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To separate or not? A comparison of wastewater management systems for the new city district of Hiedanranta, Finland

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ABSTRACT

In this study, life cycle assessment (LCA) and life cycle costing (LCC) methods were applied for the new city district of Hiedanranta, where source-separating sanitation systems are being considered. Two source-separating systems were compared to the conventional sanitation system with a centralized wastewater treatment plant (WWTP). With a separating system, three to 10 times more nitrogen could be recovered compared to the conventional system. If the nutrient potential of the reject water of the sludge digestion were to be utilized, the recovery rate would be even higher. For phosphorus, the recovered amount would be at the same level for all the alternatives. However, the plant availability of phosphorus is higher in separating systems. Based on the environmental impacts of separating systems with improved nutrient recovery, the climate and eutrophication impacts could be reduced, but the acidification impact may be higher. However, the actual climate benefits depend on how the avoided emissions will be realized, which is highly dependent on the policy and decision-making processes in the society. The life cycle costs of the alternative source-separating systems are higher at current prices. Source-separating sanitation produces new recycled nutrient products of human origin that contain fewer contaminants and could therefore be more easily accepted for end use when certain boundary conditions are met.

1. Introduction

In 2020, 62% of the population living in urban areas globally were using safely managed sanitation services (WHO & UNICEF, 2021). Safely managed sanitation services keep the environment clean and maintain the health of citizens, which has been the priority in urban sanitation systems for years. Therefore, the focus in wastewater treatment has been on efficient nutrient and organic matter removal—not their recovery. However, the increasing need for nutrients such as phosphorus and nitrogen exists in food production. The depletion of natural resources and the challenges of tackling climate change have led to a reconsideration of the sustainability of sanitation and wastewater treatment systems. Nowadays, wastewater is increasingly seen as a valuable source of nutrients, energy, and an additional source of water (Van der Hoek et al. 2002, 2016; Sutton et al., 2011; Salgot and Folch, 2018; Bisschops et al., 2019; Rodriguez et al., 2021). However, the implementation of new sustainable sanitation solutions, such as source-separating systems, is thus far limited, although several studies have shown benefits compared to conventional systems (Tidåker et al., 2006; Remy and Jekel, 2008; Spångberg et al., 2014; Kjerstadius et al. 2015, 2017; Malila et al., 2019).

One of the challenges of centralized sewer systems is that they are based on mixing and transporting all wastewaters together, resulting in a mixture of nutrients, organic matter, and various chemical compounds and pollutants. Mixing and diluting the nutrient-rich black water from toilets with flushing water, gray water from washing and other wastewater, e.g., from industry, dilutes wastewater and makes it more difficult to recover nutrients at the treatment plant. It also increases the spectrum of contaminants in sewage sludge (Rogers, 1996; Kuster et al., 2005; Díaz-Cruz et al., 2009) and discharge water.

Nutrient recovery is technically easier in systems where nutrients are not diluted with gray water and are therefore at higher concentrations. Source separation offers one option to recover nutrients effectively and retains the fertilization characteristics of each fraction (Wielemaker et al., 2018; Viskari et al., 2021). Furthermore, source separation of toilet effluents can be accomplished either by urine separating toilets or

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by collecting all toilet effluents in a single fraction (black water). In this paper, the focus is on separating low-flush and vacuum toilets because they are more suitable for urban contexts than dry toilets. Collecting and treating black water separately from gray water would allow for a safer and more efficient, i.e., containing less harmful substances, the circulation of nutrients, and the production of energy at the same time (Kjerstadius et al., 2015). When black wastewater is collected separately from gray water, it may be useful to treat household biowaste together with toilet wastewater, especially in urban areas, in order to maximize local energy production (Kjerstadius et al., 2015; Skambraks et al., 2017; Stowa, 2018; Gomez et al., 2020; Xu et al., 2021).

In European countries such as Sweden, the Netherlands and Germany, local source-separating pilot systems are already in use and new large-scale pilot areas are being planned for urban areas (Kvarnström et al., 2006; Stowa, 2014; Skambraks et al., 2017; Lennartsson and Kvarnström, 2017; Lennartsson et al., 2019; Gomez et al., 2020). There are, however, challenges to tackle before source separation can take a bigger role in sustainable urban planning. The implementation of source separation in urban areas is hindered by several factors related especially to administrative issues, unclear legislation, stakeholder responsibility, and inflexible organizational culture (Skambraks et al. 2014, 2017; Lennartsson et al., 2019; Lennartsson and Kvarnström, 2017; Lehtoranta et al., 2021; McConville et al., 2017). Decision-making in sanitation planning is also complex, comprising trade-offs between sociopolitical, environmental, technical, and economic factors (Bao et al., 2013).

As water services are rather conventional and rigid in terms of change (Heino, 2016), education and interaction between research, city planners, consultants, and water services are needed (Skambraks et al., 2017; Lennartsson et al., 2019). In addition, the renovation of long-lifetime urban infrastructure, such as wastewater networks, is a slow process that requires a change in both the technical solutions and the institutions that guide them (Frantzeskaki and Loor-Bach 2010). Urban living labs (ULL) and experimental governance have recently been introduced in cities as means and sites to enhance sustainability transition, including in infrastructure sectors (Kronsell and Mukhtar-Landgren, 2018; Von Wirth et al., 2019; Voytenko et al., 2016). However, the gap between experiments and institutional planning is deep and cities are not exploiting the full strategic potential of ULLs to steer transition (Bulkeley et al., 2019; Särkilahti et al., 2021).

So far, there are no urban-scale source-separating sanitation systems implemented in Finland, but one is being studied in the future city district of Hiedanranta, in the City of Tampere. As the city planners are now drawing up the guidelines for future urban housing in Hiedanranta, this study focused on producing preliminary information for the planners and policymakers. In this paper, the aim was to compare the nutrient and energy balances, life cycle environmental impacts, and economic aspects of two simplified source-separating sanitation systems (urine and black water separation) and conventional sanitation systems in Hiedanranta. The potential and possibilities of source-separating sanitation is examined mainly from the perspective of nutrient recovery, climate change, and costs. The novelty of this study leans on the combination of the comprehensive environmental impact analysis and economic aspects of source separation of a new residential area. In addition, the analysis of the contribution of avoided climate impacts on total climate impacts shows the importance of careful planning and management of the system and supports the role of political decision making and urban planning.

2. Material and methods

2.1. Study area

Hiedanranta is an old factory area in Tampere, Finland, where a new, smart, and sustainable city district is being planned by taking a collaborative approach. The aims of the district development include resource efficiency and “higher production than consumption” (City of Tampere, 2019). Currently, the development of Hiedanranta is in its planning phase, and the City of Tampere has made it available as an urban living lab for novel solutions to sustainable city development (Engez et al., 2021). Among other sustainability initiatives, experiments in new sanitation and local nutrient recycling have taken place in the platform in several projects such as HIERAKKA—Hiedanranta as a nutrient cycle and public awareness development area; Leväsieppari—Growing algae biomass in source-separated urine and studying the possibilities of nutrient recovery; and NutriCity—Hiedanranta as a frontrunner in urban nutrient recycling (City of Tampere 2020; Särkilahti et al., 2021). Due to its industrial history, Hiedanranta lacks municipal infrastructure, e.g., only a few buildings are connected to the sewer network and central WWTP, which makes it a tempting site for testing source-separating sanitation systems. Successful innovations hold the potential to be scaled up when the new city district and its infrastructure are built.

2.2. Scenarios and system boundaries

The studied scenarios were formulated in close cooperation between researchers and urban planners of the Hiedanranta district. The aim was to make scenarios that are realistic to implement in this case area. Thus, mature technologies were chosen; for example, the nearby WWTP was utilized for gray water treatment.

In this study, source-separating sanitation systems where toilet waters are collected separately from gray waters were compared to a conventional system (Reference system) where all wastewaters are collected and transported to a centralized WWTP (Sulkavuori WWTP) using environmental and economic analysis. The analysis covered all future wastewater generated in the Hiedanranta district with 26,000 inhabitants and 6,510 jobs after all the planned houses are built. Alongside the Reference system, two scenarios for source separation were developed: A) black water separation with vacuum toilets, and B) urine separation with separating water-flush toilets (Fig. 1).

In the Reference system and the Scenarios, either all or part of the generated wastewater fractions were assumed to be treated at Sulkavuori WWTP and the surplus sludge that was generated as a by-product digested and composted. Sulkavuori WWTP is currently under construction and will be introduced in 2025. The WWTP represents the latest technology but no efforts have been made to improve nutrient recycling. The reject water from anaerobic digestion (AD) was assumed to be recycled back to Sulkavuori WWTP in accordance with current Finnish practices. It was assumed that the energy produced in AD would be utilized as heat and electricity at Sulkavuori WWTP, and the composted sludge would be utilized for field application, with its nutrients replacing the use of mineral fertilizers.

In the separating systems, gray water (Scenario A) or gray water and feces (Scenario B), were assumed to be treated at Sulkavuori WWTP as in the Reference system. In Scenario A, black water treatment was assumed to be carried out at a local AD plant in Hiedanranta. It was also assumed that the reject water from the Hiedanranta AD plant would be circulated back to the AD unit in accordance with current practices in Finland. It was assumed that the produced biogas at Hiedanranta would be upgraded to transport fuel and replace the use of fossil fuels. Digestate was assumed to be used as fertilizer and applied to fields. In Scenario B, urine was assumed to be hygienized in the basement of block houses for six months and then collected and transported for field application, replacing mineral fertilizer.

In addition to the emissions during operation, the construction of the infrastructure was also included in the study (e.g., pipelines, pumping stations, facilities, equipment, and buildings). For infrastructure, the total life span was assumed to be 50 years. End of life was excluded from the study. The Scenarios studied were developed in close cooperation between researchers and urban planners.

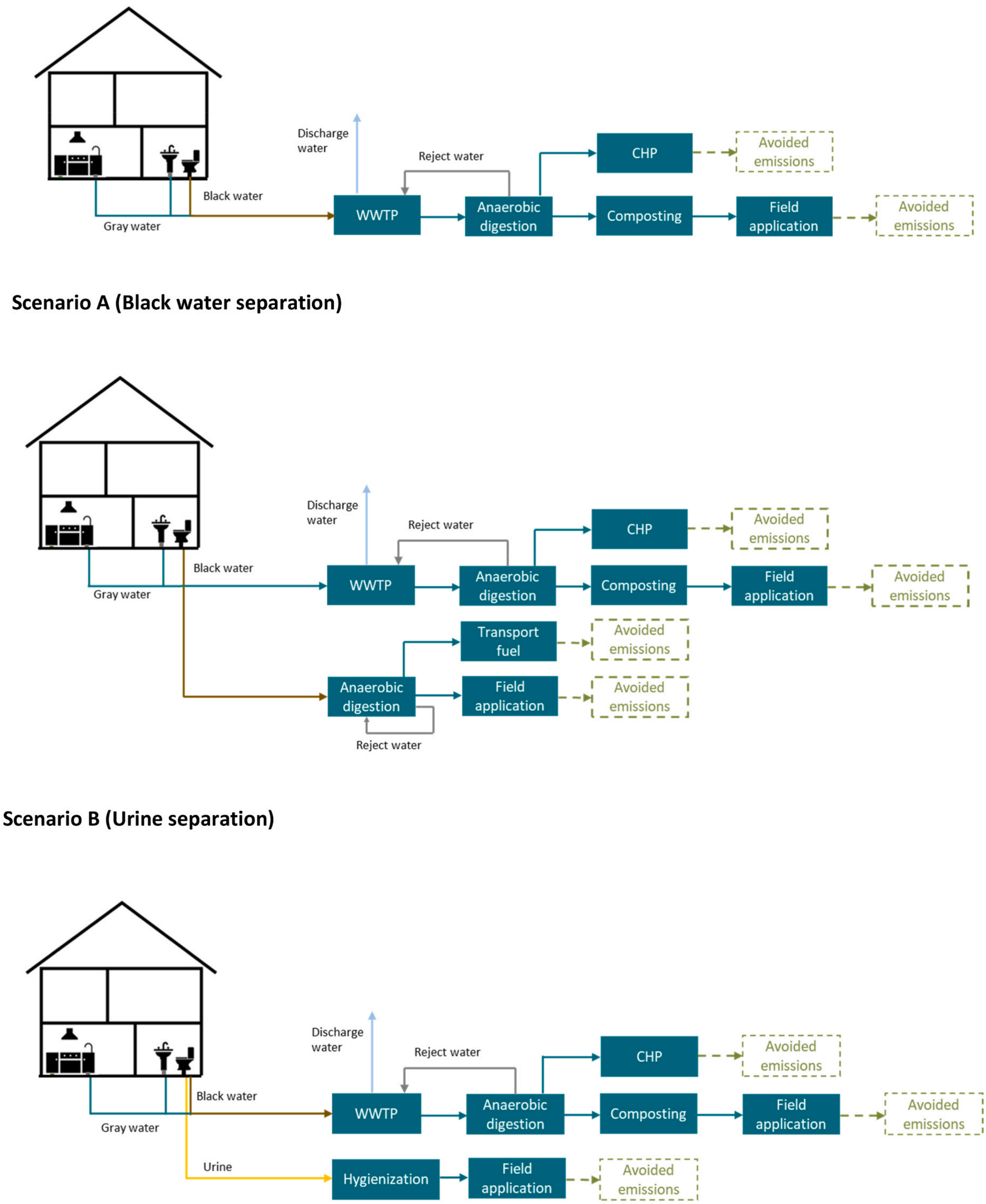

Fig. 1. Flow charts of the Reference system and Scenarios A and B. Dashed green boxes refer to avoided emissions in energy production and field application. Separated black water and urine are treated locally at Hiedanranta. Other operations (such as the WWTP) are located outside Hiedanranta. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

2.3. Environmental analysis

2.3.1. Method specification

Nutrient and energy balance calculations were based on a bottom-up mass balance approach, which is a common method used to assess engineering processes and has been used in several applications regarding nutrient recovery from wastewaters (Tervahauta et al., 2013; Cai et al., 2020).

The environmental impacts were analyzed by carrying out a life cycle assessment (LCA) (International Organization for Standardization (ISO), 2006a, International Organization for Standardization (ISO), 2006b). The goal of LCA was to compare the environmental performance of the current and future sanitation systems in Hiedanranta and analyze the impact of improved nutrient recovery and reuse. Therefore, the chosen approach was a consequential life cycle assessment (CLCA) according to Heimersson et al. (2019). The approach in CLCA is to describe how the impacts will change as a consequence of change in action (Ekvall and Weidema, 2004; Finnveden et al., 2009; Ekvall et al., 2016). Therefore, the impact of alternative sanitation systems in nutrient recovery and energy production (avoided emissions) are considered and alternative scenarios for avoided emissions are studied.

For the emissions, 100 years was chosen as the life span, which is typically used in LCA. The life cycle impacts analyzed were climate change, freshwater eutrophication, and acidification. The wastewater (Table 1), including nutrients and organic matter, produced by the planned 26,000 inhabitants and 6,510 jobs (including schools and day care) per year was chosen as a functional unit. For impacts on freshwater eutrophication, Finland-specific characterization factors (for P, N and NH_3) were used (Seppälä et al., 2004). For climate change and acidification, the ReCiPe Midpoint H method was used and completed with the updated global warming potential characterization factors for CH_4 and N_2O (IPCC, 2014). No normalization or weighting was used. The environmental impacts were calculated using Microsoft Excel and Simapro software.

2.3.2. Life cycle inventory and data used

The primary data was collected mainly from the literature, but also from urban plans, e.g., the master and process plans of Sulkavuori WWTP (Ramboll Oy, 2015), as well as the environmental permit application (Pöyry Oy, 2017) and the environmental impact assessment report (Tampereen Vesi Oy, 2012). The data was supplemented by negotiations with urban planners of the City of Tampere and expert assessments made by the consultant involved in the Sulkavuori WWTP design (Pöyry Environment Oy, currently AFRY Finland Oy). For some input data, e.g., air emissions and sludge production, the emissions, operation, and environmental permit data of Helsinki Viikinmäki WWTP, which is of a similar size, was utilized (HSY, 2018; AVI, 2015). The secondary data was obtained mainly from ecoinvent (Swiss Centre for Life Cycle Inventories, 2007). The data used in the calculation is presented in Tables 1 and 2.

A basic assumption of the amount of nutrients (nitrogen and phosphorus) and organic matter (BOD_7) produced by one person per year was: 5 kg N/person/a, 0.75 kg P/person/a, and 18.25 kg BOD_7/person/year (Ministry of the Environment, 2011; Udert et al., 2006; Weckman, 2005). Accordingly, approximately 82% of the N originates from urine, 10% from feces, and the rest from gray water. For phosphorus, assumptions were 53% from urine and 28% from feces, and for BOD_7 10% and 30%. Water consumption was assumed to be 120 l/person/day, consisting of 100 l of gray water, 1.15 l of urine, 0.11 l of feces, and the rest flush water (Malila et al., 2019). In addition, the following assumptions for home and workplace occupancy rates were made: 67% of the wastewater from toilets and its use was expected to be generated at home and 33% at the workplace, and 90% and 10% of the gray waters, respectively. In addition, in Scenario B, the separation efficiency of the urine separating toilet was estimated to be 85%, i.e., 15% of the separated urine was assumed to end up with the flush water at Sulkavuori WWTP.

According to the data, the treatment efficiencies of the WWTP for nitrogen, phosphorus, and BOD were 80%, 98%, and 99%. All the calculations, regarding emissions, energy consumption and sludge production of the WWTP, considered phosphorus, nitrogen, BOD loads and input flow of each Scenario (Table 1). Based on the calculations, the nutrient ratios were sufficient for biological growth in the Scenarios when the optimal ratio for biological growth (BOD: N: P = 100: 5: 1) was used. The amount of primary and bio-sludge generated at the WWTP was assumed to be constant due to simplification. The need for phosphorus precipitation differed between the Reference system and the Scenarios resulting in differences in the amount of chemical sludge. The amount of chemical sludge produced was 27% and 58% in the Scenarios A and B, compared to the Reference system.

The methane yield from AD was assumed to be 325 m^3 CH_4/ton VS-based (Järvinen and Rintala, 1996; Einola et al., 2001; Davidsson et al., 2007; Luostarinen et al. 2008, 2011). Methane emissions from digestion (both Sulkavuori and local plant) were calculated based on the Helsinki Viikinmäki WWTP data (HSY, 2018). The electricity and heat consumption in the Scenarios were calculated by relating their BOD loads to the Reference system. The same calculation method was also used for other operating parameters of wastewater treatment. For example, the phosphorus precipitating chemical was assumed to be proportional to the phosphorus load. Table 2 shows the input operating data of the

Table 1
Common assumptions calculated in the study regarding the Sulkavuori WWTP's overall wastewater loads, energy consumption, and sludge production in Reference system and Scenarios A & B.

Parameter	Reference system	Scenario A	Scenario B
Loads			
Input flow, 10^3m^3/a	1015	878	965
Nitrogen load, t N/a	99	9	31
Phosphorus load, t P/a	15	4	9
Organic load, t BOD/a	406	263	376
Energy			
Energy consumption, electricity, kWh/m^3	0.46	0.35	0.45
Energy consumption, heat, kWh/m^3	0.39	0.29	0.38
Self-sufficiency in electricity, %	75	79	68
Self-sufficiency in heat, %	77	81	70
Sludge			
Sludge production, kg TS/m^3 (TS 100%)	0.45	0.35	0.39
Digestate (dry fraction), kg TS/m^3 (TS 100%)	0.18	0.14	0.16
Digestate (dry fraction), kg VS/m^3 (TS 100%)	0.11	0.09	0.10
Reject water (liquid fraction of digestate), kg TS/m^3 (TS 100%)	0.12	0.10	0.11
Reject water (liquid fraction of digestate), kg VS/m^3 (TS 100%)	0.06	0.05	0.05

Table 2
The common assumptions calculated in the study for the Reference system and Scenarios regarding the wastewater management and processing of sludge.

Parameter	–	–
Emissions to air (Sulkavuori WWTP)		
Nitrous oxide, kg N_2O/kg N	0.022	
Ammonia, kg NH_3/kg N	0.00015	
Nitrogen oxides, kg NO_x/kg N	0.00062	
Methane, kg CH_4/kg BOD	0.002	
Consumption of chemicals (Sulkavuori WWTP)		
Ferrous sulfate, kg/kg P	14.4	
Lime, kg/kg P	3.8	
Methanol, kg/kg N removed	0.64	
Polyaluminumchloride, kg/kg P	0.11	
Polymer, kg/m^3	0.00074	
AD plant (Sulkavuori and Hiedanranta)		
Energy consumption, electricity, MWh/t TS		0.13
Energy consumption, heat, MWh/t TS		0.28
Energy production, electricity, MWh/t TS		0.90
Energy production, heat, MWh/t TS		0.94
Methane emissions from process, % of VS		1.3
Methane emissions form CHP process, % of CH_4 produced		1.5
Methane emissions from biogas upgrading into transport fuel, % of CH_4 produced		3
Polymer, kg/m_3	0.0099	
Composting of the sludge after Sulkavuori AD-plant		
Energy consumption, electricity, MWh/t TS	0.07	
Methane emissions CH_4–C, % of VS	3	
N_2O–N		
1% of tot N		
Nitrous oxide emissions N_2O–N, % of Ntot	1	
Ammonia emissions, NH_3–N, % Ntot	24	
Support material, peat, t/t TS	0.6	
Support material, wood chips, t/t TS	2.5	

WWTP, AD, and composting.

Methane emissions from biogas CHP use and upgrading to transport fuel were based on literature (Liebetrau et al. 2010, 2013; Poeschl et al., 2012; Adams et al., 2015; Jacobs et al., 2017; Reinelt et al., 2017). The operation of the local AD plant was assumed to be optimized with the residence time of black water, so no methane emissions from the storage of the digestate were expected to occur. The digestate from Sulkavuori WWTP was not assumed to be stored, but composted. For the composting, emission factors and process data from Myllymaa et al. (2008) and Manninen et al. (2016) were applied. Emissions from transportation were estimated based on LIPASTO (VTT, 2017) and the ecoinvent database (Swiss Centre for Life Cycle Inventories, 2007).

Regarding to avoided emissions, electricity produced with biogas was assumed to replace Finnish electricity mix. In addition, for heat production, biogas was assumed to replace heat produced by natural gas (39%), oil (5%) and wood (56%). For the substitutions of mineral fertilizers, total soluble nitrogen and 60% of total phosphorus of recovered nutrients were considered according to Finnish environmental compensation system. Emissions from the avoided processes were calculated based on the ecoinvent database (Swiss Centre for Life Cycle Inventories, 2007).

The gaseous emissions (ammonia, nitrous oxide) from storage and the field application of nutrients were calculated based on international emissions calculation guidelines for animal manure (EMEP/EEA, 2016; Grönroos et al., 2017) and IPCC et al. (2006) guidelines.

For leaching (field application of recycled nutrients), emission factors for nitrogen (N) and phosphorus (P) from manure field application developed in the Baltic Manure project (INTERREG) (Grönroos et al., 2013a; Grönroos et al., 2013b) were used.

2.4. Economic analysis

The economic analysis was carried out by applying the Life Cycle Costing (LCC) method. LCC does not have any widely established framework or commonly agreed methodology (Heijungs et al., 2013) but it is often used as a decision-making tool for wastewater management (e.g., Korpi and Ala-Risku, 2008; Muhammad et al., 2021). Life cycle cost is, in simplified terms, the cost an activity incurs during its life cycle. Costs at different stages of the life cycle, including investment, operating and maintenance, and depreciation costs, are converted using the present value method at a given point in time, usually at the time of purchase, in cash using the discount rate and usually taking inflation into account.

In this study, the streamlined LCC was carried out by using the same assumptions and system boundaries as in the environmental analysis. The calculations took into account the necessary investments and operating costs in each Scenario. The investment cost included facilities, networks and pipes, toilets, and lost building land, if applicable. The operating costs included costs related to various stages, e.g., pumping, wastewater treatment, digestion, composting, storage, transportation, and application of the end product to the field, such as energy and chemical costs. The cost benefits included produced energy and nutrients, as well as reduced water consumption. The cost benefits were taken into account, when applicable. However, it should be noted that with regard to water consumption, only differences in the consumption due to the use of toilets were taken into account, which is reflected in savings in water consumption in Scenarios A and B.

Cost data from Sulkavuori WWTP was based on the same input data as the LCA, including the master and process plans of Sulkavuori WWTP (Ramboll Oy, 2015), and expert assessments made by the consultant involved in the Sulkavuori WWTP design (Pöyry Environment Oy, currently AFRY Finland Oy). A market price of €0.1/kWh was used for energy and an expert estimate of €6/kg of precipitated phosphorus for chemicals used in the WWTP. Retail prices for toilets were €300, €880, and €450/piece for a conventional water closet, vacuum toilet, and separating water-flush toilet, respectively. As the apartments in the residential area were quite small, the average number of toilets was assumed to be 1.2 per apartment or office (in total approximately 16,000 apartments and 100 offices). In Scenario A, it was assumed that the vacuum toilet system would require one vacuum pump (€5,000/piece) per 10 toilets. In Scenario B, it was assumed that each building has an average of two urine hygiene tanks (€4,000/piece). The remainder of the prices for required pipes, equipment, and HVAC accessories, etc. were based on average data from suppliers, expert estimates, and Pöyry Environment Oy's FORE database. In addition, it was assumed that in Scenario A the price of the lost building land due to the AD plant to be built in the Hiedanranta area would be €1,300,000. When calculating the cost benefits of nutrients, the lowest reference prices for fertilizers were used, i.e., €0.73/kg for nitrogen and €1.47/kg for phosphorus (Luke, 2018). Compensation due to the decrease in water consumption, in Scenarios A and B, was calculated using the consumer prices of the local water company (Tampereen Vesi Oy), which was €3.55/m^3. The list price of the biogas market leader (Gasum Oy) of €1.45/kg was used to calculate the fuel credits. All costs were based on 2018 data.

A 50-year time period, a discount rate of 3.0%, and inflation at 1.5% were used based on the cost of living index and the consumer price index (Consumer price index). The basic assumption for the service life of facilities and sewers was 50 years, and 20 years for pumping stations, 20 years for toilet seats, 15 years for vacuum pumps (Scenario A), 20 years for urine hygiene tanks (Scenario B), and 25 years for urine pipes inside buildings (Scenario B).

3. Results and discussion

3.1. Environmental analysis

3.1.1. Nutrient and energy balances

According to the nutrient balance results (Fig. 2.), the recovery potential is several times higher in the source-separating alternatives. Within the Reference system, less than a tenth of the nitrogen is recovered, while the rest of the nitrogen ends up in water bodies and the air as a result of biological nitrogen removal. Also, most of the recovered nitrogen in the composted AD sludge is in organic form and therefore slowly becomes available for plants (about 50%). In Scenario A (black water separation), the amount of nitrogen recovered is more than double compared to the Reference system, since most of the nitrogen ends up in the reject water at the local AD and is recycled back to digestion. If the reject water (both in the Reference system and the scenarios) were utilized as such or the nutrients were recovered by implementing new technologies, the recovery of nitrogen would be greater. In Scenario B (urine separation), it is possible to recover more than half of the nitrogen contained in the wastewater (see Fig. 3).

In the Reference system, all the phosphorus recovered is chemically bound, which weakens plant availability. With source-separating systems, most of the phosphorus can be recovered in a more valuable and plant-useable form, although there is no significant difference in the total amounts. In Scenario A, more than half of the phosphorus could be recovered without utilizing the potential of the reject water. In Scenario B, the total phosphorus recovery is greatest (without considering the potential of the reject water), and half of it is in plant-useable form, the other half being chemically precipitated at Sulkavuori WWTP.

The results strongly support previous studies which suggest that the highest potential for nutrient recovery in the urban context could be achieved with source-separating systems, either by black water or urine separation (Kjerstadius et al. 2015, 2017; Wielemaker et al., 2018; Turlan, 2019). This study shows that even if the nutrient potential of reject water is not effectively utilized, nutrient recovery is still many times greater in both source-separating Scenarios than the Reference system.

The energy balance shows that there are no significant differences between the Scenarios and Reference system if the credits from energy production are included. The credits are highest in Scenario A, as well as

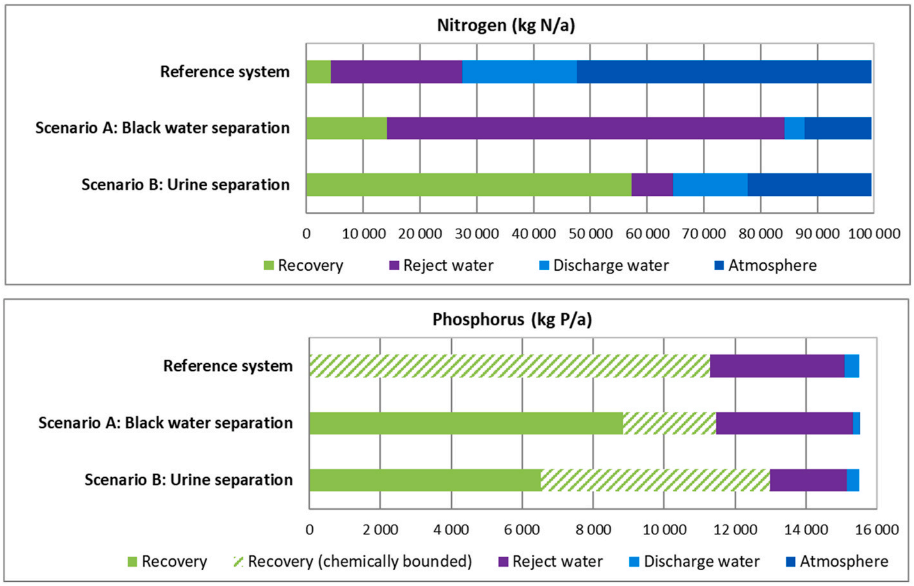

Fig. 2. Nutrient balances in the Reference system and Scenarios A & B.

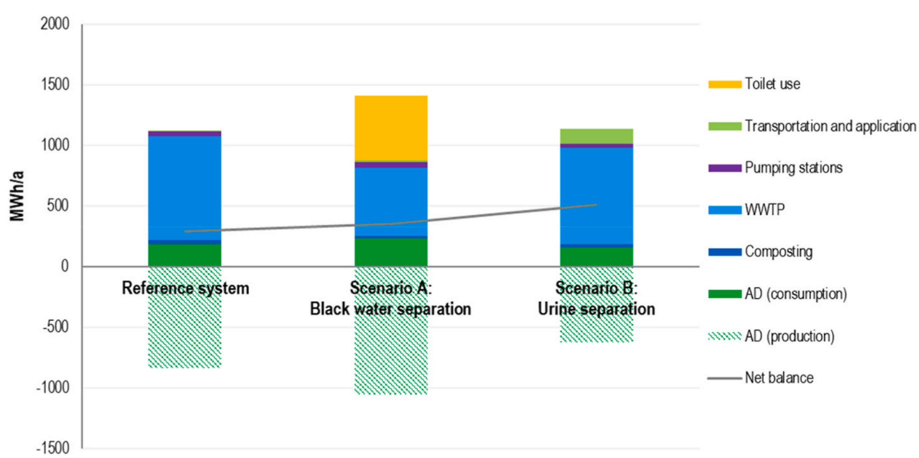

Fig. 3. Energy balances (MWh/a) in the Reference system and Scenarios A & B.

energy consumption. In total, Scenario B has slightly larger net energy consumption compared to Scenario A and the Reference system due to the lowest energy production at the AD unit. In Scenario A, energy production of AD is higher than the consumption of AD and the WWTP. However, biogas produced in the Reference system and Scenario B did not fully cover the energy needs of Sulkavuori WWTP and AD.

3.1.2. *Life cycle assessment*

According to the results, the climate change and eutrophication impacts are significantly lower in both source-separating scenarios (77% in Scenario A and 72% in Scenario B for climate change, 76% in Scenario A and in Scenario B 63% for eutrophication) compared to the Reference system (Fig. 4). Acidification, on the other hand, is higher in both Scenarios. The climate impact per person in the Reference system is 55 kg CO_2 eq./a, in Scenario A 13 kg CO_2 eq./a, and in Scenario B 16 kg CO_2 eq./a. The impacts of eutrophication and acidification per person in the Reference system and the Scenarios A and B, respectively, are 0.4, 0.1 and 0.2 kg PO_4 eq./a, and 0.2, 0.6 and 0.4 kg SO_2 eq./a.

Based on the LCA results, wastewater treatment in the WWTP causes a major part of the greenhouse gas and eutrophic emissions in the Reference system. The greenhouse gas emissions from the wastewater treatment plant are largely derived from nitrous oxide emissions, which are generated in the nitrogen removal process and the consumption of chemicals, for example. Through source separation, climate emissions are reduced by about a quarter and eutrophication by less than half, taking into account the recycled fertilizer products and the energy produced, and the avoided emissions related to their utilization, e.g., the production and use of mineral fertilizers and fossil fuels. The decrease is largely due to the reduced need for wastewater treatment, especially the energy-intensive removal of nitrogen from wastewater, but also to the more efficient management of nutrients, as they are not mixed and diluted with other wastewaters. As a result of the nutrient recovery from wastewater, the nitrogen load to Sulkavuori WWTP is reduced to less than one fifth and the phosphorus load to about one third, decreasing the climate emissions of wastewater treatment.

In the urine separating system, the transportation of hygienized urine has a significant effect on emissions, so further processing of the urine to reduce the volume on site should be considered. The processing of urine requires the implementation of new technologies and consumes more energy, but it can facilitate the utilization and acceptance of human-based nutrients (Simha and Ganesapillai, 2017; Viskari et al., 2021).

The results of this study support the findings of Kjerstadius et al. (2017). They found that the environmental benefits of source-separating

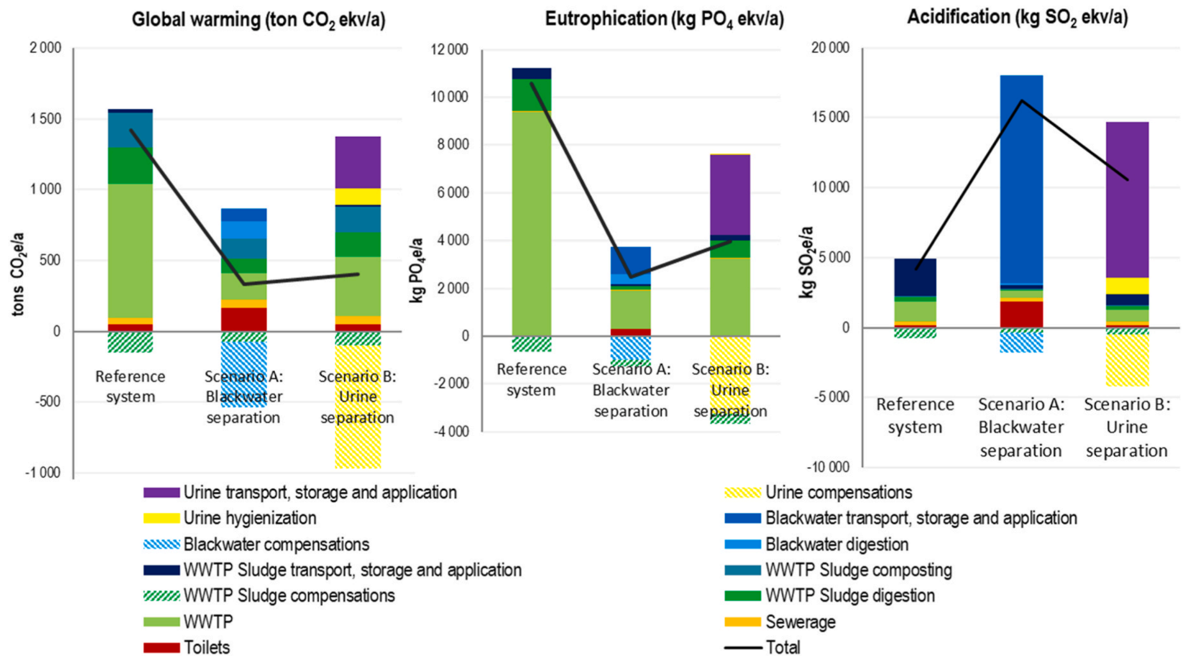

Fig. 4. Global warming, eutrophication, and acidification impacts of Reference system and Scenarios A and B.

systems in urban areas are based on improved nutrient recovery, but that climatic benefits can also be achieved compared to the conventional system, especially when food waste is collected and treated together with black water (Kjerstadius et al. 2015, 2017). Kjerstadius et al. (2015) concluded that the carbon footprint decreased, mainly due to increased biogas production, increased replacement of mineral fertilizers in agriculture, and less emissions of nitrous oxide from wastewater treatment. According to Remy and Jekel (2012), the energy benefits of mineral fertilizer substitution are relatively small compared to the energy recovered from organic matter in source-separating systems if household biowaste is considered along with wastewater. Therefore, it can be assumed that in Hiedanranta, the possibilities to utilize household biowaste produced in the area would result in additional climate benefits, but only if collecting biowaste and blackwater together would result in higher collection rate of biowaste in the area. It should also be kept in mind that if biowaste and wastewater are treated locally in densely populated urban areas, the risk of odors should be minimized. The volume of digestate and the possibilities to utilize it in nearby areas should also be taken into account in decision-making. Therefore, depending on the area, the processing of these biomasses should be carried out far enough away from the settlement.

The eutrophication impacts of the Reference system are higher due to the fact that the Sulkavuori WWTP discharge water ends up directly in the nearby lake. When source-separating sanitation is being applied, the load of Sulkavuori WWTP decreases. If nitrogen and phosphorus are considered separately, the eutrophication load of phosphorus is reduced by 24–32%, but the highest net decrease is caused by nitrogen (67–85%). The results are in line with other studies showing that source-separating systems have lower eutrophication impacts compared to conventional systems (Tidåker et al., 2006; Spångberg et al., 2014; Malila et al., 2019). However, the eutrophication impacts from field application are strongly dependent on several agricultural pressures and physical characteristics of the catchment areas (Dupas et al., 2015), and therefore LCA results should always be considered critically (Morelli et al., 2018).

The results show that the risk of acidification is multiple in both Scenarios, being highest in the Scenario A. This results from the risk of ammonia evaporation from digestate storage and field application. In Scenario B, multiple amounts of nitrogen are recovered from urine, but the urine is assumed to be stored in closed tanks and applied by deep injection to the fields, which contributes to reduced total emissions compared to Scenario A. However, the emission factors used for storage and field application may overestimate the gaseous nitrogen emissions of digestate and urine. The use of manure-based conservative values for black water digestate has led to similar results found by Thibodeau et al. (2014). The increased risk of acidification in source-separating systems has also been recognized in other studies (Tidåker et al., 2006; Spångberg et al., 2014). However, it should be noted that some studies suggest that the application of digestate by subsurface injection reduces ammonia emissions compared to mineral fertilizers (Riva et al., 2016). All in all, urine and black water require further processing and appropriate storage and spreading practices should be used to ensure that ammonia evaporation is kept to a minimum (Webb et al., 2005).

The impacts on soil and land use are typically excluded from LCA studies due to incomplete methods and uncertain impact assessments (Brandão et al., 2013; Petersen et al., 2013; Arzoumanidis et al., 2014; Soimakallio et al., 2015; Celestina et al., 2019). In general, nutrient recycling improves the level of environmental performance by reducing the need for mineral fertilizers and helping to restore soil organic matter, which improves soil structure and stimulates the activities of micro-organisms and reduces nutrient leaching (Lashermes et al., 2009; Roberts et al., 2010; Liang et al., 2017; Wiesmeier et al., 2019). However, the role of soil microbes in the carbon cycle is still poorly understood (Liang et al., 2017; Chenu et al., 2019). Furthermore, any changes in nutrient recovery as well as sludge processing and use affects the carbon balance. The effects of harmful substances on soil biota and carbon sequestering are also not well known. For example, some studies suggest ecotoxicity of digestate (Teglia et al., 2010; Pivato et al., 2016; Tigini et al., 2016), which may have effects on carbon sequestration of soil biota. However, more research is needed to include these effects in LCA studies.

3.1.2.1. Consequences of improved energy and nutrient recovery. The consequences of improved energy and nutrient recovery were examined more closely by mapping the climate impacts of alternative avoided processes for both the Reference system and the Scenarios. For the Reference system, it was calculated what the benefits would be if the digestate was utilized as such without composting. In Scenario A, it was studied how the impacts would change if the biogas from black water digestion were utilized as CHP instead of being utilized as a transport fuel. In addition, it was studied how the recovery and utilization of nutrients of the AD reject water from both the Sulkavuori and Hiedanranta AD plants would reduce the use of mineral fertilizers.

The results show that the assumptions made about avoided emissions have a significant effect on the results (Table 3). If the nutrients from reject waters were recovered more effectively, the climate impacts of wastewater treatment could be further reduced in all Scenarios. In addition, it is important to note that improved nutrient recovery in the Reference system would significantly increase the environmental performance of the Reference system. On the other hand, if the nutrients of the reject waters are recovered and utilized, it may in practice require the introduction of new technologies, which in turn often consume more energy, partially reducing climate benefits but producing more nitrogen fertilizers, thus replacing the use of mineral fertilizers.

Utilizing biogas as a transport fuel leads to higher emissions savings compared to CHP production as a result of lower compensation benefits in CHP production. However, it is important to note that avoided emissions from energy and substituted mineral fertilizers will not be achieved unless their use is reduced in the same proportion (IPCC, 2014). For example, if the ongoing reform of the national fertilizer legislation (Ministry of the Agriculture and Forestry, 2020) restricts the use of wastewater-based nutrients in Finland in the future, the benefits from the assumed replacement of mineral fertilizers will be lost. Therefore, the benefits realized might be less than the calculations suggest.

3.2. Economic analysis

The results of the streamlined economic analysis show that net life cycle costs are the lowest in the Reference system (Fig. 5). This is largely due to the fact that in source-separating systems, the investment costs associated with toilet systems are higher. In terms of toilet costs, prices are retail prices for which significant savings could be made through tendering and good planning. The double piping required in alternative systems also doubles the piping costs, but this does not play a significant role in the overall picture.

In the Scenarios, the cost of treatment in the WWTP is assumed to be proportional to the loads of organic matter and nutrients treated in each scenario. In practice, however, the WWTP is not obliged to reduce the unit price for wastewater treatment when the load decreases, for example only for gray water. A possible lower price must be agreed on a case-by-case basis between the WWTP and the area in question.

The annual net operating costs are clearly lower in the Scenarios compared to the Reference system. In fact, Scenario A, especially, is almost self-sufficient, i.e., the potential cost benefits cover the costs incurred. However, this requires the realization of all the achievable cost benefits. In the black water and urine separating systems, the biggest cost benefits arise from savings in water use, which account for 73% and 67% of the total cost benefits. Other cost benefits consist of nutrients and energy produced and are 7% and 19% of the total cost benefits in the black water separation in Scenario A and 31% and 3% in the urine separation in Scenario B. The annual cost benefits of the Reference system are less than a tenth compared to both scenarios.

Per person, the annual net cost is approximately €70, €138, and €85 in the Reference system and in Scenarios A and B, respectively. In euro terms, the differences in costs are significantly lower than the average

Table 3
The impact of avoided processes (default and alternative avoided processes) in avoided emissions in Reference System and Scenarios A & B.

Default avoided processes	Avoided emission (t CO_2 eq./a)	Impact on total emissions (%)	Alternative avoided processes	Avoided emission (t CO_2 eq./a)	Impact on total emissions (%)	Difference between default and alternative (%)
REFERENCE SYSTEM						
Digested sludge from Sulkavuori AD composted and used for fertilization	−54	−3%	Digestated sludge from Sulkavuori AD used for fertilization	−75	−5%	−1%
Biogas from Sulkavuori AD to CHP	−97	−6%	Default assumption	−97	−6%	0%
Reject waters from Sulkavuori AD not utilized			Reject waters from Sulkavuori AD used for fertilization	−226	−14%	−14%
In total	**−151**	**−10%**		**−398**	**−25%**	**−16%**
SCENARIO A: Blackwater separation						
Digested sludge from Sulkavuori AD composted and used for fertilization	−6	−1%	Digestated sludge from Sulkavuori AD used for fertilization	−8	−1%	0%
Digested sludge from Hiedanranta AD used for fertilization	−250	−29%	Default assumption	−250	−29%	0%
Biogas from Sulkavuori AD to CHP	−66	−8%	Default assumption	−66	−8%	0%
Biogas from Hiedanranta AD to transport fuel	−214	−25%	Biogas from Hiedanranta AD to CHP	−62	−7%	17%
Reject waters from Hiedanranta AD not utilized			Reject waters from Hiedanranta AD used for fertilization	−694	−80%	−80%
Reject waters from Sulkavuori AD not utilized			Reject waters from Sulkavuori AD used for fertilization	−21	−2%	−2%
In total	**−536**	**−62%**		**−1101**	**−127%**	**−65%**
SCENARIO B: Urine separation						
Digested sludge from Sulkavuori AD composted and used for fertilization	−19	−1%	Digestated sludge from Sulkavuori AD used for fertilization	−26	−2%	−1%
Urine used for fertilization	−869	−63%	Default assumption	−869	−63%	0%
Biogas from Sulkavuori AD to CHP	−81	−6%	Default assumption	−81	−6%	0%
Reject waters from Sulkavuori AD not utilized			Reject waters from Sulkavuori AD used for fertilization	−71	−5%	−5%
In total	**−969**	**−71%**		**−1047**	**−76%**	**−6%**

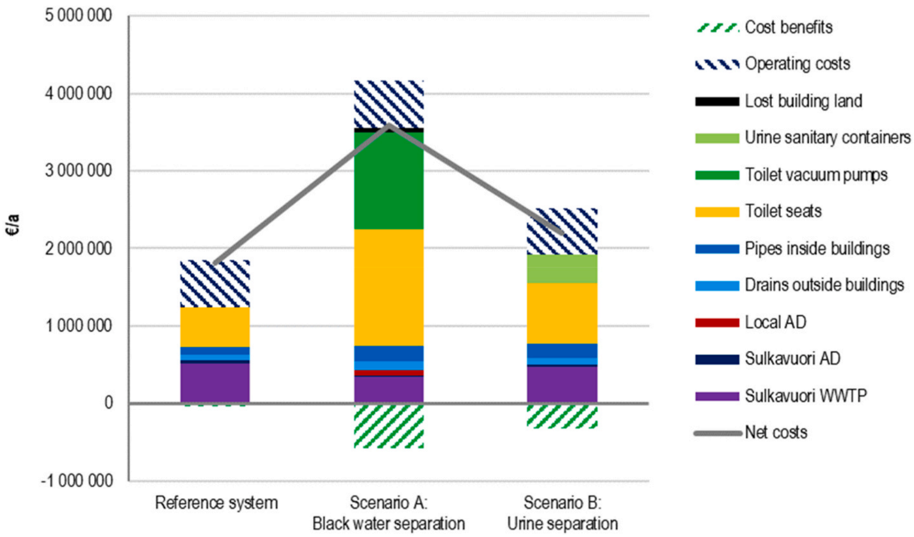

Fig. 5. The investment and operating costs of Reference system and Scenarios A and B (calculated on a yearly basis).

yearly cost of the total water consumption for one person (approximately €150/person/year with 120 l/person/day). It should be noted that the cost of water consumption for gray water was not included in the operating costs, as it was the same for all alternatives. Thus, the decrease in water consumption in Scenarios A and B has been taken into account as cost benefits.

Similar results have been obtained in several studies, indicating that operating costs are lower in source-separating systems compared to conventional ones. However, this requires that the value of the resources produced, such as energy and fertilizers, can be fully included in the calculation (Wood et al., 2015; Schoen et al., 2017; Xue et al., 2016; Vidal et al., 2019). However, there are also indications that investment costs might be higher in the separating systems (Lennartsson et al., 2009), which is in line with the results of this study. The background material of this study was highly indicative, resulting in uncertainties in the results. For example, the cost associated with toilet systems in the Scenarios are likely to be overestimated due to the lack of more detailed technical plans.

4. Conclusions

The nutrients and organic matter in wastewater are currently underutilized resources and their recovery could be improved in urban areas by implementing source-separating sanitation systems in housing. According to the study, separating systems could recover up to 10 times more nitrogen than a conventional system. By utilizing the nutrient potential of reject water from digestion, the recovery rate would be even higher. For phosphorus, the recovered amount would be at the same level in all the alternatives, but plant availability is higher in the source-separating systems because of major part of the phosphorus is not chemically precipitated.

The environmental impacts of improved recovery of wastewater nutrients clearly show that climate and eutrophication impacts could be decreased, but drawbacks in acidification impacts may occur. However, according to the results, the actual environmental benefits rely strongly on decisions made in planning and design of the system. After all, the actual impacts depend on how the avoidable emissions will be realized, which is highly dependent on policy- and decision-making in society. For example, the benefits of improved nutrient recovery and produced recycled nutrient products will not be realized if they do not replace inorganic, energy-intensively produced fertilizers. The same applies to the renewable energy produced. The full realization of the benefits usually requires the introduction of new policies and policy instruments as well as good planning and management.

At current market prices, the total costs of source-separating systems are higher than in conventional systems in new residential areas, according to this study. However, the market for separating systems is still marginal compared to the mainstream, which is reflected in their price levels. In the future, for example, water scarcity may trigger the need for alternative separating sanitation solutions, making them more widespread and likely lowering their prices. Moreover, source separating systems might be more attractive option in the context of urban renewal, especially when the cost of renovation of outdated sewer networks and WWTPs and the potentially emerging markets for recycled nutrients are considered in the total life cycle costs. Furthermore, the feasibility of a source-separating system depends on local conditions and technical choice.

In this study, gray water was assumed to be treated in the WWTP in all alternatives. The local treatment of gray water would have made the wastewater management scenarios fully local and would subsequently have changed the results. This kind of closed-loop system would be particularly interesting in remote locations with a local drinking water source and without a connection to centralized water and sanitation systems. The Hiedanranta district is located only 4 km from the city centre and is close to wastewater pipes. Therefore, the connection to centralized systems is (too) attractive.

Attempts to transfer knowledge from this study to practical urban planning have again revealed a gap between (informal) urban living lab projects and (formal) institutional planning. Implementing source separation at the urban scale requires major structural changes in infrastructure and practices, and the process of implementation of research results in practice is not straightforward. The technical solutions are available and ready to implement, but the biggest challenge is breaking up current business and service models. New sanitation solutions that support nutrient recovery also require legislative changes and acceptance in society. Source-separating sanitation can produce completely new recycled nutrient products of human origin, the utilization of which should be allowed under certain boundary conditions.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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