuming discharge of wastewater treatment plant effluent in to the ocean, thereby avoiding freshwater eutrophication.

The reported values in other studies ranges from 0.023-0.17 kg P cap^{-1} $year^{-1}$ for the conventional system and 0.023-0.18 kg P cap^{-1} $year^{-1}$ for the source separation system, thus having roughly the same reported range. However, there is no consistency in what system has been reported to have lower impact on freshwater eutrophication; the conventional system being presented to have a lower impact in two studies (Witeveen Bos, 2014; Hillenbrand; 2009) while the source separation system have been reported to have a lower impact in the other two studies (Meinzinger, 2010; Remy, 2010). Thus, comparison to other studies leaves inconclusive results in regards to what system could generally have been said to have a lower impact in freshwater eutrophication. One study (Hillenbrand, 2009) stands out in particular for having reported much higher values for this impact category. These high values are due to a higher calculated release of phosphorus in to water bodies; likely being an effect due to assumed higher discharge limits since the effects are similar for both systems. When comparing the studies it should be stressed that only one of the cited studies (Witeveen Bos, 2014) used an impact category labelled Freshwater eutrophication similar to the present study, the other cited studies in Table 4 used combined eutrophication impact categories for freshwater and marine water and the results thus had to be re-calculated in to comparable values. However, the re-calculated values, excluding the study by Hillenbrand (2009), don't differ greatly from each-other. In conclusion the results from the present study are much lower than any reported values, an effect of the present study considering discharge of wastewater treatment plant effluent in to marine water bodies rather than freshwater water bodies, as done in the reference studies.

Table 4. Comparison of results for freshwater eutrophication compared to results from similar studies. Results given in units of kg P per capita and year.

	Remy (2010)	**Hillenbrand (2009)**	**Meinzinger (2010)**	**Witeveen Bos (2014)**	**Present study**	**Unit**
			Freshwater Eutrophication			
Conv. system	0.046[1]	0.15[2]	0.070[3]	0.023-0.026	0.007	kg P cap^{-1} $year^{-1}$
Source sep. system	0.023[1]	0.18[2]	0.056[3]	0.054	0.010	kg P cap^{-1} $year^{-1}$

1) Re-calculated from joint results for eutrophication. 2) Re-calculated from joint results for aquatic eutrophication. 3) Re-calculated from joint results on eutrophication to soil and water.

3.2.2. Normalizing impact on freshwater eutrophication

Normalization of the impact of freshwater eutrophication is difficult due to the lack of a clear choice of normalization data. The present study considers management of food waste and household wastewater and thus reasonably should be normalized against the total national impact on freshwater eutrophication of such management in Sweden. However, no such values was found in the present study. However, since a majority of the impact on freshwater eutrophication for both systems stems from processes related to wastewater treatment (Figure 11) or sludge use the decision was made to normalize against the discharge of phosphorus from Swedish wastewater treatment plant effluents. Additionally, to give a sense of proportion the results are also compared to the net accumulation of phosphorus in Sweden (net sum of import/deposition and removal processes). The results are given in Table 5 and it should be reminded that the majority of the impact from the systems stem from diffuse sources while being normalized against direct mass transfer of phosphorus. The results show that the eutrophication potential for the investigated systems constitute a large potential compared to the impact from wastewater treatment plants; the conventional system having an impact equivalent to 64% of the impact of Swedish wastewater treatment plants and the source separation system having an impact equal to 91%. However, compared to the national net accumulation of phosphorus in Sweden either system have only a small contribution (6% for the conventional system and 9% for the source separation system). This is not surprising since the majority of the phosphorus in the investigated system ends up being returned to farmland or soil production, from which only a small percentage affects freshwater eutrophication by agricultural run-off. In conclusion, the impact of the investigated systems on freshwater eutrophication constitutes only a minor part compared to the overall net accumulation of phosphorus in Sweden.

Table 5 – Normalization of impact on freshwater eutrophication against selected literature values. Results for the conventional system (0.007 kg P. cap^{-1} year^{-1}) and the source separation system (0.010 kg P. cap^{-1} year^{-1}) are normalized as percentage of the reference values.

	Tonnes year^{-1}	kg cap^{-1} year^{-1}	Reference	Conventional system	Source separation system
Release of phosphorus from WWTP's	290	0.032[1]	SEPA (2013)	22%	31%
Release of phosphorus from WWTP's	260	0.029[1]	Statistics Sweden (2016b)	24%	34%
Of which fresh water	99	0.011[1]	Statistics Sweden (2016b)	64%	91%
Net annual accumulation of phosphorus in Sweden	9 788	1.09	SEPA (2013)	6%	9%

1) Assuming a population of 9.0 M in Sweden.

3.3. Impact on marine eutrophication

Comparing the results for impact on marine eutrophication (Figure 12) it is clear that the source separation system have a smaller impact (0.44 kg N eq. FU^{-1}) than the conventional system (0.49 kg N eq. FU^{-1}). Looking at the detailed results it is clear that the higher impact for the conventional system stem from sludge and nutrient management. The impact from sludge & nutrient management was mainly due to the larger amount of nutrients being returned to agricultural soil with organic fertilizers as sludge and digestate, which caused more leakage due to their higher degree of organically bound nutrients. This was also seen for freshwater eutrophication (section 3.2). Added to this, for marine eutrophication, also the larger amount of ammonia emissions from storage and spreading of the sludge and digestate in the conventional scenario contributed to the higher results.

Looking further at the detailed results it is clear that the majority of the impact on marine eutrophication stem from wastewater treatment plants (WWTP – Other) for both systems. This impact is due to direct release of nitrogen in the effluent from wastewater treatment plant, being assumed to be discharged in to the ocean. This impact is similar for both system due to the wastewater being treated down to the same discharge limit (10 mgN/L), however not exactly the same due to different amounts of water flowing in the systems (Table A1-2 and A1-4 in Appendix A1) which effects the concentration of nitrogen in the wastewater. The amount of nitrogen leaving the systems in the wastewater treatment plant through the effluent roughly corresponds to 9% of the total nitrogen in the mass balances for the systems, as evident the Figure 8 and 9 as well as the detailed mass balances being presented in Appendix A1. It can also be noted that there is a large difference in impact from sludge and nutrient management between the investigated systems. This difference stems from differences in runoff (as mentioned above), but also from different amounts of nitrogen that is lost at storage and spreading and thus cause a difference in the total amount of nitrogen reaching the agricultural soil. In addition, heat pumps used to replace average Swedish district heating result in avoidance of marine eutrophication, due to the emissions of primarily NO_x related to combustion of fuels in the Swedish average district heating mix, dominated by biofuels. Biomass combustion exhibits relatively high emissions of NO_X in comparison to the combustion of natural gas or light fuel oil (Nussbaumer, 2010).

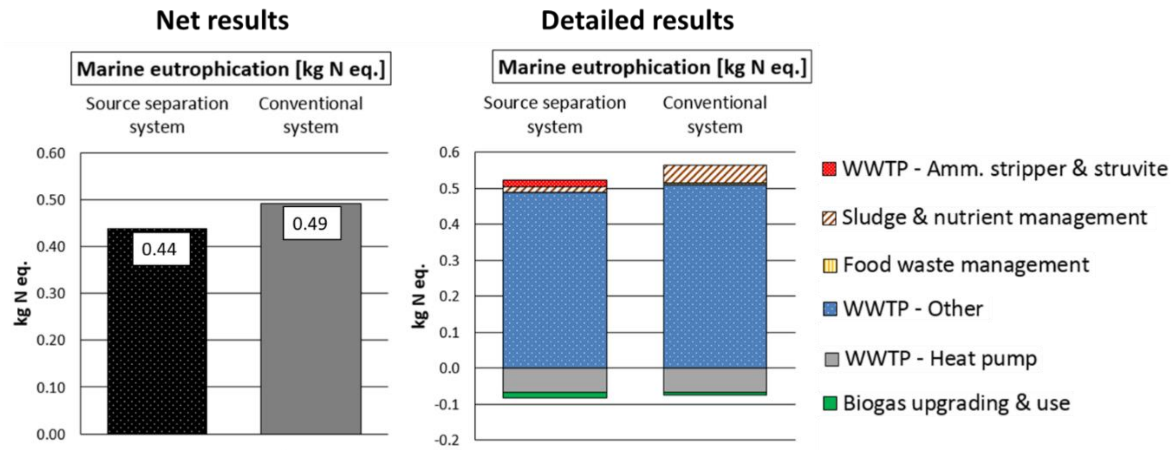


Figure 12 – Results for impact on marine eutrophication. Results given per functional unit (FU).

3.3.1. Comparison to other studies

Table 6 presents reported values for impact on marine eutrophication in selected studies similar to the present work. The reported values ranges from 0.43-1.24 kg N cap^{-1} $year^{-1}$ for the conventional system and 0.05-0.48 kg N cap^{-1} $year^{-1}$ for the source separation system, with the results from the present study being in the mid-region (for the conventional system) or upper part (source separation system) of these reported values. For the conventional system two of the cited studies (Witeveen Bos, 2014; Remy, 2010) states lower impacts compared to the present study while the other two (Meinzinger, 2010; Hillenbrand, 2009) states higher impacts. For the source separation system only the study by Meinzinger (2010) states a higher impact. Being between reported values, the results for impact on marine eutrophication in the present study can likely be seen as a fair approximation of the impact of the studied systems. It should here be explained that for comparative reasons the results from the cited studies have in most cases been re-calculated from their original values (according to notes in Table 6). In order to compare the potential impact on marine eutrophication values for discharge of nitrogen from wastewater treatment plants from the cited studies were used in Table 6 when no specific data was found. Since direct discharge of nitrogen from WWTP effluents was the main source of impact in the present study this simplification can be justified for comparative reasons.

Furthermore, when comparing the results from the studies in Table 6 it becomes apparent that the source separation system have always been stated to give a smaller impact on marine eutrophication than the studied conventional systems. This reduced effect corresponds to a range between 0.31-0.95 kg N cap^{-1} $year^{-1}$ in the cited studies, being considerably larger than the difference between the systems (0.05 kg N cap^{-1} $year^{-1}$) in the present study. The small difference in the present study being due to the treatment plants releasing close to the same amount of nitrogen through their effluents, it becomes interesting to study the reason for the large variation seen in literature. The main difference is found in the study by Hillenbrand (2009) and it is clear from his results that he considered the source separation system to release much less nitrogen through the effluent from the wastewater treatment plant; likely being an effect of the usage of nitrogen recovery through an ammonium stripper to a greater extent than assumed in the present study. In conclusion, the results from the present study can be seen as fair approximations of the impact on marine eutrophication and the source separation system have always been found to have a smaller impact than the studied conventional systems.

Table 6. Comparison of results for marine eutrophication compared to results from similar studies. Results given in units of per capita and year.

	Remy (2010)	**Hillenbrand (2009)**	**Meinzinger (2010)**	**Witeveen Bos (2014)**	**Present study**	**Unit**
			Marine Eutrophication			
Conv. system	0.43[1]	1.24[2]	0.81[3]	0.036-0.040	0.49	kg N cap^{-1} $year^{-1}$
Source sep. system	0.12[1]	0.24[2]	0.48[3]	0.05	0.44	kg N cap^{-1} $year^{-1}$

1) Re-calculated from joint results for eutrophication. 2) Re-calculated from joint results for aquatic eutrophication. 3) Re-calculated from emissions to soil and water

3.3.2. Normalizing impact on marine eutrophication

Since the majority of the impact on marine eutrophication stems from discharge of nitrogen from the wastewater treatment plants it is reasonable to compare the obtained results to national values on discharge of nitrogen from wastewater treatment (Table 7). Normalizing the obtained results against national statistics (Statistics Sweden, 2016b) it becomes apparent that both systems only constitute 27-28% of the national Swedish discharge per capita. This low amount could be explained by two reasons. Firstly, due to the exclusion of stormwater from the present study the amount of nitrogen leaving the conventional treatment plant becomes roughly half of a real life system. In the present system only 9% of the nitrogen which enters the wastewater treatment plant in the conventional system leaves via the effluent while standard values for a similar treatment plant including stormwater have been considered to be around 20% (Siegrist, 2008). The difference is due to the low discharge limit (10 mg N/L) in which the inclusion of stormwater dilutes the effluent and allows the discharge of more nitrogen while still meeting the discharge limit. The amount of stormwater entering wastewater treatment plants in Sweden has been shown to vary greatly (Molander, 2015) which affects the amount of nitrogen released. Since stormwater is excluded from the present study the discharge limit for the conventional system had to be reached with more intensive biological nitrogen removal as well as some post-precipitation (see section 2.1.4, section 2.2.4 and appendix A2). Taking this in to account the conventional system in the present study has considerable less impact than would a real-life system. For the source separations system this would not have been the case due to stormwater exclusion being part of the concept of source separation (Kjerstadius et al., 2015). The second reason for the discrepancy against the normalization values could possibly be found in the differences on discharge limits for wastewater treatment plants in Sweden. The different discharge limits, together with the highly variable amount of stormwater entering different wastewater treatment plants (Molander, 2015), will be an additional parameter that could explain the difference between the values for the conventional system in the present study and real-life wastewater treatment plants. In conclusion, the impact on marine eutrophication from the conventional system in the present study is likely an underestimation of a real life system due to the exclusion of stormwater from the present system. The source separation system will thus be a fair representation due to the exclusion of stormwater from such systems being central to the source separation concept.

Table 7 – Normalization of impact of marine eutrophication against selected literature values. Results for the conventional system (0.57 kg N cap^{-1} $year^{-1}$) and the source separation system (0.46 kg N cap^{-1} $year^{-1}$) are normalized as percentage of the reference values.

	Tonnes $year^{-1}$	kg cap^{-1} $year^{-1}$	Reference	Conventional system	Source separation system
Release of nitrogen from WWTP's in to both fresh and marine water	15 742	1.75[1]	Statistics Sweden (2016b)	28%	25%
Release of nitrogen from WWTP's in to marine water only	8 077	0.897[1]	Statistics Sweden (2016b)	55%	49%

1) Assuming a population of 9.0 M in Sweden.

3.4. Return of nutrients to farmland

Source separation systems have a potential for increased recovery of nutrients from wastewater (Kjerstadius et al., 2015). Indeed, this potential has been the driving force for implementation of source separation systems in a pilot area of the city of Helsingborg, Sweden (City of Helsingborg, 2011). The recovered nutrients fractions from source separation systems contain, relatively conventional sludge from wastewater treatment plants, less heavy metals which likely makes them more attractive for return to farmland to be used as fertilizer (Kjerstadius et al., 2015). In the present study nutrient recovery in the source separation system is performed through ammonia stripping (for nitrogen recovery) and struvite precipitation (for phosphorus recovery) as well as through return of excess sludge from the anaerobic digester. The usage of struvite precipitation on wastewater is reported from several sources and can be assumed relatively well tested compared to ammonia stripping on wastewater which has been reported more scarcely (Le Corre et al., 2009; Maurer et al., 2003). For the conventional system nutrients are recovered in the dewatered sludge fraction of anaerobically stabilized sludge. The amount of sludge returned to farmland was assumed to be 43%, which is the average for Southern Sweden (Statistics Sweden, 2016b). This amount is higher than the national average in Sweden of 25% sludge return to farmland (Statistics Sweden, 2016b; SEPA, 2013). A return of 43% was also assumed for the dewatered sludge of the source separation system. For food waste a return of digested food waste is present as a separate stream from the food waste treatment plant in the conventional system. A return of digested food waste as biofertilizer to farmland is commonly practiced in Southern Sweden due to a biofertilizer certification system (SWMA, 2016) and thus 100% of the digested food waste is assumed for the conventional system. For the source separation system food waste is treated together with blackwater and nutrients from food waste are thus returned at the local treatment plant in the same fashion as for blackwater. The final results for return of nutrients (phosphorus and nitrogen) to agricultural soil are shown in Figure

13 below. It should be noted that of the nutrients not all of them can be assumed plant-available and thus replace mineral fertilizer. Plant availability of nutrients depends on how strongly they are bound to the substrate and thus e.g. phosphorus from sludge which have been precipitated with precipitation chemical is bound to the chemical used and nitrogen in organic substrates have a lower plant-availability than mineral fertilizers where the nitrogen is in mineral form only. Plant-availability of phosphorus in sludge is recorded to be between 25-75% (Foley et al., 2010) where many studies use the upper values of 60-70% (Willén, not published; Hospido et al., 2005; Peter & Rowley, 2009). Plant-availability of nitrogen in organic fertilizers, compared to mineral fertilizers of 100%, range between 6-80% depending on substrate (Delin et al., 2012) with digestate having an availability of 70% and sludge 53%. This explained why the source separation system returned a larger amount of plant-available nutrients as a higher proportion of the nutrients returned to agricultural soil were in mineral form. Taking the final results in to account it can be concluded from Figure 13 that the source separation greatly increases the return of plant-available nutrients to farmland compared to the conventional system.

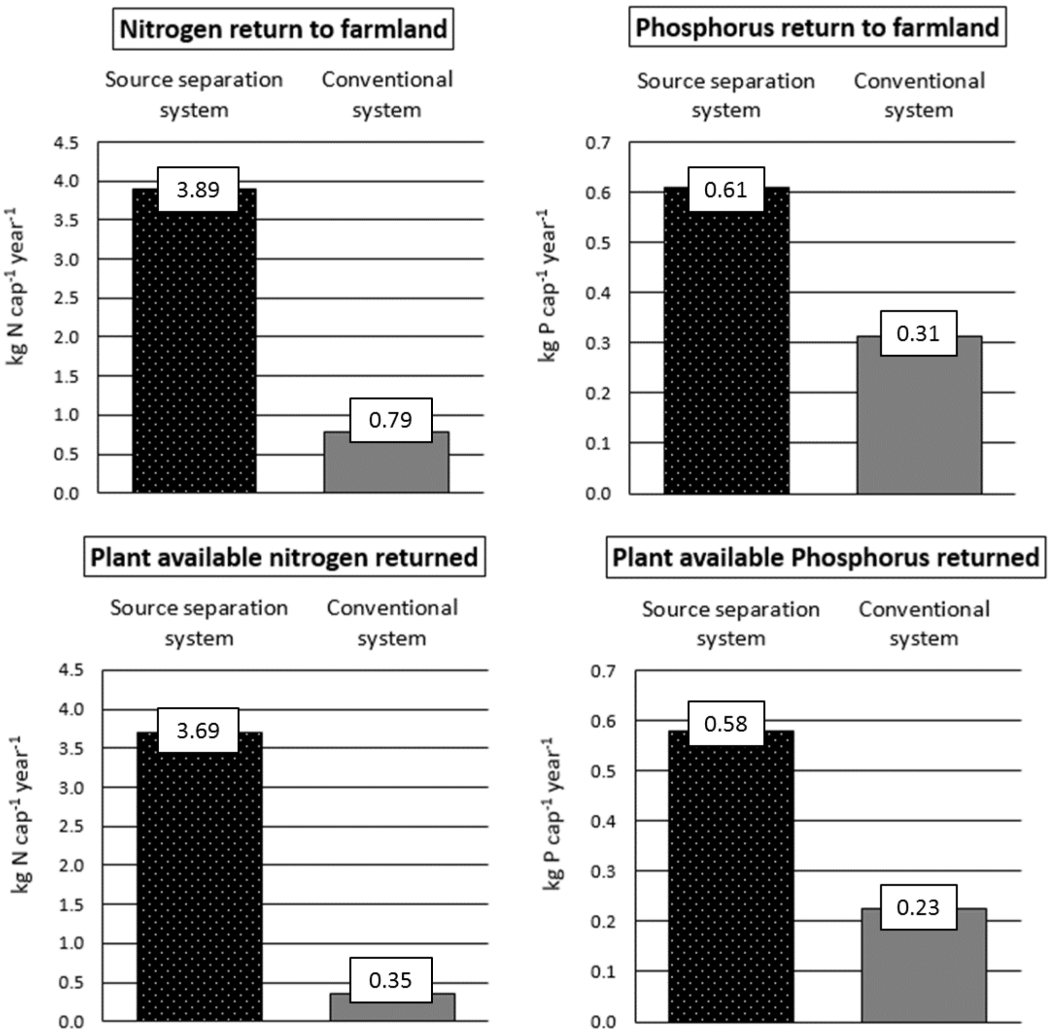


Figure 13 – Return of the nutrients nitrogen and phosphorus to farmland for the source separation system and the conventional system. A sludge return of 43% was assumed for both systems while 100% of digested food waste (conventional system) and 100% of ammonium sulphate as well as struvite was assumed (source separation system).

3.4.1. Comparison to other studies

Table 8 presents reported values for potential of return of nutrients from selected studies similar to the present work. It is clear from the table that source separation systems are always stated as giving higher return of nitrogen and phosphorus than conventional systems. For nitrogen the reported values ranges from 0.11-0.40 kg N cap^{-1} $year^{-1}$ for the conventional system and 3.09->4.29 kg N cap^{-1} $year^{-1}$ for the source separation system. It is clear that the results from the present study are in the upper range for the source separation system while being far higher than reported for the conventional system. The higher value for the conventional system in the present study is mainly a result of an assumed higher sludge return (43%) than for the studies on German conditions (Meinzinger, 2010; Remy, 201; Hillenbrand, 2009). Still, it is clear that the potential for return of nitrogen in the present study is much higher than any reported values in similar studies. For phosphorus the reported values ranges from 0.03-0.49 kg P cap^{-1} $year^{-1}$ for the conventional system and 0.44-0.72 kg P cap^{-1} $year^{-1}$ for the source separation system. The results from the present study are in the mid-range of reported values. Furthermore, when comparing the results from the studies in Table 8 it becomes apparent that source separation systems have always been stated to give a higher return of both nitrogen and phosphorus. The increased potential corresponds to a range between 1.73-4.29 kg N cap^{-1} $year^{-1}$ and 0.15-0.54 kg P cap^{-1} $year^{-1}$ in the cited studies, the range covering the calculated increases in the present study (3.1 kg N cap^{-1} $year^{-1}$ and 0.30 kg P cap^{-1} $year^{-1}$) in the present study. Thus, in conclusion the calculated values for nutrient return to farmland in the present study seem to be fairly reasonable in comparison to similar studies. Additionally, it can be concluded that source separation systems was always found to have a higher potential for nutrient return than the studied conventional systems.

Table 8. Comparison of results for return of nitrogen and phosphorus to farmland compared to results from similar studies. Results given in units of per capita and year.

	Remy (2010)	Hillenbrand (2009)	Meinzinger (2010)	Witeveen Bos (2014)	Thibodeau (2014)	Present study	Unit
				Nitrogen to farmland			
Conv. system	0.40[1]	-	0.11[4]	-	0.39[5]	**0.79** (0.54-0.79)	kg N cap^{-1} $year^{-1}$
Source sep. system	3.24[2]	+4.29[3]	3.09	-	2.12[2,5]	**3.89** (3.82-3.89)	kg N cap^{-1} $year^{-1}$
				Phosphorus to farmland			
Conv. system	0.49	-	0.03[4]	-	0.54[5]	0.31	kg P cap^{-1} $year^{-1}$
Source sep. system	0.72	+0.54[3]	0.44	-	0.69[2,5]	0.61	kg P cap^{-1} $year^{-1}$

1) Assumed 100% sludge to farmland. 2) Return of entire treated wet fraction. 3) Results only given as excess return with source separation system for nutrients with mineral fertilizer plant availability. 4) No nutrients are returned from the WWTP, only from food waste management. 5) value is for plant available nutrients after emissions and run-off.

3.4.2. Normalizing return of nutrients to farmland

Normalization of the values for retuned nutrients to farmland can be done against different references, as presented in Table 9 and Table 10 respectively.

In Table 9, the results are normalized against present national return of nutrients from wastewater treatment plants as well as suggested legal demands of recovery from wastewater treatment plants. It should be noted that no legal demands or official statistic on return of nutrients from food waste exist in Sweden. The results for nutrient return are compared first to the Swedish national average of nutrient return from wastewater treatment plants via sludge (Statistics Sweden, 2016b). These results show that both the conventional and the source separation system have a potential to increase the return of nutrients compared to the national average of Sweden. The potential for increase by the conventional system (210% for phosphorus and 309% for nitrogen) can partly be explained by the higher sludge return to farmland in Southern Sweden (43%) compared to the national average of 25% (Statistics Sweden, 2016b); and a smaller contribution (0.02 kg P capita^{-1} year^{-1} and 0.19kg N capita^{-1} year^{-1}) is explained by the included return of nutrients from food waste digestate. However, taking these two issues in to account the calculated return of nutrients would still be larger than from an average treatment plant in Sweden (13% larger per capita for phosphorus and 35% larger per capita for nitrogen) which show that the conventional system assumed for the study is in fact better than the national average in terms of phosphorus and nitrogen which is fractionated in to sludge. This should be kept in mind if comparing the conventional system in the present study to actual treatment plants in Sweden. The reason for the increased fractionation of nutrients in to sludge is explained by the exclusion of stormwater from the study, which causes a need for excess precipitation of nutrients at the conventional treatment plan in order to meet the discharge limits for nitrogen and phosphorus. The discharge limits in Sweden are usually expressed in units of mg/L, causing a need for increased precipitation if the wastewater is not diluted with stormwater. For the source separation system, it can be seen that the return of nutrients can be greatly increased (408% for phosphorus and 1 520% for nitrogen) compared to the present return of nutrients from wastewater treatment plants. The potential for increase is mainly explained by the extraction of nutrients as struvite and ammonium sulphate in the source separation system. These two fractions constitute the majority of the potential increase (340% of the P-increase and 1450% of the N-increase). Similar to the conventional system, the increased return of nutrients due to inclusion of food waste in to the wastewater only causes a minor contribution to the potential increase (0.04 kg P capita^{-1} year^{-1} and 0.23kg N capita^{-1} year^{-1}). In conclusion, although the conventional system in the present study is in fact more effective in sludge return than the national Swedish average the source separation system still has a potential to greatly increase nutrient return to farmland compared to the return of nutrients via sludge from wastewater treatment plants in Sweden today.

Comparing the results in Table 9 to suggested legal demands for return of nutrients from wastewater to farmland in Sweden (SEPA, 2013), it can be seen that both the conventional system as well as the source separation system can meet these suggestions. However, the conventional system is fairly close to the suggested targets (104% for P and 119% for N) and it should be kept in mind that return of nutrients by the conventional system in the present study is larger than the Swedish national average for treatment plants, due to the exclusion of stormwater from the present study as explained in the previous paragraph. In addition, the assumed sludge return to farmland (43%) is higher than the national average (25%) which makes the conventional system in the present study a better functioning system than it would be in many other regions in Sweden. As discussed, further in section 4.3, if the national average would have been used in the present study the suggested national goals would not have been reached. On the other hand, it can be seen from Table 9 that the source separation system meets the suggested demands with ease (203% for P and 734% for N). Thus, the addition of even a smaller catchment of a city with source separation system could greatly increase the average nutrient return of the city, thus boosting the return of nutrients from the city to meet the suggested legal demands. In conclusion, the conventional system barley meets the suggested legal demands for nutrient recovery while the source separation system greatly exceeds the suggested demands.

Table 9 – Normalization of amount of nutrients returned to farmland against selected literature values. Results for the conventional system and the source separation system are normalized as percentage of the reference values.

	Tonnes year^{-1}	**kg cap^{-1} year^{-1}**	**Reference**	**Conventional system [kg cap^{-1} year^{-1}]**	**%**	**Source separation system [kg cap^{-1} year^{-1}]**	**%**
Return of nutrients from WWTPs to farmland via sludge							
Phosphorus	1343	0.149[1]	Statistics Sweden (2016b)	0.313	210%	0.609	408%
Nitrogen	2300	0.256[1]	Statistics Sweden (2016b)	0.792	309%	3.89	1 520%
Suggested legal demands of return of nutrients from wastewater to farmland							
Phosphorus	40% of inflow to WWTP	0.28[2] / 0.30[3]	SEPA (2013)	0.291	104%	0.609	203%
Nitrogen	10% of inflow to WWTP	0.50[2] / 0.53[3]	SEPA (2013)	0.597	119%	3.89	734%

1) Calculated assuming a population of 9.0 million in Sweden. 2) Legal demands calculated for the conventional system, excluding food waste treated separately at food waste plant. 3) Legal demands calculated for the source separation system, including food waste since treated at the wastewater treatment plant

It is also of interest to compare the results for nutrient recovery to the current usage of mineral fertilizer in Sweden (Statistics Sweden, 2016a), as well as suggested planetary boundaries for use of mineral fertilizer (Nykvist et al., 2013); both of which are presented in Table 10. In relation to the amount of mineral fertilizer currently used in Sweden, it can be seen that the potential for return of nutrients for the conventional system is 4% for nitrogen and 22-31% for phosphorus. It should be kept in mind that the amount of nutrients returned by the conventional system in the present study is actually larger than what can be expected due to the exclusion of stormwater from the mass balance boundary, as explained earlier. Thus, even the small potential for the conventional system in comparison to the use of mineral fertilizer is most likely an over-estimate of a real life system. For the source separation system, the potential for nutrient return constitute a larger (18-22% for nitrogen and 44-61% for phosphorus) amount compared to the amount of imported mineral fertilizer. Still, neither system has the potential to completely replace mineral fertilizer usage in Sweden. This is most likely due to losses of nutrients through the food chain before reaching human consumption and subsequent release to the wastewater treatment plants (Matassa et al., 2015; SEPA, 2013). In conclusion, neither system can replace the need for mineral fertilizer, although the source separation system can greatly decrease the need of mineral fertilizer compared to the conventional system of today.

Human use of mineral fertilizer, through mining of phosphate rock or fixation of atmospheric nitrogen by the Haber-Bosch process, affects the global planetary cycles of phosphorus and nitrogen (Steffet et al., 2015; SEPA, 2013). It has been suggested that this anthropogenic interference in the nutrient cycles causes a threat to humanity's well-being and that there exist planetary boundaries for the limit of interference of the global cycles of nutrients (Steffen et al., 2015; Nykvist et al., 2013). These suggested boundaries for phosphorus and nitrogen are currently greatly exceeded (Steffen et al, 2015; Nykvist et al., 2013). Thus, replacing mineral fertilizer with recovered nutrients from wastewater will decrease the impact on these cycles. In Table 10, the suggested planetary boundaries for usage of mineral fertilizer are compared to present use of mineral fertilizer, as well as the potentials for nutrient return from the conventional and source separation systems. It is clear from the table that the planetary boundaries for mineral nitrogen use (1.57 kg P capita^{-1} year^{-1} and 6.89 kg N capita^{-1} year^{-1} in Sweden) are well below the current national use in Sweden (17.8-21.1 kg N capita^{-1} year^{-1}). Furthermore, it is clear that potential for return of nutrients from wastewater constitutes a substantial fraction of the planetary boundaries for the conventional system (11% of the N-boundary and 20% of the P-boundary), and substantially more in the case of the source separation system (56% of the N-boundary and 39% of the P-boundary). In Figure 14, it is clearly shown that the present use of P-mineral fertilizer is just below the suggested planetary boundary, and that this could be decreased further by the use of the suggested source separation system. For nitrogen, the present use of mineral fertilizer is much higher than the suggested planetary boundary, however this could to some extent be levitated by the use of source separation systems. In conclusion, the potential for nutrient recovery of phosphorus constitute a large percentage of present usage of mineral fertilizer while the potential for recovery of nitrogen only constitute a smaller part of present use of mineral fertilizer; albeit being possible to increase substantially by the use of source separation systems.

Table 10 – Normalization of amount of nutrients returned to farmland against import of mineral fertilizer and suggested planetary boundaries. Results for the conventional system and the source separation system are normalized as percentage of the reference values.

	Tonnes $year^{-1}$	kg cap^{-1} $year^{-1}$	Reference	Conventional system [kg cap^{-1} $year^{-1}$]	%	Source separation system [kg cap^{-1} $year^{-1}$]	%
Import of mineral fertilizer							
Phosphorus	9 400	1.0[1]	SEPA (2013)	0.313	31%	0.609	61%
Phosphorus	12 500	1.4[1]	Statistics Sweden (2016a)	0.313	22%	0.609	44%
Nitrogen	160 000	17.8[1]	SEPA (2013)	0.792	4%	3.89	22%
Nitrogen	190 200	21.1[1]	Statistics Sweden (2016a)	0.792	4%	3.89	18%
Planetary boundaries (Sweden)							
P-mineral fertilizer use	14 143	1.57[1]	Nykvist et al. (2013)	0.313	20%	0.609	39%
Nitrogen fixation - updated (N-mineral fertilizer use)	62 000	6.89[1]	Steffen et al. (2015)	0.792	11%	3.89	56%

1) Calculated assuming a population of 9.0 million in Sweden.

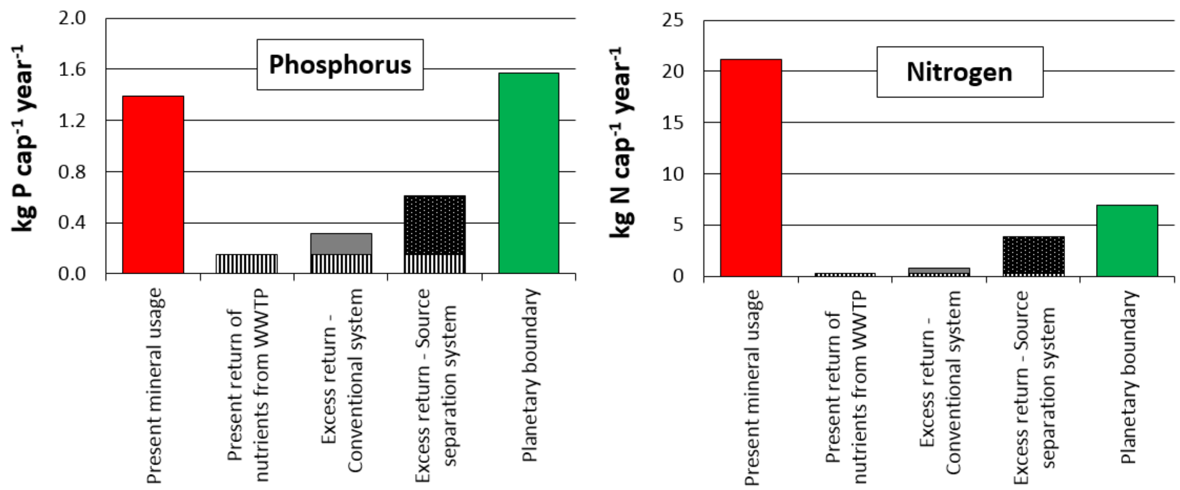


Figure 14 – Graphical comparison between present usage of mineral fertilizer, the potential for recovery and the suggested planetary boundaries.

3.5. Environmental impacts from infrastructure

It is debated whether or not to include infrastructure in LCA of waste management systems. According to Brogaard and Christensen (2015), capital goods should not be excluded from waste LCAs. The present study had the aim to include infrastructure extensively. The impact from infrastructure on the gross contribution (i.e. before subtracting negative emissions from vehicle fuel substitution with biogas usage etc.) to the investigated impact categories is shown in Figure 8 below. As shown in the figure, the contribution of infrastructure is equal to or less than 15 % for all impact categories. This is interesting since one of the major differences between the systems lies in the usage of infrastructure, which was the reason for including environmental impacts from infrastructure in the study. The main differences in infrastructure is summarized in Table 11 and it was hypothesized previous to the study that the concrete used for large sedimentation basins at the wastewater treatment plant for the conventional system would cause a noticeable difference in impact between the systems. However, as apparent from Figure 15, the infrastructure has only a smaller impact for both systems, and the source separation system has a slightly larger impact for climate change and freshwater eutrophication. This increased impact is in fact due to the increased amount of metal (mainly steel) used for the wastewater treatment plant in the source separation system. The increased amount of metal is due to the assumption of the wastewater treatment plant in that system being covered, since data for the material use is based on a pilot area in the Netherlands, which utilizes a covered treatment plant (Witveen Bos, 2014). Overall, the infrastructure causes only a smaller, and very similar, impact for either system. This conclusion is strengthened by similar results by Hillenbrand (2009), who found that infrastructure only constituted 7% of the climate change impact when comparing systems similar to the ones compared in the present study. Thus, if making a simplified rough estimate of similar systems one could likely disregard the impact of infrastructure without affecting the relative outcome between the systems.

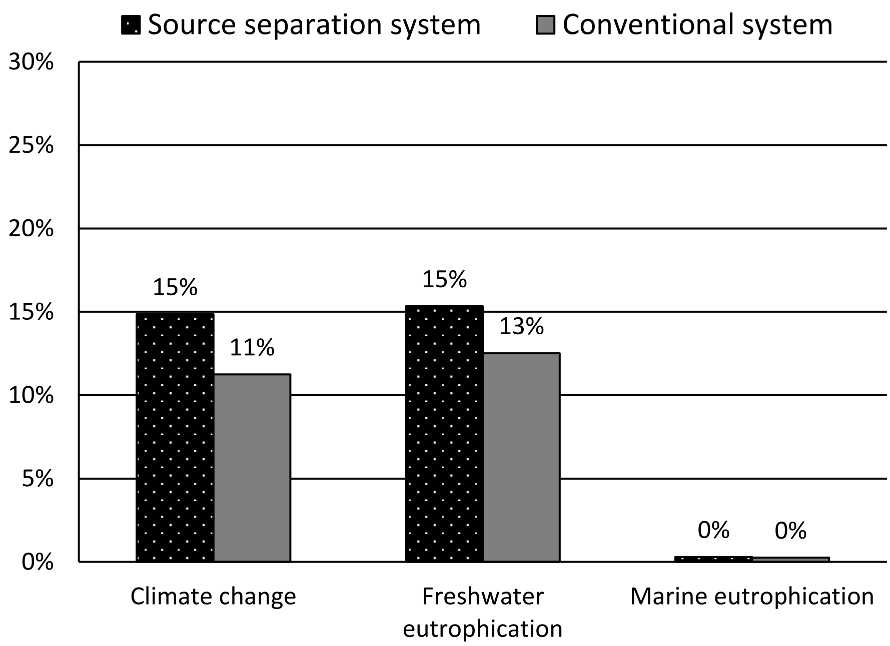


Figure 15. Contribution of infrastructure to the gross contribution to respective environmental impact.

Table 11. Description of selected infrastructure between the systems, in selection on the basis of highlighting differences between the systems.

Conventional system	Source separation system
Large concrete sedimentation basins at wastewater treatment plant	Small concrete sedimentation basin for greywater treatment only.
Less metal in wastewater treatment plant (under open sky)	More metal in wastewater treatment plant (covered)
Single sewer pipe and low-pressure pump system	Triplicate sewer system and pumps (for blackwater, greywater and foodwaste)
Food waste treatment plant, including pretreatment, in both cases largely using steel	No separate food waste plant (food waste handled at the wastewater treatment plant)
Large concrete foundation for storage of sludge and concrete basins for biofertilizer	Smaller concrete foundation for sludge storage due to less wet weight of sludge and biofertilizer

3.6. Usage of electricity and heat

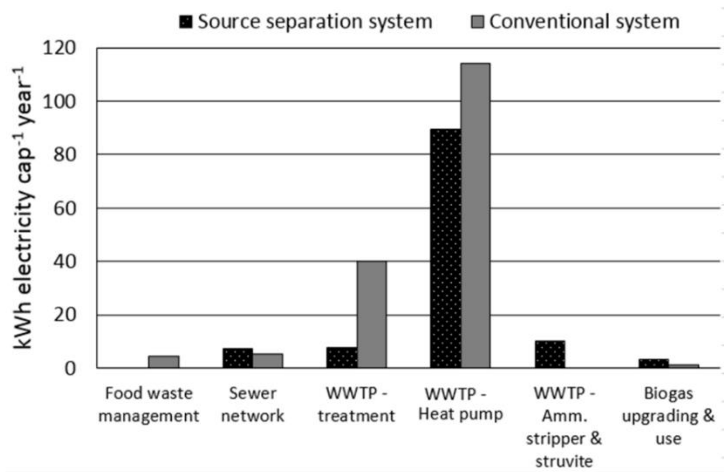
The study considers energy in many forms; from biogas replacing diesel as fossil fuel, to chemical usage in the wastewater treatment plant. Due to the aim of the study, no energy balance in any form is included. However, the direct electricity and heat usage of the systems are presented separately in Figure 16 and Table 12 in order to highlight differences between the studied systems. Note that the energy use here only includes direct energy use within the system boundaries. As such, energy required for the production of indirect products (such as chemicals, fertilizers or transports) is not included.

Figure 16 shows that mainly processes located at the wastewater treatment plants constitute the majority of the heat and electricity used as well as produced in both systems studied. In particular the heat pumps stands out for a large consumption of electricity and deliverer of heat. This effect is highlighted further in Table 12, where the impact of the heat pump is listed separately against all other processes in the studied systems. It is clear that the heat pump consumes between roughly 70% (conventional system) to 75% (source separation system) of the used electricity. Furthermore the heat pump delivers far more heat than needed by other processes (a factor over 8 for the conventional system and 3 for the source separation system). In the study, surplus heat is assumed to replace Swedish average district heating, to a large extent based on biofuels. It is clear from the table that the impact from all other processes in the studied systems have only a minor impact on electricity use (29-51 kWh) or heat consumption (52-139 kWh). The high impact of the heat pump explained by the amount of stored heat extracted from the wastewater streams in each system, mainly origin from the greywater fraction (Larsen, 2015). This high potential is not surprising, considering that Larsen (2015) clearly showed that the largest energetic potential in household wastewater was found as heat in the greywater fraction. The results for the potential for heat extraction in source separation system also agree well with a similar study by (Lindeboom, 2014) over a pilot area in Netherlands, were a potential heat pump was calculated to have a potential for 477 kWh thermal energy capita^{-1} year^{-1}. However, this is lower than the 800 kWh capita^{-1} year^{-1} potential stated by Larsen (2015) and the 840 kWh capita^{-1} year^{-1} potential for heat recovery that was calculated in sensitivity analysis SA_3 (Heat pump) in the present study. Thus, it can be said that although some controversy exist in regards to the quantity of heat recovery from source separated greywater, the potential is at least very good compared to the heat demand of the systems.

In regards to need for heat, the anaerobic digestion (WWTP-Treatment) requires heat for heating of the anaerobic digestion process. The present study assumed a mesophilic 37 °C anaerobic digestion for the conventional system and a 25 °C digestion in an upflow digester for the source separation system. Still, the source separation system has a higher need for thermal energy due to the more dilute liquid being anaerobically digested. The calculated heat demand for anaerobic digestion for the source separation system (86 kWh capita^{-1} year^{-1}) is higher than calculated by Lindeboom (2014) who stated that 50 kWh capita^{-1} year^{-1} would be sufficient for anaerobic digestion at 25 °C as assumed in the present study (section 2.2.4) for the source separation system. If a 37 °C anaerobic digestion of the dilute source separated blackwater and food waste would have been performed the heat requirement would have instead increased to 148 kWh capita^{-1} year^{-1}. Thus, a low temperature anaerobic process of 25 °C is required in order to minimize heat use for the source separation system.

Additionally, the extraction of nutrients (WWTP-Amm.stripper & struvite) in the source separation system demands considerable electricity and heat. Remaining processes related to sewer pumps, food waste management and biogas upgrading demanded relatively little energy.

In conclusion, the heat pump is the main consumer of electricity for either system as well as delivers much more heat than needed by either system. Furthermore, the source separation system uses less electricity but more heat compared to the conventional system.



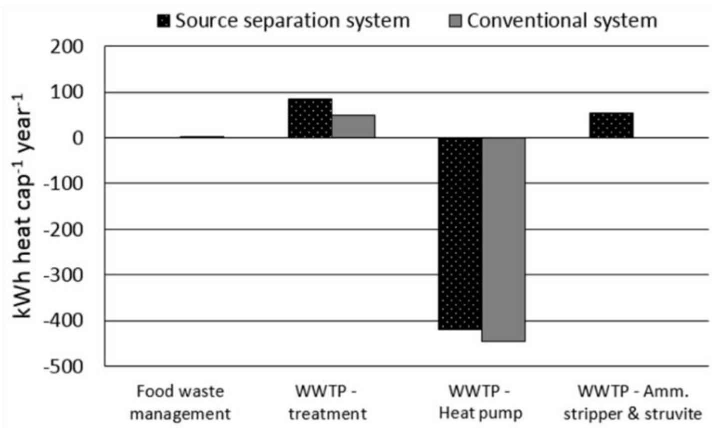


Figure 16. Detailed results for all impact categories. Results given per functional unit (FU).

Table 12. Net direct heat and electricity use of the systems per functional unit.

	Conventional system	**Source separation system**
Electricity demand [kWh]		
- Net value	**165**	**119**
- Heat pump	114	90
- Other	51	29
Heat demand [kWh]		
- Net value	**-393**	**-281**
- Heat pump	-445	-420
- Other	52	139

4. Sensitivity analysis

4.1. Sensitivity analysis 1 – Heat pump with stormwater inclusion

As apparent from the results in Figure 10-12 the heat pump constitutes a large fraction of the negative impact for climate change and considerable negative impacts for freshwater and marine eutrophication. Additionally, heat pumps constitute a large fraction of the electricity use and heat production for both systems as evident from Table 12. Heat pumps for extraction of heat from the treated wastewater also already exist at larger wastewater treatment plants in Sweden and it has been shown that they can constitute a large effect on the carbon foot print of the treatment plants (Gustavsson & Tumlin, 2013). Therefore, heat pumps were included in the present study for both the conventional and the source separation system. However, it should be made clear that the present study only considers household wastewater which makes calculations on a heat pump different from the calculation of most real-life treatment plants in Sweden that treats combined wastewater, in which stormwater is included (Molander, 2015). The addition of stormwater to wastewater constitute a large portion of the total wastewater received at treatment plants (Molander, 2015) and it thus increases the volume and decreases temperature of the wastewater that flows through the heat pump. Inclusion of stormwater would likely have decreased the benefit of a heat pump for the conventional system due to decreased temperature of the wastewater, which lowers the coefficient of performance (COP) of the heat pump (Baaring, 2015; Cipolla & Maglionico, 2014) and thus the efficiency of the heat extraction. Due to the high impact of the heat pump, in combination with the qualitative noticed difference between real life systems (including stormwater in the heat pump at wastewater treatment plants) it was decided to perform a sensitivity analysis on the efficiency of the heat pump.

4.1.1. Coefficient of performance of the heat pump

The COP of the heat pump is affected by the temperature of the wastewater from which heat is extracted as well as the temperature of the medium to which heat is transferred. Wastewater treatment plants that treat combined wastewater have been reported to have a COP between 2.9-3.3 (Baaring, 2015) or 3.2-3.5 (Cipolla & Maglionico, 2014). This should be compared to the higher COP of 3.9 used for household wastewater (GW+BW) for the conventional system in the present study. The higher COP used in the present study is explained by the exclusion of stormwater as well as a short transport distance in the sewer net, both decreasing temperature losses (Hellborg Lapajne, 2016). The source separation system has been calculated to have a COP of 4.7 due to the increased efficiency when heat exchanging from the separated hotter greywater (Hellborg Lapajne, 2016). This value also corresponds with the results of Lindeboom (2014) who stated a potential COP between 4.5 and 5.0 for a source separation system, based on a pilot area in Netherlands. Gustavsson & Tumlin (2013) who studied Swedish WWTP's stated that a COP above 4.0 was needed in order to achieve a reduction in climate change impact. As the environmental impacts from European electricity mix, the assumed electricity input in Gustavsson & Tumlin (2013) is considerably higher than in the present study (where Swedish electricity mixed is used), a lower COP could be beneficial from a GHG-perspective in the present study. Overall, it should be noted that the heat pumps have a considerable impact on the climate impact of the studied systems due to the amount of district heating they can replace.

For clarification, it has been assumed in the present study that the heat pump delivers district heating at 50 °C.

4.1.2. Potential for heat extraction using heat pumps

The amount of heat available for extraction from household wastewater is large compared to the amount of energy available for biogas production or replacing nutrients in mineral fertilizers (Larsen, 2015). A summary of the reported energy available for extraction is presented in Table 13 along with the values used in the present study. It should be noted that the values used in the present study (331 kWh cap^{-1} year^{-1}) are smaller than reported in the literature (477-876 kWh cap^{-1} year^{-1}). This represents the difference between the total potential for extraction in the present study and what be expected to be extracted in real life situations. In Figure 17, real-life data of the heat pump at Öresundsverket WWTP is presented on a monthly basis. The figure shows that a majority of the heat (>95 %) is extracted in the colder 6 months of the year, the reason most likely being a decreased need of district heating in the summer months. Lindeboom (2014), who stated decreased need for household heating as the reason, also saw decreased need for heat in summer months. Furthermore, Lindeboom noticed that in the winter months the heat recovered from greywater could cover only 20% of the heat demand for households while in the summer months the potential for heat extraction from source separated wastewater was just enough to cover the entire household needs. This contrasts the present study, in which it was assumed that heat recovery from wastewater only occurs 6 months per year (November-April), based on statistics from Öresundsverket. For comparative reasons, values for calculated heat extraction from combined wastewater (including stormwater) has been included at the end of Table 13. It can be seen that although the total heat recovery increases with combined wastewater (Baaring, 2015), the fraction stemming from household wastewater (GW+BW) is much lower than what is assumed in the present study.

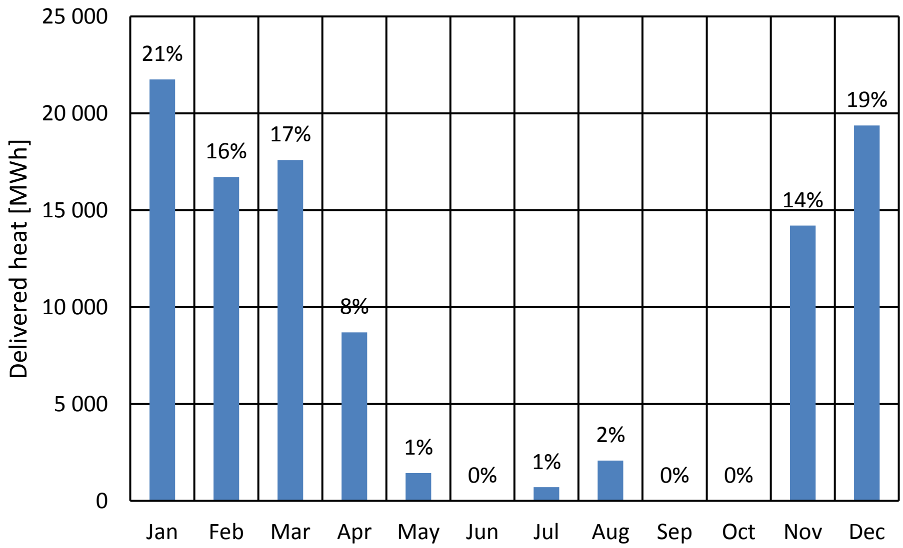


Figure 17 – Heat delivered from the heat pump at Öresundsverket WWTP for the year 2013. Over 95% of the delivered heat was extracted from November-April.

Table 13 – Reported values for possible heat extraction from wastewater using heat pumps.

Reference	Wastewater	Flow [L cap^{-1} d^{-1}]	Heat for extraction [kWh cap^{-1} year^{-1}]	COP	Comment
Larsen (2015)	GW	65.5[1]	800[2]	n/a	Calculated thermal energy from temperature increase from 10 °C to 38 °C.
Wallin & Claesson (2014)	Household wastewater (GW+BW)	No data	810	No data	Measured for building with 110 apartments.
Nykvist (2012)	GW	No data	876	4	Calculated.
Lindeboom (2014)	GW	60	477	4.5-5.0	Calculated based on pilot area with source separation system.
Present study – Source separation system	GW	130	331	4.7	Assumed extraction 6 months per year.
Present study – Conventional system	GW+BW	144	331	3.9	Assumed extraction 6 months per year.
Baaring (2015)	Combined wastewater	550	714	2.9-3.3	Real data for Öresundsverket WWTP.
Present study	GW + BW in combined wastewater	144	102	2.9-3.3	Extraction 6 months per year.

1) Fraction of GW that is hot (38 °C). 2) Total potential for extraction thermal energy.

4.1.3. Change of input data for sensitivity analysis

In a sensitivity analysis, input data was changed in order to simulate a scenario in which heat extraction from the source separation system can be done all year round in order to cover domestic heat demands, based on Lindeboom (2014). In the conventional system, heat extraction at the central wastewater treatment plant is limited to winter months due to a lower COP and heat demand in this system. Additionally the greater potential for heat extraction in the conventional system due to the large flow of stormwater was included based on the real life data from Öresundsverket WWTP (Baaring, 2015). It was assumed that stormwater in the source separation system is not treated in any way that allows central heat extraction. This scenario thus represent an inclusion of stormwater for the heat pump in the conventional system but not for the source separation system. In order to make the comparison more realistic, the increased energy demand for the wastewater treatment plant when including stormwater was calculated using the method presented by Remy (2010). The indata used for the sensitivity analysis is presented in Table 14.

Table 14 – Indata used in the sensitivity analysis SA_1 (Heat pump) compared to the indata used in the original scenario.

Process	**Conventional system**		**Source separation system**		**Unit**
	Original scenario	**Sensitivity analysis SA_1**	**Original scenario**	**Sensitivity analysis SA_1**	
Heat pump electricity use	114	238	90	179	kWh cap^{-1} $year^{-1}$
Heat pump delivered heat	447	714	420	840	kWh cap^{-1} $year^{-1}$
Wastewater treatment plant electricity use	40.3	67.4	No change	No change	kWh cap^{-1} $year^{-1}$

4.1.4. Results of sensitivity analysis

The results of the sensitivity analysis SA_1 (Heat pump) on the impact categories are given in Table 15. The results show that the changes made in relation to heat pump assumption have a great effect on decreasing the climate impact in both systems. However, the effect on climate change impact is most dramatic for the source separation system, explained by extraction of heat from greywater all year round, suggested possible by Lindeboom (2014), compared to the assumed 6 months per year in the original scenario. Oppositely, the conventional system is also possible of a much increased potential for heat extraction due to the inclusion of a large flow of stormwater, however this also decreases the COP and increases the electricity use at the wastewater treatment. The increased electricity use decreases the benefits from the extra heat extraction. This is still the case although the study considers Swedish electricity mix, which has a low climate impact compared to European electricity mix (ecoinvent, 2013. A switch to European electricity mix would thus have increased the gain of source separation systems even further. Finally, the sensitivity analysis also has an impact on the freshwater and marine eutrophication due to substituted biofuel in the production of district heating.

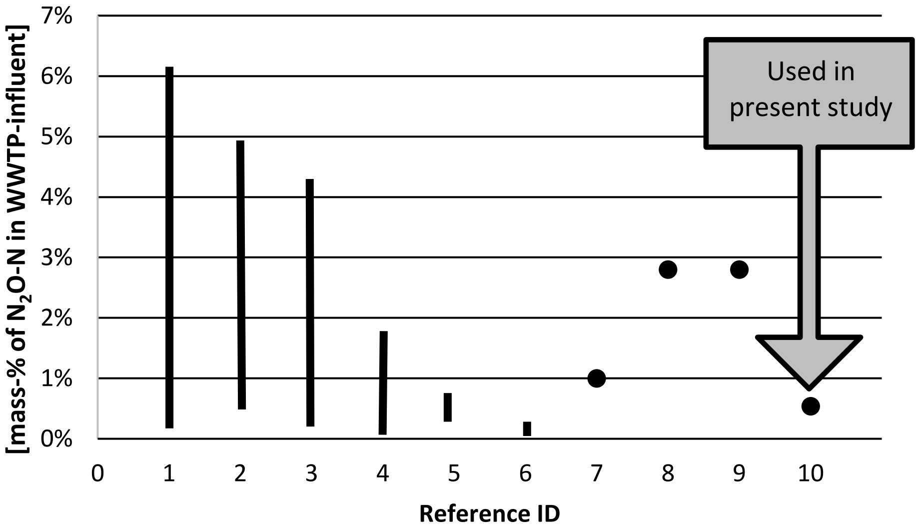
Table 15 – Results for the sensitivity analysis SA_1 (Heat pump) compared to the original scenario. Results given per functional unit (FU).

	Original scenario	**Sensitivity analysis SA_1 (Heat pump)**
	Climate change	
Conventional system	-12.6	-22.9
Source separation system	-37.1	-63.4
	Freshwater Eutrophication	
Conventional system	0.007	0.004
Source separation system	0.010	0.006
	Marine Eutrophication	
Conventional system	0.49	0.42
Source separation system	0.44	0.37
	Returned P	
Conventional system	No change	No change
Source separation system	No change	No change
	Returned N	
Conventional system	No change	No change
Source separation system	No change	No change

4.2. Sensitivity analysis 2 – Increased N_2O emissions from WWTP

Emission of nitrous oxide from wastewater treatment has been widely debated, with varying numbers being presented in literature (Yoshida et al., 2014; Desloover et al., 2012; Foley et al., 2010; Larsen, 2015). The variation could possibly be due to the fact that emissions of nitrous oxide from biological nitrogen removal (BNR) seems to be highly depended on process stability (GWRC, 2011; Yoshida et al., 2014). This is especially interesting since emissions of nitrous oxide has been reported as the main contributor to the carbon footprint of WWTP's (Desloover et al., 2012; Foley et al., 2010). The emissions used in the original scenario of the present study (0.01 kg N_2O-N kg N^{-1} denitrified) is in the lower range of the values reported in literature and thus constitutes a stable operation scenario.

Still, in the original scenario, emission of nitrous oxide from biological nutrient removal in the activated sludge at wastewater treatment plants constitute a large part (26% or 15 kg CO_2-eq. capita^{-1} year^{-1}) of gross GHG-emissions from the conventional system, and 4% (or 3 kg CO_2-eq. capita^{-1} year^{-1}) of gross GHG-emissions from the source separation system. These results supports the findings of Remy (2010), showing decreased emissions of nitrous oxide with a source separation system compared to a conventional system. Similarly, Hillenbrand (2009) found that emissions of nitrous oxides from BNR and mineral fertilizer production (30-40 kg CO_2-eq. capita^{-1} year^{-1}) constituted 13-17 % of the total impact on climate change caused by a conventional system, compared to minimal emissions from a source separation system. Both Remy (2010) and Hillenbrand (2009) stated that emissions of nitrous oxide is an important factor when assessing impacts on climate change from systems similar to the ones investigated in the present study. The difference in amount of emissions of N_2O between the conventional system and the source separation system is one of the main variances between the systems; the difference caused by more nitrogen being removed through activated sludge biological nitrogen removal (BNR) in the conventional system. However, the amount of reported emissions of nitrous oxide from BNR is varying largely in the literature and seems to depend on several different process parameters at the wastewater treatment plants (Daelman et al., 2015; GWRC, 2011; Desloover et al., 2012). The original scenario in the present study used an emission factor previously used for Swedish conventional WWTP´s (Gustavsson & Tumlin, 2013) based on Foley et al. (2010). In Figure 18, reported and assumed emissions of nitrous oxide from different activated sludge systems have been summarized and expressed as emissions percentage of influent nitrogen to the wastewater treatment plant. Thus, the figure covers different processes and climates, both of which factors than can affect the emissions of nitrous oxide (Larsen. 2015; Daelman et al., 2015; GWRC, 2011). The present study models a system in a colder (Swedish) climate, for which emissions have been shown both to increase (Daelman et al., 2013) and decrease (Guo and Vanrolleghem, 2014), giving inconclusive results in regards to N_2O emissions from BNR in colder climate (Larsen, 2015). Regardless, when comparing the emission factors in Figure 18 it becomes apparent that the emission factor used in the present study is in the lower region of reported values. Thus, the present study can be assumed to cover a very well-functioning biological nitrogen removal system and this may potentially underestimate the climate impact of BNR in activated sludge system. Therefore, a sensitivity analysis with increased N_2O emission was performed in order to investigate a possible impact of a less well-functioning BNR process in both systems. Such a sensitivity analysis will affect the emissions from the conventional system more due to the larger amount of nitrogen being removed in BNR compared to the source separation system in which a large fraction of nitrogen is removed by ammonia stripping as described in section 4.2.



References: 1) STOWA (2010). 2) Foley et al. (2008). 3) Yoshida et al. (2014). 4) Chandran et al. (2010). 5) Larsen (2015) calculated assuming 60% denitrification. 6) Yoshida et al. (2014) re-calculated guidelines from IPPC (2006). 7) VROM (2008). 8) Daelman et al. (2008). 9) Kosonen et al. (2013). 10) Present study, based on Gustavsson & Tumlin (2013) and Foley et al. (2010).

Figure 18. Reported emission factors of nitrous oxide (N_2O) in literature given as percentage of influent N to the wastewater treatment plant. Original values

4.2.1. Change of input data for sensitivity analysis

For the sensitivity analysis, input data was changed in order to simulate a scenario in which nitrogen removal in wastewater treatment is functioning less well than in the original scenario, thus causing increased nitrous oxide emissions. The choice of indata for the sensitivity analysis is based on Daelman et al. (2015) and Kosonen et al. (2013) both being results of long-term studies in similar climate (Rotterdam in Netherlands and Helsingfors in Finland respectively) as the present study. Both studies assumed emissions of nitrous oxide as 2.8% of nitrogen in influent to the wastewater treatment plant, as an average value over their measured time spans. The data used for the sensitivity analysis is presented in Table 16.

Table 16 – Indata used in the sensitivity analysis SA_2 (N_2O-emissions) compared to the data used in the original scenario.

	Conventional system		**Source separation system**		
Process	**Original scenario**	**Sensitivity analysis SA_2**	**Original scenario**	**Sensitivity analysis SA_2**	**Unit**
Emissions of N_2O in BNR	0.54%[1]	2.8%[2]	0.54%[1]	2.8%[2]	kg $N\text{-}N_2O$ kg N in influent^{-1}

1) Recalculated from Foley et al. (2010) stating 0.01 kg $N\text{-}N_2O$ per kg N denitrified. 2) Based on Daelman (2015) and Kosonen et al. (2013).

4.2.2. Results of sensitivity analysis

The results of the sensitivity analysis SA_2 (N_2O-emissions) on climate change (other impact categories remain unchanged) are given in Table 17. The results show that the sensitivity analysis greatly increases the climate impact of both systems. However, the increase is largest for the conventional system due to more nitrogen being removed by BNR in this system. The end result shows a clear benefit for the source separation system, which is less dependent on the nitrous oxide emission factor due to less biological nitrogen removal in this system. It should also be clarified that the increased emission of nitrogen in the form of N_2O is assumed to be replacing nitrogen in the form of N_2; both being removed in BNR. Therefore, the sensitivity analysis does not affect the amount of nitrogen being returned to farmland. Overall, the effect of a changed emission factor for N_2O emissions from BNR greatly affects the final results of the study, something that was also seen by Larsen (2015) who investigated the impact of different emissions factors on the overall climate impact of wastewater treatment. In conclusion, the choice of N_2O-emission factors should be given sever attention in studies over similar systems.

Table 17 – Results for the sensitivity analysis SA_2 (N_2O-emissions) compared to the original scenario. Results given per functional unit (FU).

	Original scenario	**Sensitivity analysis SA_2 (N_2O-emissions)**
	Climate change	
Conventional system	-12.6	48.5
Source separation system	-37.1	-9.2
	Freshwater Eutrophication	
Conventional system	No change	No change
Source separation system	No change	No change
	Marine Eutrophication	
Conventional system	No change	No change
Source separation system	No change	No change
	Returned P	
Conventional system	No change	No change
Source separation system	No change	No change
	Returned N	
Conventional system	No change	No change
Source separation system	No change	No change

4.3. Sensitivity analysis 3 – Decreased sludge return to farmland

In Sweden, sludge from wastewater treatment plants is to some extent returned to farmland to be used as fertilizer (SEPA, 2013). This practice has been heavily debated over the past decades since sludge, apart from beneficial nutrients and humic substances, contain heavy metals and micro-pollutants (Kjerstadius et al., 2013; SEPA, 2013; Bengtsson & Tillman, 2004). The past decades has also seen periods with shifting degree of sludge return as well as complete sludge bans by the Swedish farmers association for return to farmland (SEPA, 2002). The present study assumed a sludge return of 43% to farmland based on the average data for Southern Sweden (region of Scania) by Statistics Sweden (Statistics Sweden, 2016b). This amount is much higher than the Swedish national average of 25% (Statistics Sweden, 2016b; SEPA, 2013). Due to more nutrients being present in the sludge from the conventional system than for the source separation system (in which nutrients are mainly recovered through struvite precipitation and ammonia stripping) the original scenario in the present study represents a well-functioning conventional system compared to Swedish national averages values for sludge return.

It should be noted that food waste in the conventional system is handled separately in a food waste plant. All residues from this plant are assumed to be returned to farmland due to a certification system (SWMA, 2012) labelling residues from anaerobic digestion of food waste as biofertilizer and being attractive for return to farmland. At the current, 19 Swedish biogas plants have been certified according this system (SWMA, 2016).

Return of nutrients from food waste and wastewater is a long term goal in Sweden (SEPA, 2013). Currently, the Swedish Environmental protection agency is preparing a new legislation for sludge recovery and limits for heavy metals in sludge being spread on farmland (SEPA, 2013). In the working material, the SEPA estimates that the sludge return to farmland can increase by 20% in the following

15 years in Sweden, due to present work on sludge quality (SEPA, 2013). This increase, compared to today's return, would correspond to a national recovery of 30% of sludge to farmlands. Thus, even compared to this estimate, the assumed sludge return in the present study (43%) is large. In order to compare the effect of a sludge return on national average numbers (25%) a sensitivity analysis was performed. The sensitivity analysis, SA_3 (Sludge return), thus represents a scenario where sludge return is decreased from high regional values (43%) to Swedish national average values (25%).

4.3.1. Change of input data for sensitivity analysis

For the sensitivity analysis, input data was changed in order to simulate a scenario in sludge return from wastewater treatment plants is decreased to the national Swedish average. The choice of input data is based on statistics from Swedish Statistics (Statistics Sweden, 2016b) as well as the Swedish Environmental protection agency (SEPA, 2013). The input data used for the sensitivity analysis is presented in Table 18. It should be noted that the return of biofertilizer from food waste management in the conventional system is always considered to be 100%, since this fraction can be certified as biofertilizer as explained previously. Food waste in the source separation system is handled at the wastewater treatment plant and is thus included in the sensitivity analysis.

Table 18 – Indata used in the sensitivity analysis SA_3 (Sludge return) compared to the indata used in the original scenario.

Process	Conventional system		Source separation system		Unit
	Original scenario	Sensitivity analysis SA_3	Original scenario	Sensitivity analysis SA_3	
Return of sludge from wastewater treatment plants to farmland (replacing mineral fertilizer)	43%[1]	25%[2]	43%[1]	25%[2]	Mass-%
Return of biofertilizer from food waste plant to farmland	100%	100%	n.a.[3]	n.a.[3]	Mass-%

1) Based on regional values for Scania (southern Sweden) by Statistics Sweden (2016b). 2) Based on Swedish national average values (Statistics Sweden, 2016b; SEPA, 2013). 3) Food waste is treated at the wastewater treatment plant in the source separation system.

4.3.2. Results of sensitivity analysis

The results of the sensitivity analysis SA_3 (Sludge return) on the impact categories are given in Table 19. The results show a minor effect on climate change. This could be perceived as counter-intuitive as returned sludge replace mineral fertilizer and a decreased sludge return would thus imply less mineral fertilizer being replaced. Especially given that replaced mineral fertilizer were shown to give a large decreased climate impact in the original scenario (Figure 10). However, the assumed losses of of methane and nitrous oxide during sludge storage decrease the climate benefit of returning sludge to farmland. Additionally, sludge which is not being returned to farmland is assumed processed in to constructed soil by composting; a process assumed to have less emissions of nitrous oxide than sludge storage. Thus, a decrease in sludge return to farmland causes less emissions from sludge storage and relatively little emissions from composting. This decreased emissions is roughly equal to the lost benefit of replacing mineral fertilizer during sludge return to farmland. The overall effect is a minor change to the overall impact on climate change. It is important to realize that the emission factors from sludge storage is an important factor here and that sludge spreading techniques will effect these emissions (Willén, unpublished). Contrarily to nutrients in the sludge fraction, mineral fertilizer or nutrients recovered as struvite or ammonium sulphate are assumed not to have any emissions during storage which of course decreases impact on climate change. In summary, a change of sludge return to farmland from the 43% in souther Sweden down to the national Swedish average of 25% does only have a minor impact on climate change, a fact mainly contributed to the emissions factors used for sludge storage in the present study (Appendix A2).

For the impact categories freshwater and marine eutrophication, the sensitivity analysis with decreased sludge return does barely affect the final impact at all. For marine eutrophication these results are not surprising, since a majority of the impact comes from nitrogen being discharged in to the ocean as effluent from the wastewater treatment plant (Figure 12). However, a small fraction of the impact on marine eutrophication is caused by sludge and nutrient management and for freshwater eutrophication, sludge and nutrient management constitute a slightly larger impact (Figure 11). Thus some change in impact on freshwater eutrophication due to decreased sludge return could be expected. And correctly, as a decrease in amount of sludge going to farm land implies less leakage of phosphorus from soil, which is not included for sludge used as constructed soil, it is naturally that a slight decrease in freshwater eutrophication is seen in the results of the sensitivity analysis. Also a minor decrease in the impact on marine eutrophication is seen and this is due to less leakage of nitrogen from soil as for phosphorus on freshwater eutrophication. This is to some extent balanced out by increased emissions of ammonia emissions in the compost process but this effect is not as big as the impact of decreased leakage. That the results on the source separation scenario is not affected as much is due to the significantly lower amounts of nutrients, thus implying lower levels of leakage and ammonia emissions, returned to farm land with sludge in this scenario. Conclusively the results show that less sludge returned to farm land implies less leakage and thus less impact on eutrophication. This is to some extent balanced out by larger emissions of ammonia from the composting process

Finally, for the amount of actual returned nutrients to farmland (phosphorus and nitrogen) it can be seen that the decreased sludge return have a clear effect on reducing the amount of nutrients returned. The effect is most prominent for the conventional system, where the return is decreased by 0.12 kg P capita^{-1} year^{-1} and 0.25 kg N capita^{-1} year^{-1}. For the source separation system, were more nutrients are returned as struvite and ammonium sulphate, a decrease by 0.04 kg P capita^{-1} year^{-1} and 0.07 kg N capita^{-1} year^{-1} is seen. In conclusion, a decreased sludge return to farmland will have a more adverse effect on the conventional system than for the source separation system.

Table 19 – Results for the sensitivity analysis SA_3 (Sludge return) compared to the original scenario. Results given per functional unit (FU).

	Original scenario	**Sensitivity analysis SA_3 (Sludge return)**
	Climate change	
Conventional system	-12.6	-12.6
Source separation system	-37.1	-37.1
	Freshwater Eutrophication	
Conventional system	0.007	0.005
Source separation system	0.010	~0.010
	Marine Eutrophication	
Conventional system	0.49	0.48
Source separation system	0.44	~0.44
	Returned P	
Conventional system	0.31	0.19
Source separation system	0.61	0.57
	Returned N	
Conventional system	0.79	0.54
Source separation system	3.89	3.82

A decreased return of nutrients will also have an effect on the suggested national targets for nutrient recovery from wastewater (SEPA, 2013). These targets (recovery of 40% of P and 10% of N, as national averages) will be more difficult to reach if the sludge return decreases, as made evident in Figure 19, where the return of nutrients is presented together with lines marking the suggested targets. It is clear from Figure 19 that the source separation system still would meet the suggested targets, while the conventional system does not meet the goal for neither nitrogen nor phosphorus. Thus, targets on nutrient recovery, as suggested by SEPA (2013), may be difficult to reach in Sweden with conventional systems alone. However, looking at the vast potential for nutrient recovery via the source separation system, it is clear that smaller urban areas with source separation systems increases the possibilities to meet the targets suggested by SEPA (2013). The increased nutrient recovery through a smaller area with source separation systems will thus make the city meet the average recycling goals. The decreased need for mineral fertilizer with will also reduce human interference with the global nutrient cycles of nitrogen, being described as currently exceeded planetary boundaries (Steffen et al., 2015).

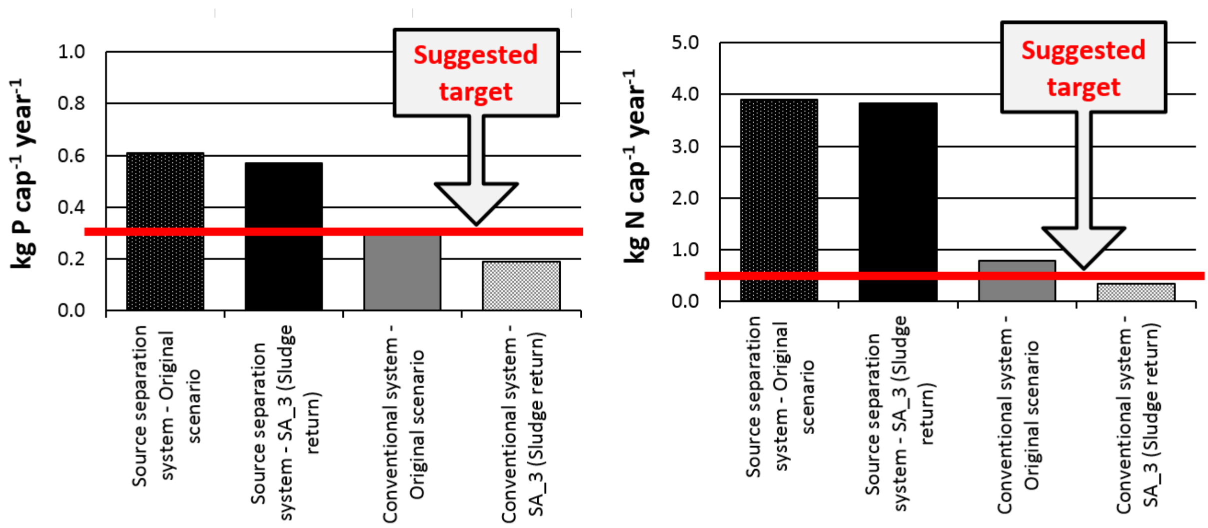


Figure 19 – Results for nutrient return to farmland for the original scenario and SA_3 (Sludge return) compared to targets for nutrient recovery from wastewater, suggested by SEPA (2013).

4.4. Summary of sensitivity analysis

The three performed sensitivity analysis dealt with a more realistic scenario for heat pumps (SA_1), increased N_2O emissions from wastewater treatment (SA_2) and a decreased sludge return to farmland (SA_3). Results are summarized together with the results for the original scenario in Table 20. It is clear that the performed sensitivity analyses mainly affected the impact on climate change, while giving only minor effect on the freshwater and marine eutrophication.

It is seen that climate change impacts from both systems are sensitive to assumptions regarding the use of heat pumps as well as emissions of N_2O from WWTP-processes. The conventional system has a higher sensitivity to assumed level of N_2O from WWTP-processes. Highly variable results were obtained through the sensitivity analyses. The conventional system varied between -12.4 and 48.5 kg CO_2-eq. cap^{-1} $year^{-1}$ while the source separation system varies between -63.4 and -9.2 kg CO_2-eq. cap^{-1} $year^{-1}$. This corresponds to a decrease of climate change impact for the source separation system between 21.2 and 55.9 kg CO_2-eq. cap^{-1} $year^{-1}$.

The effect on freshwater eutrophication was minor, the conventional system ranging between 0.034-0.037 kg P cap^{-1} $year^{-1}$ and the source separation system between 0.045-0.049 kg P cap^{-1} $year^{-1}$, which corresponded to a higher impact of 0.011-0.012 kg P cap^{-1} $year^{-1}$ for the source separation system.

For marine eutrophication the effect of the sensitivity analysis was again minor. The conventional system ranging between 0.50-0.57 kg N cap^{-1} $year^{-1}$ and the source separation system between 0.39-0.46 kg N cap^{-1} $year^{-1}$, which corresponded to a higher impact of 0.11 kg N cap^{-1} $year^{-1}$ for the conventional system.

Decreasing the fraction of sludge returned to farmland will have a larger effect on the conventional system, compared to the source separating where more nutrients are returned as struvite and ammonium sulphate. The increased nutrient return to farmland with the source separation system thus ranges between 0.30-0.38 kg P cap^{-1} $year^{-1}$ and 3.10-3.28 kg N cap^{-1} $year^{-1}$.

Table 20 – Summary of the results for the original scenario together with the performed sensitivity analyses. All results given per FU.

	Original scenario	**SA_1 (Heat pump)**	**SA_2 (N_2O emissions)**	**SA_3 (Sludge return)**	**Unit**
	Climate change				
Conventional system	-12.6	-22.9	48.5	-12.6	kg CO_2-eq. cap^{-1} year^{-1}
Source sep. system	-37.1	-63.4	-9.2	-37.1	kg CO_2-eq. cap^{-1} year^{-1}
	Freshwater Eutrophication				
Conventional system	0.007	0.004	No change	0.005	kg P cap^{-1} year^{-1}
Source sep. system	0.010	0.006	No change	~0.010	kg P cap^{-1} year^{-1}
	Marine Eutrophication				
Conventional system	0.49	0.42	No change	0.48	kg N cap^{-1} year^{-1}
Source sep. system	0.44	0.37	No change	~0.44	kg N cap^{-1} year^{-1}
	Returned P				
Conventional system	0.31	No change	No change	0.19	kg P cap^{-1} year^{-1}
Source sep. system	0.61	No change	No change	0.57	kg P cap^{-1} year^{-1}
	Returned N				
Conventional system	0.79	No change	No change	0.54	kg N cap^{-1} year^{-1}
Source sep. system	3.89	No change	No change	3.82	kg N cap^{-1} year^{-1}

4.5. Study uncertainty

4.5.1. System boundary

The study aimed to cover food waste and wastewater management all the way from household collection and sewer net to the wastewater treatment plant and sludge management. Thus, the management system was modeled in detail, aiming at including all processes involved. However, it can be seen from the detailed results in figures 10-12 that a few posts are responsible for the vast majority of the contribution to the impact factors studied. In fact, the wastewater treatment plant (WWTP), biogas usage, heat pump and sludge and nutrient management constituted a majority of the impact; the remaining posts (household installations, sewer network and food management) giving a very small contribution in comparison. Thus, it could be suggested that the posts of low impact could be excluded from future LCA work on similar systems.

An interesting issue with the system boundary is that consumption of potable water was excluded from the present study. In similar studies (Witeveen Bos, 2010; Meinzinger, 2010), the water consumption has been included, which makes the present study stand out in choice of system boundaries. This decision was based on the fact the usage of water was fairly similar between the systems, using input data from Jönsson et al. (2005) and Kjerstadius et al. (2012). The only differences being the vacuum toilet (replacing water closet for the conventional system) and food waste disposer (food waste being collected in bags in the conventional system) for the source separation system. The vacuum toilet is assumed to reduce toilet flushing water from 14 L capita^{-1} day^{-1} to 9 L capita^{-1} day^{-1} while the food waste disposer increases water consumption with 1.2 L capita^{-1} day^{-1} (Kjerstadius et al., 2012). These two differences was assumed to reduce the water consumption from 144 L capita^{-1} day^{-1} (conventional system) to 140 L capita^{-1} day^{-1} (source separation system), i.e. a reduction of 3%.

In sludge & nutrients handling the constructed soil was assumed to not replace any other soil, nor nutrients, and also to not cause any leakage. This is a simplified assumption and could depend with type of use though today this type of soil does not have any greater value.

4.5.2. Data collection

The mass balances for the systems has previously been published and compared to empirical data for pilot areas and the city of Helsingborg (Kjerstadius et al., 2015; Wiersma & Elzinga, 2014) and was thus assumedly representative for the systems studied. However, it should be noted that the wastewater treatment plant assumed for the conventional system in the present study has some characteristics that needs highlighting. Firstly, the use of chemicals is rather low since no external carbon source is needed in the activated sludge treatment. An external carbon source can cause a large contribution to the climate impact of a wastewater treatment plant (Gustavsson and Tumlin, 2013). Secondly, from the mass balances it is clear that 27% of the influent nitrogen ends up in the produced sludge rather than the 20% that is the case in the city of Helsingborg today (NSVA, 2014) as well as suggested in literature (Siegrist et al., 2008). The high amount of nitrogen in the sludge is partly due to the exclusion of stormwater in the present study. The exclusion of stormwater increased nutrient concentrations and thus post-precipitation was included in order to reach the discharge demands (10mg N/L and 0.5mgP/L); the post precipitation causing mainly P-removal but to a less extent also N-removal to sludge (Lindquist, 2003). However, keeping the stormwater exclusion in mind, the mass balance for the conventional wastewater treatment plant conformed well to an extensive study by Yoshida et al. (2015) and can thus be seen as a fair representation of a real life conventional wastewater treatment plant.

The infrastructure, electricity and heat demands for the 120 000 pe conventional wastewater treatment plant is based on a large assessment of German conventional wastewater treatment plants (Remy, 2010), which also conformed well with data on electricity demand of the treatment plant in the city of Helsingborg (NSVA, 2014). Thus, the used data appears to be a good approximation for a conventional system. The infrastructure, electricity and heat demands of the source separation wastewater treatment plants are based on an existing pilot area (Witteveenbos, 2014). This pilot area being fairly well studied (Wiersma & Elzinga, 2014; Lindeboom, 2014; Witeveen Bos, 2014, STOWA, 2014; Wiersma, 2013) and the values are likely correct. Together with the post-precipitation considered in the present study the source separation system is thus assumed to meet the same discharge demands as the conventional system (10mg N/L and 0.5mgP/L). Additionally, it is worth mentioning that since the studies on the pilot area with source separation system (Wiersma & Elzinga, 2014; Lindeboom, 2014; Witeveen Bos, 2014, STOWA, 2014; Wiersma, 2013) considers an implementation of a few thousand inhabitants, the present study (considering 12 000 people for the source separation system) may have over-estimated the impact for a large-scale source separation system, as beneficial scale-effect are likely to appear.

The time span of the study was set to 50 years, in order to cover the minimum life span of sewer network systems. Especially emissions related to the production of electricity and heat can be assumed to change greatly during this time period. It can be speculated that process operation at wastewater treatment plants will also be improved during this time, possibly reducing the emissions of nitrous oxide from wastewater treatment. However, such changes falls outside the scope of this study; being to produce an environmental impact comparison on systems available today.

Environmental impacts from provision of electricity were in the present study based on average Swedish data, available in ecoinvent v3.0 (ecoinvent, 2013). Swedish national environmental objectives clearly point at a need for reducing primarily GHG-emissions from energy production. Thus, it is likely that the current and future electricity provision will result in lower environmental impacts. Such a development would reduce overall climate change impacts from both systems, but particularly from the source separation system, which has a higher electricity input per FU. In the case of thermal energy provision, it was assumed that surplus energy substitutes Swedish average district heating data, based on Gode et al. (2011). Also in this case, reference data is aged (from 2008). Thus, it is likely that the present/future situation would generate in lower environmental impacts than the ones used in the present study. As both systems result in a surplus of thermal energy, such a development would, opposite to the case of electricity provision, increase overall environmental impacts from the systems.

Data for modelling of the collection and handling of food waste in both systems were to a large extent collected from full-scale studies in a Swedish municipality. Households' behavior could affect data on electricity use in disposers, number of paper bags used per kg food waste etc., while technological development could improve efficiency primarily in the mechanical pretreatment of food waste collected in the conventional system.

Chemicals used in the WWTP were in all cases gathered from the ecoinvent v3.0 database. Chosen datasets represent production of chemicals as global or European averages. However, due to a large need for electricity input per output sodium hydroxide, the environmental impact from the product is very sensitive to the type of electricity used in the process (Thannimaly et al. 2013). Producers of sodium hydroxide can be found in Sweden and thus, it was assumed that a Swedish product was used in the process. Consequently, the electricity input in the production process was adjusted to a Swedish electricity mix. Also in the case of sulphuric acid production, electricity input has been identified as having a key importance on overall environmental impacts (Kennecott, 2012). Also sulphuric acid is produced in Sweden, thus, also in this case, the electricity input in the ecoinvent process was adopted to reflect use of Swedish electricity.

Data on emissions from management and use of organic substrates on farm land is very hard to estimate based on literature. First of all not many studies have been conducted for a variation of organic substrates. Secondly, emissions depend on parameters as chemical and physical characteristics of the substrate, spreading technique, weather and soil conditions (Thibodeau, 2014). These two variables make it difficult to estimate a correct level of emissions when not having data from field studies of the exact area being estimated. Also as these emissions have a great impact on the results of these kinds of systems, e.g. nitrous oxide emissions being a strong greenhouse gas, an uncertainty of these data puts a great uncertainty on the final results of the study. The difficulty of determining these emissions and their impact is further discussed elsewhere (Spångberg, 2014; Thibodeau, 2014; Berglund et al., 2009; Johansson et al., 2008).

4.6. Final discussion

The present study investigated the impact of two sanitation systems (a source separation system and a conventional system) for a hypothetical urban area in Southern Sweden. The studied systems were assumed to have 120 000 inhabitants (conventional system) or 12 000 inhabitants (source separation system) respectively, thus being a comparison between a conventional system benefitting from large-scale infrastructure compared to a smaller urban area with source separation system. The rationale behind the difference in studied scale is that source separation systems in the foreseeable future, are more likely to be considered for implementation in smaller parts of cities, as a complement to existing wastewater and food waste handling infrastructure. However, the reader should be reminded that due to the functional unit being per capita and year, the results are fully comparable between the two studied systems, albeit the conventional system having the efficiency benefits that follows large-scale implementation.

The present study compared impact on climate change, freshwater eutrophication, marine eutrophication as well as potential for return of nutrients to farmland. To help comparison, all the net results are summarized in Table 21 together with relevant results from similar studies. As discussed previously, the results from the present study mainly agrees with the literature. The main deviance are seen in relation to climate change, where the present study showed the lowest results for both studied systems, likely being due to using Swedish electricity mix in the present study while the literature has used European electricity mix or country specific mixes, resulting in much higher climate impact per kWh. Results from the present study agree with most of the literature in relation to other impact categories. Keeping in mind that the works of Meinzinger (2010; Remy (2010) and Hillenbrand (2009) are dissertations, devoted to investigating the differences between source separation systems and conventional systems, this indicates a high reliability of the results gained in the present study.

Overall, results suggest a slightly decreased impact on climate change from the source separation system, in line with two out of three studies used for comparison in Table 21. In the disagreeing study, results are largely related to use of a large amount of electricity with high CO_2-impact in the source separation system. Thus, given that less CO_2-intensive electricity is available, it seems fairly reasonable to conclude that source separation systems have the potential to decrease GHG-emissions from wastewater treatment. However, when comparing the impact of either system to the suggested planetary boundaries for climate change for Sweden (2 000 kg CO_2-eq. capita^{-1} year^{-1}), the relative potential for decrease is small (roughly 1-2% of the boundary value). Thus, although source separation systems can decrease climate impact compared to conventional systems in Sweden today, the effect is small compared to other sources of anthropogenic impacts on climate change.

Table 21. Comparison to results from similar studies. Both results from the original scenario (in bold) and results obtained in sensitivity analyses (in brackets) are presented for the present study. A denotation of n.a. means not applicable.

	Remy (2010)	Hillenbrand (2009)	Meinzinger (2010)	Witeveen Bos (2014)	Thibodeau (2014)	Present study	Unit
Climate change							
Conv. system	140	244	-	32-40	52.8	**-13** (-23 to 48)	kg CO_2-eq. cap^{-1} year^{-1}
Source sep. system	85	315	-	-22	65.3	**-37** (-63 to -9)	kg CO_2-eq. cap^{-1} year^{-1}
Freshwater Eutrophication							
Conv. system	0.046[1]	0.17[2]	0.070[3]	0.023-0.026	n.a.	**0.007** (0.004-0.007)	kg P cap^{-1} year^{-1}
Source sep. system	0.023[1]	0.18[2]	0.056[3]	0.054	n.a.	**0.010** (0.006-0.010)	kg P cap^{-1} year^{-1}
Marine Eutrophication							
Conv. system	0.43[1]	1.24[2]	0.81[3]	0.036-0.040	n.a.	**0.49** (0.42-0.49)	kg N cap^{-1} year^{-1}
Source sep. system	0.12[1]	0.29[2]	0.48[3]	0.05	n.a.	**0.44** (0.37-0.44)	kg N cap^{-1} year^{-1}
Nitrogen to farmland							
Conv. system	0.40[4]	-	0.11[7]	-	0.39[5,8]	**0.79** (0.54-0.79)	kg N cap^{-1} year^{-1}
Source sep. system	3.24[5]	+4.29[6]	3.09	-	2.12[5,8]	**3.89** (3.82-3.89)	kg N cap^{-1} year^{-1}
Phosphorus to farmland							
Conv. system	0.49	-	0.03[7]	-	0.54[5,8]	**0.31** (0.19-0.31)	kg P cap^{-1} year^{-1}
Source sep. system	0.72	+0.54[6]	0.44	-	0.6[5,8]	**0.61** (0.57-0.61)	kg P cap^{-1} year^{-1}

1) Re-calculated from joint results for eutrophication. 2) Re-calculated from joint results for aquatic eutrophication. 3) Re-calculated from emissions to soil and water. 4) Assumed 100% sludge to farmland. 5) Return of entire treated wet fraction. 6) Results only given as excess return with source separation system for nutrients with mineral fertilizer plant availability. 7) No nutrients are returned from the WWTP, only from food waste management. 8) Value is for plant available nutrients after emissions and run-off.

For freshwater eutrophication, the results from the present study suggest a slight increase with source separation systems. Results from the references studies are diverse; two studies suggesting similar results as the present study, while the other two studies state a decreased impact from using source separation systems compared to conventional systems. Thus, the literature seems in-conclusive on the potential impact on freshwater eutrophication. A possible reason could be the assumed technology for nutrient recovery, since the increased potential for freshwater eutrophication for the source separation system in the present study mainly stems from the usage of chemicals for ammonia stripping used in the recovery of nitrogen. The method in other studies could also have included direct emissions from wastewater treatment to freshwater. Another reason for a larger impact of the source separation system in this study was the larger amount of nutrients added to farm land in the source separation system which caused a larger total leakage of phosphorus from soil. As the estimated leakage in this study was larger than the average leakage from Swedish agricultural soils, the total results on freshwater eutrophication were somewhat overestimated.

In regard to marine eutrophication, the results are more straightforward. The main source of impact from both systems is direct discharge of nitrogen via the wastewater treatment plant effluent. However, a smaller contribution stems from agricultural run-off of nutrients used as fertilizers on farmland and ammonia emissions from storage and spreading, and the results suggest a slightly decreased impact for the source separation system. In this system, nutrients are mainly recovered as struvite and ammonium sulphate that give less run-off than sludge, which is the main source of nutrient return in the conventional system. The literature on similar studies all state that source separation system cause a decreased impact on marine eutrophication. Since this is in line with the results in the present study, it thus seems reasonable to conclude that source separation systems causes a decreased impact on marine eutrophication. When normalizing the results against the national average values of release of nitrogen from wastewater treatment plants, it becomes clear that the results from the present study, although the calculated discharges constitutes half of the national average, seems to have under-estimated the impact for the conventional system. This is probably explained by the exclusion of stormwater from the present study, which causes need for excessive nitrogen removal at the wastewater treatment plant to reach the discharge limits for nitrogen. This is not the case for the source separation systems since exclusion of stormwater is integrated in the concept of source separation it-self. In conclusion, it thus seems that the present study under-estimated the real life impact on marine eutrophication of the conventional system.

For the potential of returning nitrogen to farmland, the present study suggest that this can be greatly increased in source separation systems. These results are also in agreement with all of the reference studies who considered nutrient return to farmland in the system boundaries. The literature also seems also to agree fairly well with the results from the present study on the amounts of possible recovery. It thus seems fair to conclude that source separation system have a potential to greatly increase return of nitrogen to farmland. When normalizing the results against national average values of return of nitrogen to farmland it became apparent that the conventional system in the present study over-estimated the potential for recovery. This was due to an assumed higher degree of sludge return to farmland (43%) than the national average (25%), as well as the present study likely over-estimating the amount of nitrogen in sewage sludge for the conventional system. The latter is an effect of exclusion of stormwater from the present study. When normalizing against the suggested national targets for nutrient recovery from wastewater (10% of N) (SEPA, 2013), it became apparent that the conventional system barely could reach the suggested target for nitrogen, even though the recovery for the systems seems to be over-estimated in the present study. The source separation system, on the other hand, had the potential to greatly exceed the suggested target. Thus, an interesting option occurs where source separation systems could be partly integrated in a city to increase the average return of the entire urban area. Finally, if normalizing against the national use of mineral of nitrogen fertilizer, it was apparent that the potential for nitrogen recovery was low (4% for the conventional system and around 20% for the source separation system). It should however be kept in mind that this corresponds to as much as half of the Swedish planetary boundary for nitrogen fixation. In conclusion, the source separation system could greatly increase the amount of nitrogen returned to farmland, however still far from levels that can reduce the national dependence on mineral nitrogen fertilizer.

For the potential of returning phosphorus to farmland, the present study suggest that source separation system could increase the return. This result was also in agreement with all studies who included return of nutrients to farmland in their system boundaries (Table 21). It thus seems fair to conclude that source separation system can increase the return of phosphorus to farmland. When normalizing the results against national average values for return of phosphorus to farmland via sludge from wastewater treatment plants, it became apparent that the present study likely gives a fair estimation of phosphorus return for the conventional system. The results suggest a higher phosphorus return than the national average, however this is due to the assumption of a higher sludge return in the present study. If the effect of the increased sludge return is taken in to account, the present study gives a fair estimation of the phosphorus return. If normalizing the results against suggested national targets for nutrient recovery (40% of P) it becomes apparent that the conventional system barely reach the suggested legal targets, even if assuming the high sludge return as in the present study. The source separation system can, on the other hand, increase the return of phosphorus greatly, albeit not as much as the increase seen for nitrogen. Still, the beneficial aspects of having a source separation system in a smaller part of the urban catchment to increase the average return and thereby reaching the suggested targets still exists. Finally, if normalizing against the Swedish use of phosphate rock and the suggested planetary boundaries, it became apparent that the potential recovery of phosphorus from the source separation system constitute a larger fraction of the need (40-60%), when compared to the conventional system (20-30%). This corresponded in similar degree to the planetary boundary (40% for the source separation system and 20% for the conventional). In conclusion, source separation system could increase the return of phosphorus and to a substantial degree reduce the need for use of phosphate mineral fertilizer.

Finally, when comparing all of the results it seems clear that the source separation system has a good potential for reaching the suggested national goals for nutrient recovery from wastewater. Source separation system also seem very likely to decrease the marine eutrophication, as well as likely reducing climate change impacts. For freshwater eutrophication, the results seems to suggest a slight increase for source separation systems. Due to the high potential for nutrient recovery with source separation systems, these systems could beneficially be constructed in a part of an urban area to increase the average return of nutrients from a city to above suggested targets for nutrient recovery, otherwise being difficult to reach with the conventional system.

5. Conclusions

The study compared two hypothetical sanitation systems for management of domestic food waste and wastewater with aim of investigating impacts on climate change, freshwater eutrophication, marine eutrophication and potentials for return of nutrients to farmland. The first system was a conventional system currently operating in Sweden, while the other system represented a more uncommon source separation system, currently existing only in a few pilot areas in Europe. The main results of the study were:

- The source separation system was shown to decrease the climate impact with between 21 and 56 kg CO_2-eq. capita^{-1} year^{-1} compared to the conventional system. The decrease is mainly explained by higher biogas production, less emissions of nitrous oxide from wastewater treatment and higher recovery of nutrients to farmland in the source separation system.

- The source separation system was shown to greatly increase the potential for nutrient return to farmland, especially for nitrogen. The results showed that source separation system can be used to reach the suggested national goals for nutrient recovery from wastewater, which may not be reached by conventional Swedish systems alone.

- The source separation system was shown to have a larger impact on freshwater eutrophication due to chemical usage in extraction of nitrogen at the wastewater treatment plant, using an ammonia stripper. Oppositely, the source separation system was shown to have a decreased impact on marine eutrophication due to less emissions from nutrient fractions used in agriculture.

- The wastewater treatment plant, biogas usage and sludge and nutrient management constituted most of the contribution to all impact categories. Only a small part of the impact was constituted by household installations, sewer net and food waste management, which could potentially be excluded from future LCA-assessment of similar systems, given that the same impact categories are investigated.

- Infrastructure gave only a small contribution (7-15%) to the total impact on climate change and freshwater eutrophication and negligible impact on marine eutrophication. The contribution was similar in both systems, and infrastructure did not change the relative impact between the source separation systems and the conventional system. Thus, infrastructure can likely be omitted from similar studies in the future without changing the outcome between the studied systems and causing only a slight decrease in their total impact, given that the same impact categories are investigated.

6. Future work

The study showed that source separation systems can present environmental benefits when compared to conventional systems, and presented an overview of processes with higher and lower impact on the different environmental aspects investigated (climate change, eutrophication and nutrient recovery). This is a necessary start for an optimization of the source separation system. Several optimization strategies can be investigated in future work.

Since the ammonia stripper proved to give a large impact on many impact categories for the source separation system, the source separation system should preferably be extended to include a less chemical and heat intensive method for nitrogen recovery.

Emissions from storage and handling of fertilizers, especially organic fertilizers, should be further analyzed and emissions factors for different Swedish conditions including soil types should be developed since emissions from sludge storage were shown to have a great impact on the results in the present study.

A comparison between similar systems for more countries/regions would be preferable. The present study investigated source separation in a Swedish context, using national average data for modelling of energy systems. In Sweden, biogas is mainly used for vehicle fuel. However, in most other countries, such as Germany, France, Denmark, Austria and the UK, biogas is mainly used for electricity production (IEA Bioenergy, 2016). This makes sense, since average electricity production in these countries to a larger extent can be expected to be based on fossil energy. In addition, the present study showed the large influence from WWTP heat pumps on overall impacts to climate change and freshwater eutrophication, when heat is assumed to substitute average Swedish district heating. However, extensive district heating systems are rare in an international context. Thus, an investigation of source separation for more countries would be interesting.

Lastly, it should be pointed out that improvements to the conventional systems should be investigated in order to see how today's system could be improved. For example, the emissions of nitrous oxide from activated sludge was seen to give a big impact on climate change in the present study. The comparison to other nutrient removal technologies in the conventional system could potentially decrease the emissions of nitrous oxide from the wastewater treatment plant in the conventional system. Such changes could potentially decrease the climate impact for the conventional system.

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Appendix A1 – Mass balances

Mass balances for the two systems were calculated using Excel spread sheets. The mass balances were based on already published data using the System 1 and System 5 (alt.A) in the article by Kjerstadius et al. (2015) with the following alterations:

1) **Both systems:** The mass balances for heavy metals in Kjerstadius et al (2015), which were presented as weak data in the article, were compared to more recent data in (Yoshida et al., 2015). Out of mass balances for seven heavy metals three (Pb, Cd, Zn) showed good comparability between the studies while the mass balances where upgraded for the remaining heavy metals (Cu, Cr, Hg, Ni) according to Yoshida et al. (2015).
2) **Source separation system:** The post-precipitation in the source separation system was increased in order to meet the assumed discharge demands (10mgN/L and 0.5mgP/L). Increased removal was calculated for Fe^{3+} commercial agent (PIX) using data from Lindquist et al. (2003) and values is found in Appendix A2.
3) **Conventional system:** In order to meet the assumed discharge demands (10mgN/L and 0.5mgP/L) increased treatment was assumed since stormwater was not included in the study. This treatment consisted of increased BNR and post-precipitation. Increased removal was calculated for Fe^{3+} commercial agent (PIX) using data from Lindquist et al. (2003) and values are found in Appendix A2.
4) **Source separation system:** Including nitrification and denitrification in the activated sludge system for the source separation system according to data from an existing system (Wiersma & Elzinga, 2014; Wiersma, 2013). The effect of this on the mass balance is that roughly 10% of the total incoming N is released as N_2. The effluent N is decreased from 20% to 15% of total incoming and the amount of NH_3 strip is decreased from 73% to 68% (compared to Kjerstadius et al. (2015)). The methane production is also slightly decreased to constituting roughly an increased potential of 60% compared to conventional system instead of 70% as reported by (Kjerstadius et al., 2015). The final methane production is given in Table A1-1.

Table A1-1. Calculated methane production (as biogas) for the two systems. Calculations based on Kjerstadius et al. (2015).

Conventional system		Unit
FW AD-plant	7.2	NL CH_4 cap^{-1} d^{-1}
WWTP-biogas reactor	14.8	NL CH_4 cap^{-1} d^{-1}
Conventional system		**Unit**
WWTP-biogas reactor	35.1	NL CH_4 cap^{-1} d^{-1}

MASSBALANCE CONVENTIONAL SYSTEM:

The final mass balances for the conventional system is shown graphically in Figure A1-1 and the data are given in Table A1-2.

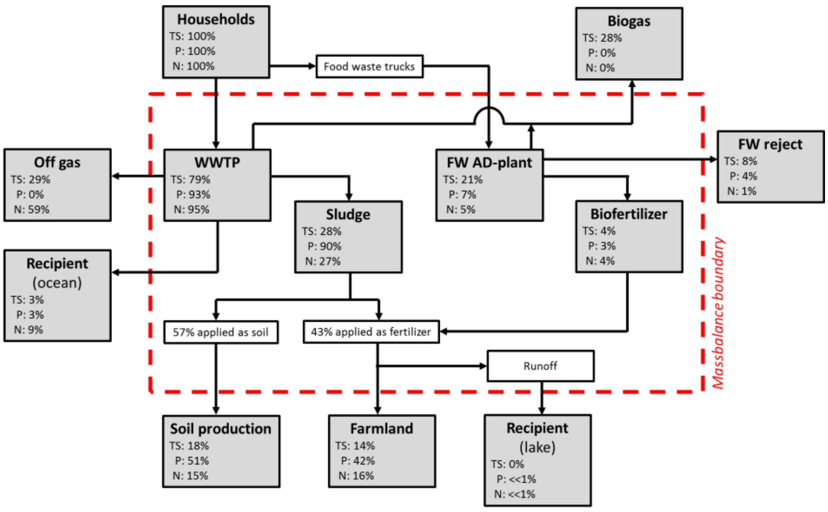

Figure A1-1 – Mass balances for total solids (TS), phosphorous (P) and nitrogen (N) over the conventional system.

Table A1-2. Mass balance total solids (TS), volatile solids (VS), phosphorous (P) and nitrogen (N) over the conventional system. *(For greywater COD was used instead of VS).

Conventional system	Wet weight	TS	VS*	P-tot	N-tot	Unit
INFLOWS						
Households	-	161.7	130.8	2.06	14.5	g cap^{-1} d^{-1}
OUTFLOWS						
WWTP-Recipient	-	4.4	3.6	0.07	1.4	g cap^{-1} d^{-1}
Biogas	-	45.2	47.8	0.00	0.0	g cap^{-1} d^{-1}
Activated sludge off gas	-	46.2	36.8	0.00	8.5	g cap^{-1} d^{-1}
WWTP sludge	193.8	45.9	27.6	1.85	3.8	g cap^{-1} d^{-1}
FW reject to incineration	36.0	12.6	10.7	0.08	0.2	g cap^{-1} d^{-1}
FW AD-plant biofertilizer	195.1	7.4	4.2	0.06	0.6	g cap^{-1} d^{-1}
SUM OUTFLOWS	**	161.7	130.8	2.06	14.5	g cap^{-1} d^{-1}

*For greywater COD was used instead of VS.

** Since water was not included in the mass balance the wet weight is only included for fractions that needed transport by truck.

Table A1-3. Mass balance for heavy metals [mg cap^{-1} d^{-1}].

Conventional system	Cd	Cr	Cu	Hg	Ni	Pb	Zn
INFLOWS							
Food waste[1]	0.01	0.34	1.36	0.0006	0.20	0.68	1.74
Blackwater	0.01	0.03	1.00	0.01	0.07	0.02	8.97
Greywater	0.03	1.00	6.82	0.004	1.23	0.95	9.95
OUTFLOWS							
WWTP-Recipient	0.01	0.13	0.17	<0.01	0.70	0.02	1.14
WWTP-sludge	0.04	0.90	7.65	0.01	0.60	0.95	17.78
FW reject to incineration	<0.01	0.06	0.25	<0.01	0.04	0.12	0.32
FW AD-plant biofertilizer	<0.01	0.11	0.43	<0.01	0.06	0.22	0.55

1) This value represent 100% of the generated food waste. However, only 50% of this was assumed being sorted at household level.

MASSBALANCE SOURCE SEPARATION SYSTEM:

The final mass balances for the conventional system is shown graphically in Figure A1-2 and the data are given in Table A1-4.

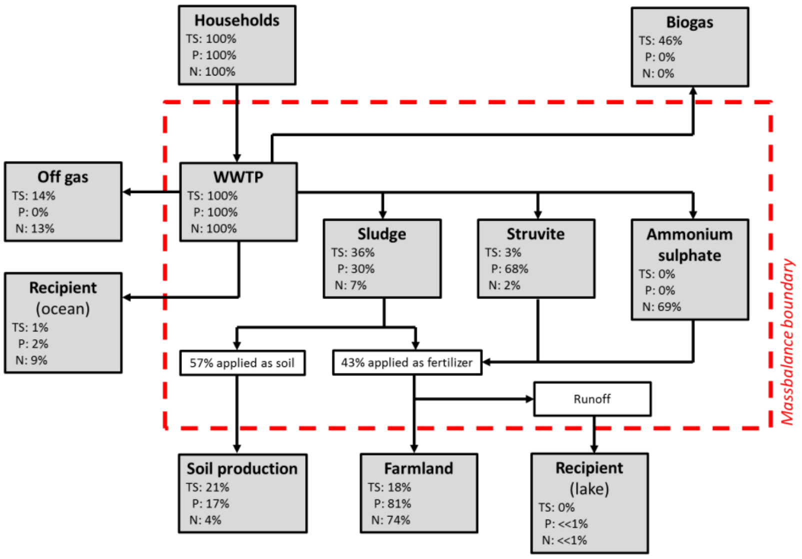

Figure A1-2 – Mass balances for total solids (TS), phosphorous (P) and nitrogen (N) over the source separation system.

Table A1-4. Mass balance total solids (TS), volatile solids (VS), phosphorous (P) and nitrogen (N) over the source separation system. *(For greywater COD was used instead of VS).

Source separation system	Wet weight	TS	VS*	P-tot	N-tot	Unit
INFLOWS						
Households	-	161.7	130.8	2.06	14.5	g cap^{-1} d^{-1}
OUTFLOWS						
WWTP-Recipient	-	1.5	1.2	0.04	1.3	g cap^{-1} d^{-1}
Biogas	-	74.7	72.1	0.00	0.0	g cap^{-1} d^{-1}
Ammonium sulphate	36.0	0.0	0.0	0.00	9.9	g cap^{-1} d^{-1}
WWTP sludge	243.3	57.7	35.1	0.62	1.1	g cap^{-1} d^{-1}
WWTP struvite	11.8	4.9	2.7	1.40	0.3	g cap^{-1} d^{-1}
Activated sludge off gas	-	23.0	19.8	0.00	1.9	g cap^{-1} d^{-1}
SUM OUTFLOWS	**	161.7	130.8	2.06	14.5	g cap^{-1} d^{-1}

*For greywater COD was used instead of VS.

** Since water was not included in the mass balance the wet weight is only included for fractions that needed transport by truck.

Table A1-5. Mass balance for heavy metals [mg cap^{-1} d^{-1}].

Source separation system	Cd	Cr	Cu	Hg	Ni	Pb	Zn
INFLOWS							
Food waste[1]	0.01	0.34	1.36	0.0006	0.20	0.68	1.74
Blackwater	0.01	0.03	1.00	0.01	0.07	0.02	8.97
Greywater	0.03	1.00	6.82	0.004	1.23	0.95	9.95
OUTFLOWS							
WWTP-Recipient	0.01	0.09	1.46	<0.01	1.06	0.04	1.74
WWTP-sludge	0.04	1.10	7.04	0.01	0.33	1.27	18.04
Struvite	<0.01	<0.01	<0.01	<0.01	0.01	<0.01	0.01
Ammonium sulphate[2]	0.00	0.00	0.00	0.00	0.00	0.00	0.00

1) This value represent 100% of the generated food waste. However, only 50% of this was assumed being sorted at household level. 2) Metal content in ammonium sulphate assumed zero.

Appendix A2 – Indata for Life cycle assessment

Table A2-1 Data used in modelling of households.

Included			Not considered		
Component name	Reference	Comment	Component name	Reference	Comment
Production of piping material (conventional system).	Remy (2010).	Conventional system: PP50-100mm for apartment installations. Down pipe and Base pipe both PP200mm. Total length of pipes (divided per apartment) was 12.25m.	Construction of toilets and vacuum toilets.	-	Was not included in Remy (2010).
Production of piping material (source separation system).	Remy (2010).	Conventional system: PP50mm (for GW), PE50mm (for BW) and PE110mm (for FW) apartment installations. Down pipe and Base pipe for GW is PP100mm and PP200mm respectively. Down pipe and Base pipe for BW is PE50mm. Down pipe and Base pipe for FW is PE110mm. Total length of pipes (divided per apartment) was 24.25m.	Emissions for installation of piping.	-	-
			Production of fresh water.	-	Water consumption not included in study.
			Less water consumption from vacuum toilets.	-	-
			Electricity consumption vacuum toilets	-	Is included in sewer net since electricity is consumed by the vacuum pumps.

Table A2-2 Data used in modelling of food waste handling (conventional system).

Included			Not considered		
Component name	Reference	Comment	Component name	Reference	Comment
Vessel for food waste separation	Bissmont (2014)	Weight = 0.22kg Material = HDPE	Transformation of HDPE into vessel		
Paper bags for food waste separation	Bissmont (2014)	Weight = 19g/bag	Transformation of paper into bags		
Production of food waste disposers	Annerhall (2010)	Weight = 5.5kg 32% iron, 7% steel, 31% HDPE, 2.5% rubber, 0.5% copper, 1.5% aluminum	Manufacturing of the disposer		
Electricity in food waste disposers	Annerhall (2010)	Assuming 373W and 2.5 uses per day, 30s each time			
Transport of food waste	Rehnlund (2010).	8.2 kWh diesel km^{-1}			
Screwpress	SWMA (2013)	3.7t steel 35.5kWh electricity/t food waste	Other materials, maintenance and end-of-life		
Biogas reactor for digestion of food waste	Remy (2010)	Concrete 557.3kg/m^3, Steel non-alloyed 13.9kg/m^3, Cast iron 2.3kg/m^3, Steel alloyed 166.9kg/m^3, Polyethylene 1.7kg/m^3 reactor volume.	Control-system, maintenance		
Incineration of pretreatment residues	Sysav (2015)	15% electricity recovery, 85% heat recovery, 85% overall energy recovery. Auxiliar materials (per t treated ww): $CaCO3$: 7.6kg, $CaOH2$: 2.6kg, NaOH: 3.8kg, $NH4OH$: 4.4kg, HCl: 0.15kg, $FeCl3$: 0.03kg, Fuel oil: 0.001m^3.			
Pretreatment residues	Truedsson (2010)	Energy content = 4475 MJ/ton residues (ww).			

Table A2-3. Data used in modelling of sewer net.

Included			Not considered		
Component name	Reference	Comment	Component name	Reference	Comment
Sewer net length (conventional system)	Dahl, 2013.	4.9m/cap. Calculated for the city of Helsingborg.	Excavation due to reparations during technical life-span.	-	-
Sewer net length (source separation system)	Kärrman et al. (in press)	Assumed 3.7m/cap. Based on the rationale that a local treatment plant is used for source separation system. 3.7m/cap is based on the Floor Space Index of the H+ area in the city of Helsingborg (2.0 compared to 1.5 for the rest of the city).	Manholes	-	-
Sewer piping material	Remy (2010)	PP-200mm for conventional system and for GW sewer. PE-100mm for blackwater and FW sewer. Selection based on Remy (2010) but excludes cast iron.			
Sewer excavation	-	Assumed 2m^3/m sewer with rammed earth (1 600 kg m^{-3}).			
Excavation pump pit	-	Assumed 9 600kg earth per pump (same for all pumps). Calculated from supplier data of needed pump pit volume (4.3m^3). Assumed 6m^3 excavation with rammed earth (1 600 kg m^{-3}).			
Infrastructure pumps (LPS & vacuum)	Remy (2010)	Same for all pumps (LPS and vacuum)			
Electricity demand LPS pumps	Dahl, 2015.	Assumed to 0.1 kWh/m^3. Pumping gives 17m head which is deemed sufficient.			

Electricity demand vacuum generator	Markstedt, 2015.	Data for Jets 190 MB. 5 500 kWh/year.			

Table A2-4. Data used in modelling of WWTP.

Included			Not considered		
Component name	Reference	Comment	Component name	Reference	Comment
Excavation and infrastructure for construction of conventional WWTP	Remy (2010)	Calculated using model by Remy (2010) and mass balances from Kjerstadius et al. (2015).	CH4 emissions from COD in effluent recipient (sea)	IPCC (2006a)	-
Electricity & heat for conventional WWTP.	Remy (2010)	57kWh$_{electricity}$/cap and 49kWh$_{heat}$/cap. Calculated using model by Remy (2010) and mass balances from Kjerstadius et al. (2015). Presented in Kärrman et al. (in press).	Treatment of stormwater and industrial wastewater	-	Not included in system boundaries
Excavation and infrastructure for source separation WWTP	Witteveen Bos (2014)	Based on pilot area in Sneek (NL).	Heat and electricity for personnel housing	-	Assumed same for both systems
Electricity & heat for source separation WWTP (excl. struvite and ammonia stripping)	de Graaf & van Hell (2014) and Meulman (2015)	Calculated from pilot plant in Sneek (NL). 8kWh$_{electricity}$/cap and 142kWh$_{heat}$/cap.	Material in control systems	-	Assumed same for both systems
Chemicals for sludge dewatering & foaming control	NSVA (2014) & Tumlin et al. (2014).	Calculated from environmental report for Öresundsverket WWTP (Helsingborg). Includes transport of chemicals according to Tumlin et al. (2014).	Heat exchanger on wastewater	-	Not included.
Emissions of N2O from BNR	Foley et al. (2010) and Foley et al. (2008).	0.01 kg N_2O-N/kg Ndenitrified.	Emissions from personnel manhours and transport	-	Assumed same for both systems

Methane emissions from wastewater treatment	Göthe. (2013)	0.1g CH4/kWh (0.135% of produced methane assuming density of 717 g/Nm3)			
Methane emissions from biogas upgrading	Bauer et al. (2013)	1% of methane in gas for upgrading			
N20 emissions from N in effluent recipient (sea)	Foley et al. (2008).	0.002 kg N2O-N/kg N			
Marine eutrophication potential of nitrogen discharged in wastewater treatment plant effluent.	Assumed.	1 kg N-eq. per kg N compound in the effluent. This is lower than ReCiPe who states a factor of 1.429 kg N-eq. per kg N compound released to ocean water.			
Marine eutrophication potential of phosphorus discharged in wastewater treatment plant effluent.	ReCiPe(2016)	0 kg N-eq. per kg P compound in the effluent.			
Infrastructure for struvite precipitation	Thelin (2015)	Steel (0.62 kg stainless steel/cap/life time), plastic (0.22 kg PE/cap/life time) and concrete (1.04 kg/cap/life time). Calculated for 12 000 cap. Assumed life time 30 years.			
Electricity for struvite precipitation	Thelin (2015)	Calculated for 12 000 cap. $2.5kWh_{electricity}$/cap.			
Chemical usage struvite precipitation	Thelin (2015)	0.5 kg Mg (from $MgCl_2$) and 0.12 kg citric acid $capita^{-1}$ $year^{-1}$.			
Infrastructure for ammonia stripper	Thelin (2015)	Steel (0.56 kg stainless steel/cap/life time), plastic (0.33 kg PE/cap/life time) and concrete (1.90 kg/cap/life time). Calculated for 12 000 cap. Assumed life time 30 years.			
Electricity and heat for ammonia stripper	Thelin (2015)	Calculated for 12 000 cap. $7.7kWh_{electricity}$/cap and $78kWh_{heat}$/cap.			

Possible heat reduction with heat recovery on ammonia stripper	Thelin (2015)	Reduced heat consumption from $78kWh_{heat}$/cap to $53kWh_{heat}$/cap.			
Chemical usage for ammonia stripper	Thelin (2015)	20 kg NaOH, 15 kg H_2SO_4 and 0.3 kg citric acid $capita^{-1}$ $year^{-1}$.			
Efficiency heat pump	Hellborg Lapajne (2016)	Conventional system: COP of 3.9. 445 kWh_{heat} cap^{-1} $year^{-1}$ and 114 $kWh_{electricity}$ cap^{-1} $year^{-1}$. Source separation system: COP of 4.7. 420 kWh_{heat} cap^{-1} $year^{-1}$ and 90 $kWh_{electricity}$ cap^{-1} $year^{-1}$.			
Climate impact of NaOH	Dahlgren et al. (2015).	0.259 kg CO_2-eq./kg NaOH			
Post-precipitation	Lindquist et al. (2003)	Assumed PIX. 1.5mole Fe^{3+}/moleP. The effect being reduction of 90% total solids, 95% of total phosphorus and 25% of total nitrogen.			
Climate impact of PIX	Homa & Hoffman (2014)	Used average value from report. 0.106 kg CO_2-eq/mole Fe^{3+} in PIX.			

Table A2-5. Data used in modelling of biogas upgrading & usage.

Included			Not considered		
Component name	Reference	Comment	Component name	Reference	Comment
Steel needed for construction of upgrading facility	Calculated	First calculated from Starr et al. (2012) but the needed mass seemed unreasonably small. Instead an approximation of the mass needed for the upgrading plant at Öresundsverket WWTP was used.	Infrastructure in fuel depots	-	-
Biogas process slip	Göthe (2013)	2.8%-mass of produced biogas	Infrastructure for buses	-	-
Biogas slip upgrading facility (Water scrubber)	Göthe (2013)	2%-mass of upgraded biogas	Transport distance to fuel depots and time for refueling	-	-
Energy use upgrading water scrubber	Tumlin et al. (2014).	0.25kWh/Nm3 upgraded biogas	Water needed in water scrubber	-	Water usage not included in system boundaries
Compression of upgraded biogas	Benjaminsson and Nilsson (2009)	0.18kWh/Nm^3			
Substitution of diesel in buses	Tumlin et al. (2013).	1 kWh diesel/kWh methane			
Propane dosing to upgraded biogas	Benjaminsson and Nilsson (2009)	0.001kWh/Nm^3			
Propane dosage (for grid injection)	Benjaminsson and Nilsson (2009)	131g/Nm^3			
N2O emissions from biogas use in buses	Fruergaard and Astrup (2010)	1,4E-05g N2O/kg burned CH4			

CH4 emissions from biogas use in buses	Fruergaard and Astrup (2010)	0.00073 g CH4/kg burned CH4			
Emissions from incineration of propane	Ecoinvent database 3.0	3 g CO_2/g C_3H_8			

Table A2-6. Data used in modelling of management of sludge & nutrient fractions.

Included			Not considered		
Component name	Reference	Comment	Component name	Reference	Comment
Storage container	LarvCement (2012), calculations of impact from ecoinvent database 3,0	Based on a 1000 m^3 container, about 168 tons of concrete and 1.4 ton cast iron, also excavation included			
Ammonia emissions from sludge storage	Karlsson and Rodhe (2002)	10% of N-tot, as for semi-solid manure			
Nitrous oxide emissions from sludge storage	Flodman, 2002 (direct N_2O) IPCC, 2006 (indirect N_2O)	197.6 mg N_2O/m^3,h (sludge) and indirect 1% of NH_3 emissions			
Methane emissions from sludge storage	Flodman, 2002	123.3 mg CH_4/m^3,h (sludge)			
Spreading operation, sludge	ecoinvent database 3.0	Spreading operation, solid manure, hydraulic loader			
Ammonia emissions from sludge spreading	Karlsson and Rodhe (2002)	27% of NH_4-N, average solid and liquid manure			
Nitrous oxide emissions from sludge spreading	IPCC, 2006	1% of N-tot applied and 1% of NH_3 emitted			

Spreading operation, avoided mineral fertilizer	ecoinvent database 3.0	Fertilising, by broadcaster			
Carbon sequestration, sludge	Linzner and Mostbauer, 2005	Storage factor for digestate, 3.8%			
Nitrogen loss from composting	Vogt et al., 2002	30% av total N			
Ammonia emissions from composting of sludge	Boucher et al., 1999	NH_3: 66% av total N loss			
Nitrous oxide and methane emissions from composting of sludge	Kirkeby et al., 2005	N_2O: 2% av total N loss, CH_4: 0.75% av C-tot			
Energy use composting	Kirkeby, 2005	Electricity:41 kWh/ton TS, diesel:12 l/ton TS			
Carbon sequestration, compost	Linzner and Mostbauer, 2005	Storage factor for compost, 8.4%			
Spreading operation, compost	ecoinvent database 3.0	Spreading operation, solid manure, hydraulic loader			
Ammonia emissions from biofertilizer storage	Karlsson and Rodhe (2002)	1% of N-tot, as for liquid manure			
Nitrous oxide emissions from biofertilizer storage	Rodhe et al., 2013 (direct N_2O) IPCC, 2006 (indirect N_2O)	Direct 0.24% of N-tot and indirect 1% of NH_3 emissions			
Methane emissions from biofertilizer storage	Rodhe et al., 2013	16.28 gCH4-C/kgVS			
Spreading operation, biofertilizer	ecoinvent database 3.0	Spreading operation, liquid manure, vacuum tanker			
Emissions from spreading biofertilizer	Rodhe et al., 2013; IPCC, 2006 (indirect N_2O)	NH_3: 15% of NH_4-N, N_2O direct: 0.10% of N-tot, N_2O indirect: 1% of NH_3 emissions			

Carbon sequestration	See sludge above				
Avoided nitrogen production	ecoinvent database 3.0	AN, as 100% $(NH_4)(NO_3)$, NPK (35-0-0), at regional storehous	Transport not included		
Avoided phosphorus production	ecoinvent database 3.0	TSP, as 80% $Ca(H_2PO_4)_2$, NPK (0-48-0), at regional storehouse	Transport not included		
Diesel use	ecoinvent database 3.0	Diesel, low sulphur, regional storage			
Eutrophication from N fertilizer	ecoinvent database 3.0	Fertilizer, applied: 0.073 kg N/kg N applied Manure, applied: 0.079 kg N/kg N applied			
Eutrophication from P fertilizer	ecoinvent database 3.0	Fertilizer, applied: 0.053 kg P/kg P applied Manure, applied: 0.050 kg P/kg P applied			

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Witteveen Bos (2014). *Onderzoek duurzaamheid nieuwe sanitatie (Sustainability investigation of nieuwe sanitatie)*. Deventer (NL): Witteveen Bos.

Details regarding ecoinvent LCI-processes

Geographical range: All processes were modelled using average European data (RER) when available in the ecoinvent v 3.0 database (ecoinvent, 2013). If not available, global average data (GLO) was used.

Modelling approach: In all cases, attributional LCI processes were used.

Adjustments of ecoinvent datasets: Datasets were adjusted to Swedish conditions when relevant (see details below):

Paper bags for collection of food waste: Electricity and heat in datasets for processes for provision of pulp and paper for production of paper bags were changed to Swedish processes. A process for production of graphical paper was used, with input of sulfate pulp, which accounts for the majority of the Swedish pulp production (Skogsindustrierna, 2014). Transformation of paper to paper bags was not included. Transports of paper bags from production plant to final use was not included.

Vessels for separate collection of food waste in households: Production of plastic used in vessels for food waste collection in households were not adjusted, as it is assumed that they are produced outside of Sweden. Transformation of plastic into vessels was not included. Transports of vessels from production plant to final use was not included. The amount of vessels used per person was based on an assumed change of vessel every fifth year. End-of-life treatment of vessels was no included in the study.

Plastic wastewater pipes: Production of plastic used in vessels for food waste collection in households were not adjusted, as it is assumed that they are produced outside of Sweden. Transformation of plastic into pipes was not included.

Other processes of relevance:

Heat: Average Swedish district heating was based on Gode et al. (2011), representing the national mix in 2008. Primary energy vales for used fuels were converted to thermal energy using conversion factors presented by Gode et al. (2011). Provision of fuels and emissions from combustion were modelled in Simapro using average European processes when available and global when not. In the case of wood chip and bio-oils, where no processes for generation of heat were found in ecoinvent v 3.0, a process for combustion of wood pellets was adjusted to include provision of these fuels, while emissions were assumed to be the same as for pellets. The same was done in the case of peat, however, in this case emissions of fossil carbon dioxide were added to the emission profile, based on Gode et al. (2011).

Sewer net: Excavation sewer net was calculated as adjusted to 1kg plastic wastewater pipe (15840 kg /11.39 kg plastic pipe), assuming a density of soil of 1300kg/m3 = 1.07023m3/kg excavation per kg of plastic pipe.

Chemicals: Chemicals used in conventional WWTP and ammonia stripper were collected from ecoinvent (2013). Global processes were used, indicating global average transport distances rather than actual ones. Data on emissions for chemicals was taken from Incopa (2014).

*Substitution of diesel in buses***:** Substitution was based on the ecoinvent process "Transport, regular bus CH", with the unit "personkm". 0.024986kg diesel/personkm is used in this process. This amount was adjusted to 1kg and emissions were adjusted to the same unit. Use of buss and road (infrastructure) were assumed to be similar and thus excluded from the process.

Appendix A3 – Comparison to similar studies

Functional unit:

Table A3-1 presents the functional units from selected relevant studies dealing with LCA comparison between conventional systems and source separation systems similar to the systems studied in the present study. The functional units have been translated from German for the study of Hillenbrand (2009) and from Dutch for the study of Witeveen Bos (2014).

Table A3-1. Comparison between functional units utilized in selected studies.

Functional Unit (FU)						
Process	**Remy (2010)**	**Hillenbrand (2009)**	**Meinzinger (2010)**	**Witeveen Bos (2014)**	**Thibodeau (2014)**	**Present study**
Functional unit	Provision of the primary functions per person and year.	Supply of drinking water and the disposal of wastewater per person and year.	Provision of urban water and wastewater service with integrated nutrient recovery.	Processing domestic fruit and vegetable waste and purification of domestic wastewater from 100 000 in - habitants for 50 years to a minimum of sufficient quality to be able to discharge it to surface water in accordance with environmental permit.	To ensure wastewater and organic kitchen refuse collection and treatment and by-product (digestate/sludge and biogas) recycling for one inhabitant for one year	Management of 1 person equivalent load of food waste, blackwater and greywater per year.
Comment	Primary functions include supply of drinking water as well as transport and disposal of blackwater, greywater and food waste.	Includes food waste.	Study not defined as a LCA but as a ceMFA (cost, energy and material flow analysis) to include relevant processes and flows per capita and year.	-	-	-
Study related to city/region	SCST-project Berlin, Germany.	Related to the DEUS 21 pilot area in Knittlingen, Germany.	Hamburg, Germany.	Based on the Sneek Noorderhoek pilot area in the Netherlands.	Quebec, Canada.	H+ pilot area, Helsingborg, Sweden.

Impact categories:

Table A3-2 presents the impact categories used in selected relevant studies dealing with LCA comparison between conventional systems and source separation systems similar to the systems studied in the present study.

Table A3-2. Comparison of used impact categories in selected studies on source separation systems. The usage of "AND" indicate the impact categories were presented separately in the original study.

Emission-related						
Remy (2010)	**Hillenbrand (2009)**	**Meinzinger (2010)***	**Witeveen Bos (2014)**	**Thibodeau (2014)**	**Present study**	**Indicator**
Climate change	Climate change	-	Climate change	Climate change	Climate change	kg CO_2-eq
Eutrophication[2]	Terrestrial AND Aquatic eutrophication[2]	Emission of P to soil/groundwater AND surface water	Freshwater eutrophication		Freshwater eutrophication	kg P-eq
-	-	Emission of N to soil/groundwater AND surface water	Marine eutrophication		Marine eutrophication	kg N-eq
Acidification	Acidification	-	Acidification		-	kg SO_2-eq
-	Photochemical oxidant formation[3]	-	Photochemical oxidant formation		-	kg NMV OC
-	-	-	Ozone depletion		-	kg CFC-11-eq
Toxicity-related						
Remy (2010)	**Hillenbrand (2009)**	**Meinzinger (2010)***	**Witeveen Bos (2014)**	**Thibodeau (2014)**	**Present study**	**Indicator**
Human toxicity	-	-	Human toxicity	-	-	kg DCB-eq
-	-	-	-	Human health	-	DALY
Freshwater aquatic ecotoxicity	-	-	Freshwater AND marine aquatic ecotoxicity	-	-	kg DCB-eq
Terrestrial ecotoxicity	-	-	Terrestrial ecotoxicity	-	-	kg DCB-eq
-	-	-	-	Ecosystem quality	-	PDF m^2 year
Resource-related						
Remy (2010)	**Hillenbrand (2009)**	**Meinzinger (2010)***	**Witeveen Bos (2014)**	**Thibodeau (2014)**	**Present study**	**Indicator**
Depletion of abiotic resources	-	-	-	-	-	kg Sb-eq
Cumulative energy demand[1]	Cumulative energy demand	Primary energy consumption[4]	-	Resources	-	MJ
-	Need of drinking water replacement	Groundwater extraction	Depletion of drinking water	-	-	m^3

-	Fossil depletion	-	Fossil depletion	-	-	kg oil-eq
-	-	-	Abiotic depletion	-	-	kg Fe eq
-	Need of P-replacement	Recovered P and N	-	-	P and N returned to farmland	kg P or kg N

* Used ceMFA (cost, energy and material flow analysis) rather than LCA. 1) Only fossil+nuclear fuels. 2) Used kg PO_4-eq as indicator. 3) Used kg POCP-eq and kg Ethen-eq as indicators. 4) Used kWh pe^{-1} y^{-1} as indicator.

System boundary:

Table A3-3 roughly presents the system boundary used in selected relevant studies dealing with LCA comparison between conventional systems and source separation systems similar to the systems studied in the present study.

Table A3-3. Rough description of system boundary in relevant studies.

System boundary						
Process	**Remy (2010)**	**Hillenbrand (2009)**	**Meinzinger (2010)**	**Witeveen Bos (2014)**	**Thibodeau (2014)**	**Present study**
Food waste management	Yes	Yes	Yes	Yes	Yes	Yes
Drinking water production	Yes	Yes	Yes	Yes	Yes	No
Includes infrastructure ?	Yes	Yes	No (only for cost estimation)	Yes	Yes	Yes
Includes end-of-life	No	Yes	No	Yes	Yes	No
Stormwater	No	Yes	Yes	Yes	No	No
Other wastewaters (industrial)	No	No	No	No	No	No
Nutrient recovery	Yes	Yes	Yes	No	Yes	Yes

System description:

Table A3-4 and Table A3-5 roughly presents the systems studied in selected relevant studies dealing with LCA comparison between conventional systems and source separation systems similar to the systems studied in the present study. Note that several more system were investigated in the studies, however presented below are the systems deemed most relevant to compare with the present study.

Table A3-4. Description of conventional systems in relevant studies.

<table>
<tr><th colspan="7">Processes – conventional system</th></tr>
<tr><th>Process</th><th>Remy (2010)</th><th>Hillenbrand (2009)</th><th>Meinzinger (2010)</th><th>Witeveen Bos (2014)</th><th>Thibodeau (2014)</th><th>Present study</th></tr>
<tr><td>System name in original study</td><td>R_{agri}</td><td>KONV, stadt</td><td>1 CurS</td><td>Centrale rwzi</td><td>CONV</td><td>Conventional system</td></tr>
<tr><td>Food waste (FW)</td><td>Collection in Bins. Transport to compost (80%) or incineration (20%). Compost returned to farmland replacing nutrients.</td><td>Collection in Bins. Transport to compost. Compost returned to farmland replacing nutrients.</td><td>Collection in Bins. Transport to compost. Compost returned to farmland replacing nutrients.</td><td>FW-not included in the study for the conventional system.</td><td>Transported by truck to food waste treatment facility. Treatment includes shredding, pre-treatment (grinder and pasteurizer), anaerobic digestion, and dewatering. Dewatered sludge is returned to farmland.</td><td>Collected in bins. Transport to separate anaerobic digestion plant. Digestate is returned to farmland replacing nutrients.</td></tr>
<tr><td>Blackwater (BW)</td><td rowspan="2">Combined gravity sewer. Treatment at WWTP includes CAS with BNR and P-precipitation. Sludge is anaerobically digested, dewatered and returned to agriculture - replacing nutrients.</td><td rowspan="2">Combined gravity sewer. Treatment at WWTP includes CAS with BNR and P-precipitation. Sludge is anaerobically digested, dewatered and incinerated. No return of nutrients to farmland was considered.</td><td rowspan="2">Combined gravity sewer. Treatment at WWTP includes CAS with BNR and P-precipitation. Sludge is anaerobically digested, dewatered and incinerated. No return of nutrients to farmland was considered.</td><td rowspan="2">Combined gravity sewer. Treatment at WWTP includes CAS with BNR and P-precipitation. Sludge is anaerobically digested, dewatered and incinerated. No return of nutrients to farmland was considered.</td><td rowspan="2">Combine d gravity sewer. Treatment at WWTP includes CAS with BNR and P-removal. Tickened sludge is anaerobically digested, dewatered and returned to farmland.</td><td rowspan="2">Combined gravity sewer. Treatment at WWTP includes CAS with BNR and P-precipitation. Sludge is anaerobically digested and dewatered. 43% of dewatered sludge is returned to agriculture - replacing nutrients. Remaining sludge is composted and used as soil improver.</td></tr>
<tr><td>Greywater (GW)</td></tr>
</table>

Table A3-5. System description of source separation systems in relevant studies.

Processes – Source separation system						
Process	**Remy (2010)**	**Hillenbrand (2009)**	**Meinzinger (2010)**	**Witeveen Bos (2014)**	**Thibodeau (2014)**	**Present study**
System name in original study	V1	DEUS	4: CoDig	Geoptimaliseerd decentral systeem	BWS	Source separation system
Food waste (FW)	Collected locally and anaerobically digested together with BW. Dewatered sludge returned to farmland replacing nutrients.	Collected with food waste disposers and transported in vacuum-sewer together with blackwater.	Collection in Bins. Transport to digester and anaerobically digested together with BW.	Collected with food waste disposers and transported in vacuum-sewer together with blackwater.	Kitchen refuse is transported by truck and dumped into a shredder in the treatment building. Black water and shredded kitchen refuse go into a pre-treatment, which includes a grinder and a pasteurizer and then through an anaerobic digester.	Collected with food waste disposers and transported in separate sewer. Treated together with BW.
Blackwater (BW)	Collected with vacuum sewer and anaerobically digested together with FW. Dewatered sludge returned to farmland replacing nutrients.	Collected with vacuum sewer and anaerobically digested in anaerobic membrane bioreactor together with FW and GW. Effluent is treated with struvite precipitation (mainly P-recovery) and zeolite with external air/acid stripping (N-recovery). Sludge is dewatered and incinerated without nutrient recovery.	Collected with vacuum sewer and anaerobically digested together with FW. Dewatered sludge is incinerated and P is recovered through acid leaching. Extracted P is returned to farmland replacing nutrients. Reject water from sludge dewatering is treated with ammonium stripper recovering N to farmland.	Collected with vacuum sewer and treated together with FW in up-flow anaerobic digester. Effluent treated for struvite precipitation and anammox-nitrogen removal. No return of nutrients was considered.	Collected with vacuum system and low-pressure pump. Black water and shredded kitchen refuse go into a pre-treatment, which includes a grinder and a pasteurizer and then through an anaerobic digester. Digestate from the anaerobic digester is sent and stored on farmland without dewatering	Collected with vacuum sewer and treated together with FW in up-flow anaerobic digester. Effluent treated for struvite precipitation and ammonium stripper (N-recovery). All struvite and ammonium strip is returned to farmland. 43% of dewatered sludge is returned to agriculture. Remaining sludge is composted and used as soil improver.
Greywater (GW)	Treated in sequencing batch reactor. Sludge is dewatered and incinerated without nutrient recovery.	Collected with gravity sewer and treated together with BW and FW.	Collected with gravity sewer and treated with CAS with BNR. Sludge is treated together with BW sludge.	Collected with gravity sewer. Treated in high load activated sludge.	Collected with gravity sewer and treated by septic tank and a constructed wetland.	Collected with gravity sewer. Treated in high load activated sludge. Sludge is treated together with BW.

Selected results:

Table A3-6 presents selected results from relevant studies dealing with LCA comparison between conventional systems and source separation systems similar to the systems studied in the present study.

Table A3-6. Comparison of results in selected studies on source separation systems.

	Remy (2010)	Hillenbrand (2009)	Meinzinger (2010)	STOWA (2014)	Thibodeau (2014)	Present study	Unit
Biogas production							
Conv. system	12.1	-	-	61	121.5	80	kWh cap^{-1} y^{-1}
Source sep. system	44.4	-	-	122	116.5	128	kWh cap^{-1} y^{-1}
Net energy demand							
Conv. system	66.7[1,6]	860[6]	278[2,6]	88[2,6]	219[2,6,8]	$165_{electr.}$ $-392_{thermal}$	kWh cap^{-1} y^{-1}
Source sep. system	15.6[1,6]	1 500[6]	325[2,6]	-184[2,6]	274[2,6,8]	$119_{electr.}$ $-281_{thermal}$	kWh cap^{-1} y^{-1}
Nitrogen to farmland							
Conv. system	402[3]	-	110[5]	-	390[7]	792	gN cap^{-1} y^{-1}
Source sep. system	3 241[4]	+4 290	3 090	-	2 120[7]	3 894	gN cap^{-1} y^{-1}
Phosphorus to farmland							
Conv. system	490	-	30[5]	-	540[7]	313	gP cap^{-1} y^{-1}
Source sep. system	718	+540	440	-	600[7]	609	gP cap^{-1} y^{-1}

1) Includes substituted mineral fertilizer. 2) Calculated for primary energy. A negative sign indicates net energy production. 3) Assumed 100% sludge to farmland. 4) Return of entire treated wet fraction. 5) No nutrients are returned from the WWTP, only from food waste management. 6) Includes drinking water production. 7) Considers plant available nutrients after run-off and emissions. 8) Includes hot water production.

Comparison of indata:

Table A3-7 to A3-9 presents selected results from relevant studies dealing with LCA comparison between conventional systems and source separation systems similar to the systems studied in the present study.

Table A3-7. Comparison of used indata for selected parameters of blackwater.

Blackwater	Remy (2010)	Hillenbrand (2009)	Meinzinger (2010)	Wiersma (2014) [1]	Thibodeau (2014) [2]	Present study	Unit
Toilet flush water	24-36 (conv.) 5.2-24 (sep.)	-	49.0±39.6	13.7	18 (conv.) 5 (sep.)	14.0 (conv.) 9.0 (sep.)	L cap^{-1} d^{-1}
Dry mass	105	95	63.2±41.3	-	-	73.1	g cap^{-1} d^{-1}
COD	50	70	51.5±31.1	130	42.1	72.6	g cap^{-1} d^{-1}
Nitrogen	11.5	11.9	10.7±2.7	14.1	7.5	12.5	g cap^{-1} d^{-1}
Phosphorus	1.5	1.5	1.5±0.6	1.52	1.5	1.4	g cap^{-1} d^{-1}
Heavy metals							
Cd	0.0202	0.011	-	-	0.01	0.01	mg cap^{-1} d^{-1}
Cr	0.03	0.03	-	-	0.04	0.03	mg cap^{-1} d^{-1}
Cu	1.55	1.2	-	-	1.6	1.00	mg cap^{-1} d^{-1}
Hg	0.0204	-	-	-	0.01	0.01	mg cap^{-1} d^{-1}
Ni	0.24	0.08	-	-	0.2	0.07	mg cap^{-1} d^{-1}
Pb	0.03	0.04	-	-	0.04	0.02	mg cap^{-1} d^{-1}
Zn	10.25	11.0	-	-	7.6	8.97	mg cap^{-1} d^{-1}

1) Includes kitchen waste. 2) Only considers 65% of generated blackwater.

Table A3-8. Comparison of used indata for selected parameters of greywater.

Greywater	Remy (2010)	Hillenbrand (2009)	Meinzinger (2010)	Wiersma (2014)	Thibodeau (2014)	Present study	Unit
Flow	80	-	105	71	150	130	kg cap^{-1} d^{-1}
Dry mass	120	65	59.5±33.3	-	-	54.5	g cap^{-1} d^{-1}
COD	60	46	47.7±21.3	45	82.9	48	g cap^{-1} d^{-1}
Nitrogen	1.3	1	1.0±0.4	1.14	1.9	1.18	g cap^{-1} d^{-1}
Phosphorus	0.5	0.5	0.5±0.3	0.93	0.5	0.52	g cap^{-1} d^{-1}
Heavy metals							
Cd	0.2	0.08	-	-	0.19	0.03	mg cap^{-1} d^{-1}
Cr	3	2.01	-	-	2.55	1.00	mg cap^{-1} d^{-1}
Cu	20	6.5	-	-	29.0	6.82	mg cap^{-1} d^{-1}
Hg	0.02	-	-	-	0.03	0.004	mg cap^{-1} d^{-1}
Ni	2	1.6	-	-	1.9	1.23	mg cap^{-1} d^{-1}
Pb	3	3.0	-	-	2.55	0.95	mg cap^{-1} d^{-1}
Zn	46	23.3	-	-	66.8	9.95	mg cap^{-1} d^{-1}

Table A3-9. Comparison of used indata for selected parameters of food waste.

Food waste	Remy (2010)	Hillenbrand (2009)	Meinzinger (2010)	Witeveen Bos (2014)	Thibodeau (2014)	Present study	Unit
Wet weight	0.20	0.205	0.186±0.075	-	0.26	0.195	kg cap^{-1} d^{-1}
Dry mass	50	82.2	44.0±7.7	-	-	68.2	g cap^{-1} d^{-1}
TOC	13	-	15.3±4.5	-	107.6[1]	29.6	g cap^{-1} d^{-1}
Nitrogen	0.9	1.37	1.0±0.3	-	0.9	1.57	g cap^{-1} d^{-1}
Phosphorus	0.2	0.33	0.2±0.1	-	0.2	0.27	g cap^{-1} d^{-1}
Heavy metals							
Cd	0.01	0.008	-	-	0.01	0.01	mg cap^{-1} d^{-1}
Cr	0.5	-	-	-	0.50	0.34	mg cap^{-1} d^{-1}
Cu	1	0.77	-	-	1.0	1.36	mg cap^{-1} d^{-1}
Hg	0.01	-	-	-	0.01	0.0006	mg cap^{-1} d^{-1}
Ni	0.2	0.11	-	-	0.2	0.20	mg cap^{-1} d^{-1}
Pb	0.6	-	-	-	0.60	0.68	mg cap^{-1} d^{-1}
Zn	7.3	2.52	-	-	7.3	1.74	mg cap^{-1} d^{-1}

1) Value is for COD.