Doctoral programme in Sustainable Use of Renewable Natural Resources
Faculty of Agriculture and Forestry
University of Helsinki

TOWARDS CIRCULAR ECONOMY IN WASTEWATER MANAGEMENT

ENVIRONMENTAL IMPACTS, BENEFITS AND DRAWBACKS OF IMPROVED NUTRIENT RECOVERY AND RECYCLING BY SOURCE SEPARATION

Suvi Lehtoranta

DOCTORAL DISSERTATION

To be presented for public discussion with the permission of the Faculty of Agriculture and Forestry of the University of Helsinki, in Raisio Hall (ls B2), Forest Sciences Building, on the 17th of June, 2022 at 12 o’clock.

Helsinki 2022

Supervising professor

Associate professor Hanna Tuomisto
Helsinki Institute of Sustainability Sciences
Department of Agricultural Sciences
University of Helsinki
Finland

Preliminary examiners

Professor Mika Horttanainen
Department of Sustainability Sciences
Lappeenranta University of Technology (LUT)
Finland

Dr. Pernilla Tidåker
Department of Energy and Technology
Swedish University of Agricultural Sciences (SLU)
Sweden

Opponent

Associate professor Anna Mikola
Department of Built Environment
Aalto University
Finland

The Faculty of Agriculture and Forestry uses the Ouriginal system (plagiarism recognition) to examine all doctoral dissertations.

ISBN 978-951-51-8222-7 (softcover)
ISBN 978-951-51-8223-4 (PDF)

Unigrafia
Helsinki 2022

ABSTRACT

Due to the growing need for phosphorus and nitrogen in food production, more attention needs to be paid for the efficient recovery and safe recycling of wastewater nutrients and organic matter back to agriculture. Source separation of household wastewaters has emerged as an efficient way to recover these nutrients. This can be accomplished by collecting and treating nutrient-rich and nutrient-poor wastewater separately. The nutrients can be recovered in a plant-available form without being mixed with harmful substances from sources other than toilet water. In addition, nutrient recovery is technically easier when the nutrients are undiluted and thus present in higher concentrations.

The objective of this dissertation was to analyze the life cycle environmental impacts, advantages and drawbacks, of wastewater management to improve nutrient recovery by source separation in urban, peri-urban and rural areas. Moreover, the limitations of life cycle assessment (LCA) in assessing the environmental impacts of nutrient recovery and recycling were analyzed.

The results indicate a substantial increase in the nutrient recovery potential with source separation of wastewaters in urban, peri-urban and rural areas compared to conventional wastewater treatment systems. In urban areas, the source separation of wastewater would allow up to ten times higher nitrogen recovery compared to the conventional system. In rural areas, the source separation of wastewater would bring even greater benefits. Phosphorus recovery could be improved 3-5 times and nitrogen recovery over 30 times.

Moreover, improved recovery and recycling of nutrients by source separation would reduce the climate impact of wastewater treatment in urban areas by about half, but the climate impact in rural areas would remain at the same level. Source separation of wastewater would also reduce eutrophication of water bodies, especially in rural areas. Instead, acidifying emissions may increase.

However, the actual environmental benefits of improved nutrient recovery and recycling require the realization of avoided emissions, which rely strongly on the decisions made in the planning and design of the system and on the policies and decisions made in the society. LCA offers a good tool to support planning, decision making, and policy related to nutrient recycling. However, the LCA methodology still needs further development and accepted rules to take into account the impacts of organic matter in recycled nutrients.

Tackling the inefficiencies of nutrient recovery and recycling promotes the transition towards circular economy and carbon neutrality in wastewater management. Source separation of wastewaters offers one way to accomplish these. Source separation allows for more efficient nutrient recycling and supports the self-sufficiency of fertilizers. This requires that the nutrients are recovered and processed into safe end products. To realize the nutrient

potential and environmental benefits of the agricultural use of wastewater-based nutrients, policy support and careful planning from a life-cycle perspective are needed.

TIIVISTELMÄ

Ruuan tuotannolle välttämättömien ravinteiden, fosforin ja typen, kasvavan tarpeen vuoksi tulee kiinnittää enemmän huomiota jäteveden ravinteiden talteenottoon ja turvalliseen kierrätykseen. Kotitalouksien jätevesien syntypaikkaerottelu (erotteleva sanitaatio, jätevesien erilliskeräys) on todettu tehokkaaksi tavaksi ottaa talteen jäteveden ravinteita kasveille käyttökelpoisessa muodossa. Jätevesien erottelu voidaan toteuttaa keräämällä ja käsittelemällä erikseen ravinnerikkaat (käymälävesi) ja ravinneköyhät jätevedet (harmaa vesi). Samalla voidaan vähentää ravinteisiin päätyviä haitta-aineita. Lisäksi ravinteiden talteenotto on teknisesti helpompaa, kun ravinteet ovat suurempina pitoisuuksina.

Tämän väitöskirjan tavoitteena oli arvioida jätevesien ravinteiden talteenoton tehostamista jätevesiä erottelemalla, sen elinkaarisia ympäristövaikutuksia, etuja ja haittoja kaupunki- ja haja-asutusalueilla. Lisäksi työn tavoitteena oli analysoida elinkaariarvioinnin (LCA) menetelmällisiä rajoituksia ravinteiden talteenoton ja kierrätyksen ympäristövaikutuksien arvioinnissa.

Väitöskirjan tulokset osoittavat, että jätevesien erottelulla voidaan tehostaa ravinteiden talteenottoa huomattavasti sekä haja-asutusalueella että kaupunkialueilla tavanomaiseen käsittelyyn verrattuna. Kaupunkiympäristössä jätevesien erottelu mahdollistaisi jopa yli kymmenen kertaa suuremman typen talteenoton. Haja-asutusalueella jätevesien erottelulla saavutettaisiin vielä suurempi hyöty. Fosforin talteenotto voisi tehostua noin 3-5 kertaa suuremmaksi ja typen yli 30 kertaa suuremmaksi tavanomaiseen käsittelyyn verrattuna.

Tulosten mukaan ravinteiden tehokkaampi talteenotto ja kierrättäminen jätevesiä erottelemalla vähentäisi kaupunkialueilla jätevedenkäsittelyn ilmastovaikutuksia noin puoleen, mutta haja-asutusalueella vaikutukset pysyisivät arviolta samalla tasolla. Jätevesien erottelu vähentäisi myös vesistöjä rehevöittäviä vaikutuksia, erityisesti haja-asutusalueella. Sen sijaan happamoittavat päästöt voivat kasvaa.

Tehokkaammalla ravinteiden talteenotolla saavutettujen ympäristöhyötyjen toteutuminen edellyttää usein vältettyjen päästöjen (energia, ravinteet) toteutumista, joka on voimakkaasti riippuvainen sekä erottelevan järjestelmän suunnittelusta ja toteutuksesta että yhteiskunnassa toteutettavasta politiikasta ja päätöksenteosta. LCA-menetelmä soveltuu hyvin suunnittelun työkaluksi sekä päätöksenteon tueksi ravinteiden kierrätyksessä. Menetelmä vaatii kuitenkin vielä kehitystä ja yhteisesti sovittuja käytäntöjä, erityisesti kierrätyslannoitteiden sisältämän orgaanisen aineksen vaikutusten sisällyttämiseksi.

Jätevesien ravinteiden talteenoton ja kierrätyksen tehostaminen jätevesiä erottelemalla edistää vesihuollon kiertotaloutta ja hiilineutraaliutta.

Ravinteiden tehokkaampi talteenotto jätevesistä vahvistaa myös huoltovarmuutta. Talteen otettujen ravinteiden kierrätys takaisin maatalouteen kuitenkin edellyttää, että ravinteet jalostetaan turvallisiksi lopputuotteiksi. Ravinnepotentiaalin ja ympäristöhyötyjen saavuttaminen edellyttää poliittista tukea jätevesipohjaisten ravinteiden maatalouskäytölle sekä huolellista suunnittelua elinkaarinäkökulma huomioiden.

ACKNOWLEDGEMENTS

Just over a year ago, after years and years of hesitation, I began my doctoral studies in the AGFOREE-doctoral programme (Doctoral Programme in Sustainable Use of Renewable Natural Resources) in the middle of Covid-19 pandemic. Perhaps I thought writing the last two papers and the summary this year would have offered an opportunity to retreat into my own bubble for a moment amidst a hectic family life and endlessly continuous telecommuting. In practice, it was not that simple. After all, this year has been inspiring and intensive, both in good and bad. However, the journey of this dissertation is much longer and more complicated. After all, I am grateful that I had the courage to get this far.

This thesis has been compiled from several research outcomes of wastewater related projects carried out over ten years at the Finnish Environment Institute SYKE: MASU (2009-2011/Tekes), BIOUREA (2015-2016/Ministry of the Environment), NutriCity (2018-2020/Ministry of the Environment), NORMA (2018-2020/Ministry of the Environment), ON-SITE (Interreg/2019-2022) and the LCA-study commissioned by the City of Tampere in 2018. All these projects and their funding, as well as the Academy of Finland's project 341312 for funding the writing of this summary, are greatly acknowledged.

I would like to show my greatest gratitude to the co-author Riikka Malila, who introduced me in the first place to the interesting and diverse world of wastewater. During the joint years at SYKE, we shared true inspiration on the topic and created path for projects on wastewater nutrients. When Riikka decided to try her wings at the Ministry of the Environment, I began to consider doctoral studies more seriously in order to summarise the research outcomes obtained over the years.

I would like to sincerely thank my supervisor Assoc. Prof. Hanna Tuomisto for her flexibility and patience with my ambitious timetable, but also for her support, and encouragement during this year. Thanks also to Sari Kauppi and Ari Nissinen for being a member of my thesis committee and for supporting me on this journey. I gratefully acknowledge the time taken by the preliminary examiners, Prof. Mika Horttanainen and Dr. Pernilla Tidåker, to review the manuscript of my thesis. I appreciate the valuable and constructive comments they gave that helped me improve my thesis. Thanks also to Assoc. Prof. Anna Mikola that she took the time to be my opponent.

For me, it has been a privilege to work at SYKE all these years with the best colleagues and a warm atmosphere at the Centre for Sustainable Consumption and Production directed by Prof. Jyri Seppälä. There has always been a great support (or push) for doctoral studies, too. When I finally decided to hop on this train, despite of many ongoing projects and project preparations, time was arranged for the dissertation with the support and flexibility of Sampo Soimakallio.

When I started at SYKE in 2008, Laura Sokka introduced me to the LCA methodology, for which I am grateful. I was welcomed warmly, especially by the TTO-ladies Riina Antikainen, Katriina Alhola, Helena Dahlbo, Marja-Riitta Korhonen, Sirkka Koskela, Riikka Malila, Laura Sokka and Tuuli Myllymaa. While for some of us the roads have led elsewhere over the years, it has been a pleasure to keep in touch with you outside the workplace as well.

I feel privileged for having such amazing colleagues at SYKE. Thank you, Annika Johansson, for your endless positive energy and sense of humor. And thanks for taking the time to language check the summary. Thank you, Vuokko Laukka, for reading the first version of this summary and giving me feedback. Thank you, Jáchym Judl and Marja Salo, for your peer support and encouragement on this journey. Thank you, Päivi Fjäder and Lauri Äystö, for interesting discussions and giving me new perspectives.

Special thanks belong to one of my closest colleagues at SYKE, Juha Grönroos, who has always cheered me up and supported me. Juha has always got my back, being the person to discuss (or chat) with but at the same time he has challenged and pushed me forward.

Thank you, all the co-authors, especially Eeva-Liisa Viskari, Maarit Särkilahti, Vuokko Laukka, Brenda Vidal, Inga Herrman and Elisangela Heiderscheidt for your contribution and valuable comments to improve the manuscripts. Thanks also to my LUKE colleagues, especially Sari Luostarinen and Elina Tampio. We have worked together on many projects related to nutrient recycling and manure management, which has given me more perspective on writing this summary.

I want to thank all my dear friends 'vakkarit', for their endless support. The Porvoo 'jumppaporukka' is also highly acknowledged for the weekly crazy outdoor evening gym throughout the Covid-19 pandemic (still continuing). It really was one thing, that kept me going through the dark autumn and wintertime when everything else was prohibited. Special thanks belong to Laura Wiik, my soul sister, for the long walks and the discussions on the endless learning process of life and parenthood.

I want to thank my mother Tiina for her encouragement and help with the children during all these years. I also thank my father Pekka and his spouse Kristiina and my mother and father-in-law Tuija and Ilpo for their support and help during this journey.

Last, but not least, I want to thank my beloved family, my husband Ilkka and our sons Sampo and Tuomas. You all made sure I didn't dive too deep with this dissertation while working from home this year. You constantly reminded me of what is most important to me in the world. It's you.

Porvoo, May 2022

Suvi Lehtoranta

CONTENTS

LIST OF ORIGINAL PUBLICATIONS AND AUTHORS' CONTRIBUTION

This thesis is based on the following publications:

I	Lehtoranta, S., Vilpas, R. & Mattila, T. 2014. Comparison of Carbon Footprints and Eutrophication Impacts of Rural On-Site Wastewater Treatment Plants in Finland. *Journal of Cleaner Production 65: 439-446*
II	Malila, R., Lehtoranta, S. & Viskari, E-L. 2019. The Role of Source Separation in Nutrient Recovery - Comparison of Alternative Wastewater Treatment Systems. *Journal of Cleaner Production 219: 350-358.*
III	Lehtoranta, S., Malila, R., Särkilahti, M. & Viskari, E-L. 2022. To Separate or Not? A Comparison of Wastewater Management Systems for the New City District of Hiedanranta, Finland. *Environmental Research 15: 112764.*
IV	Lehtoranta, S., Laukka, V., Vidal, B., Heiderscheidt, E., Postila, H., Nilivaara, R. & Herrmann, I. 2022. Circular Economy in Wastewater Management – the Potential of Source-Separating Sanitation in Rural and Peri-Urban areas of Northern Finland and Sweden. *Frontiers in Environmental Science 10: 804718.*

The publications are referred to in the text by their roman numerals.

In *Paper I,* the LCA was performed by Suvi Lehtoranta. The data was jointly collected and analyzed with Riikka Malila (Vilpas). Tuomas Mattila supervised the work. The manuscript was co-written by all the authors.

In *Paper II*, the conceptualization of the work was made together with all the authors. The LCA was performed by Suvi Lehtoranta. Data was collected and analyzed by Suvi Lehtoranta and Riikka Malila. Riikka Malila was responsible on writing the manuscript. Suvi Lehtoranta and Eeva-Liisa Viskari co-writed and commented on the manuscript.

In *Paper III*, the conceptualization of the work was made together with Riikka Malila, Suvi Lehtoranta and Maarit Särkilahti. The data was collected and analyzed by Riikka Malila and Suvi Lehtoranta. Riikka Malila performed the

SFA, LCA and LCC and Suvi Lehtoranta supervised the work on LCA. Suvi Lehtoranta was responsible on writing the manuscript and analyzing the results with the assistance of Riikka Malila. Maarit Särkilahti and Eeva-Liisa Viskari commented on the paper.

In *Paper IV,* Suvi Lehtoranta performed SFA and LCA synthesis. Data was collected by Suvi Lehtoranta, Vuokko Laukka and Brenda Vidal. The conceptualization of the work was made by Suvi Lehtoranta, Vuokko Laukka, Brenda Vidal and Inga Herrman. Manuscript was co-written with Suvi Lehtoranta, Vuokko Laukka, Brenda Vidal and Elisangela Heiderscheidt. All authors commented on the paper.

ABBREVIATIONS

AD	Anaerobic digestion
ALCA	Attributional life cycle assessment
BOD	Biological oxygen demand
BW	Blackwater
BWS	Blackwater separation
CHP	Combined heat and power production
CLCA	Consequential life cycle assessment
CO_2	Carbon dioxide
CO_2eq.	Carbon dioxide equivalent
e.g.	exempli gratia
etc.	et cetera
GWP	Global warming potential
i.e.	id est
LCA	Life cycle assessment
N	Nitrogen
N_2O	Nitrous oxide
Ntot	Total nitrogen
NH_4-N	Ammonium nitrogen as nitrogen
P	Phosphorus
PO_4 eq.	Phosphorus equivalent (eutrophication equivalent)
SFA	Substance flow analysis
SO_2 eq.	Sulphur dioxide equivalent (acidification equivalent)
UD	Urine diversion
US	Urine separation
WWTP	Wastewater treatment plan

1 INTRODUCTION

1.1 BACKGROUND

For decades, the priority of wastewater management has been to keep the environment clean and maintain the health of the population. From the historical point of view, advanced wastewater management and water supply have significantly improved the health of the population and reduced mortality, especially in urban areas, by reducing exposure to water and food-borne diseases (Harris and Helgertz, 2019). In addition, advanced wastewater management has improved the state of the environment.

The Sustainable Development Goals for 2030 (SDGs) include specific targets such as adequate sanitation for all (SDG 6.2), protection of water resources from (nutrient) pollution (SDG 14.1), as well as increased safe reuse of wastewater fractions (SDG 6.3) (United Nations, 2021). In 2020, 62% of the urban and 44% of the rural population worldwide had access to safely managed sanitation services (WHO and UNICEF, 2021). However, most of the world's wastewater is discharged without treatment (WWAP, 2017).

In today's Finland, approximately 85% of the population is connected to the municipal sewage collection network (Lapinlampi, 2021) and all urban wastewater is collected and treated centrally. The maximum permissible pollution loads for treated wastewater are set in the environmental permits of the plants. Since the mid-1970s, this end-of-pipe policy has succeeded in significantly improving the quality of water discharged from municipal wastewater treatment plants, and currently point sources of phosphorus load account for less than 15% of Finland's nutrient inputs to the Baltic Sea. In addition, nitrogen removal began at municipal treatment plants in the mid-1990s and the efficiency has now improved to approximately 60% (Finnish Environment Instutute, 2019), while Finland's national implementation plan of the EU's Marine Strategy Framework Directive has set a target to at least 70%. (MSFD, 2008; Laamanen, 2016; Räike et al., 2019). However, the dicharged water from municipal wastewater treatment remains the second most significant source of nitrogen load from human activities after agriculture (Tattari et al., 2015).

In contrast, in sparsely populated areas of Finland, about 350,000 permanent residences and an additional 450,000 holiday homes must treat their own wastewater on-site to avoid pollution of nearby freshwater ecosystems and groundwater. There are almost 200,000 lakes in Finland, often with settlements nearby. The Government Decree on Treating Domestic Wastewater in Areas Outside Sewer Networks (209/2011) (referred below as the "On-site Decree") sets minimum standards for wastewater treatment. The main objective of the On-site Decree is to protect the nearby water systems, such as groundwater, wells and shores. However, a significant proportion of the existing rural on-site wastewater treatment systems are still only septic tanks, thus failing to meet the legislated treatment requirements (Kallio, 2020).

At the moment in Finland, wastewater from sparsely populated areas and secondary residences is the second most significant source of phosphorus load into water systems after agriculture (Tattari et al., 2015). However, the estimate of the load is computational, and in practice the result depends on, for example, the on-site treatment practices and the distance to the water body. Despite the efforts to reduce the loads from diffuse sources, Finland has not achived its nutrient reduction targets by 2021 (Räike et al., 2020; HELCOM, 2013).

According to conservative estimates, about 68% of the rural residents are immediately required to upgrade their on-site systems due to insufficient treatment (Kallio and Suikkanen, 2019). In addition, the current sewer networks and WWTPs in urban areas have been reported to need considerable maintenance and repair (ROTI, 2021).

1.2 NUTRIENTS IN WASTEWATER

The aim of wastewater treatment has been to protect water bodies, which is why its focus has been on the efficient removal of nutrients and organic matter – not on their recovery. As the main idea of the end-of-pipe policy is to determine the maximum permissible pollution loads for wastewater discharged, the lack of focus on nutrient recovery and recycling has created other environmental problems over the years.

Firstly, centralized sewer systems are based on the mixing and transportation of urban, hospital, landfill and industrial wastewater, resulting in a dilute mixture of nutrients and organic matter along with a widerange of different chemical compounds and contaminants (Rogers, 1996; Kuster et al., 2005; Diaz-Cruz et al., 2009). As current practices in wastewater and sludge management only remove some of the contaminants, some of the harmful substances end up in discharged water and sludge (Magnusson and Norén, 2014; Vieno, 2014; Talvitie et al., 2017; Vieno et al., 2018; Ylivainio et al., 2020; Lehtoranta et al., 2021a). The direct application of sludge-based products involves uncertain risks from contamination of soil, crop and water bodies with pathogens, heavy metals, micro-organic pollutants (e.g. pharmaceuticals etc.) (Seleiman et al., 2020) and microplastics (Hurley and Nizzetto, 2018; Schell et al., 2022). The safety concerns have reduced society's willingness to recycle wastewater-based nutrients and organic matter back into agriculture (Simha et al., 2017, 2021a).

Secondly, only a small fraction of the nutrients (phosphorus and nitrogen) originally contained in the wastewater remain and are found in the recovered sludge in easily plant-available form (Warman and Termeer, 2005). In Finland, approximately 4,000 tonnes of phosphorus and 32,600 tonnes of nitrogen end up in wastewater treatment plants annually (Finnish Environment Institute, 2019). The nutrient removal efficiency is advanced on a global scale, being about 96% for P and 60% for N (Finnish Environment Institute, 2019). However, with current energy-intensive nitrification-denitrification processes in wastewater treatment plants, about 30% of the nitrogen in wastewater is lost to air in evaporation (Fig. 1.). Another one third of the nitrogen returns to the beginning of the treatment process during the drying of the sludge and the rest ends up in the discharge water and sludge. (Lehtoranta et al., 2021a.) In addition, the removal of phosphorus is based on chemical precipitation, which binds phosphorus to the sludge, which at the same time impairs its availability to plants (Tidåker et al., 2006b; Ylivainio et al., 2020).

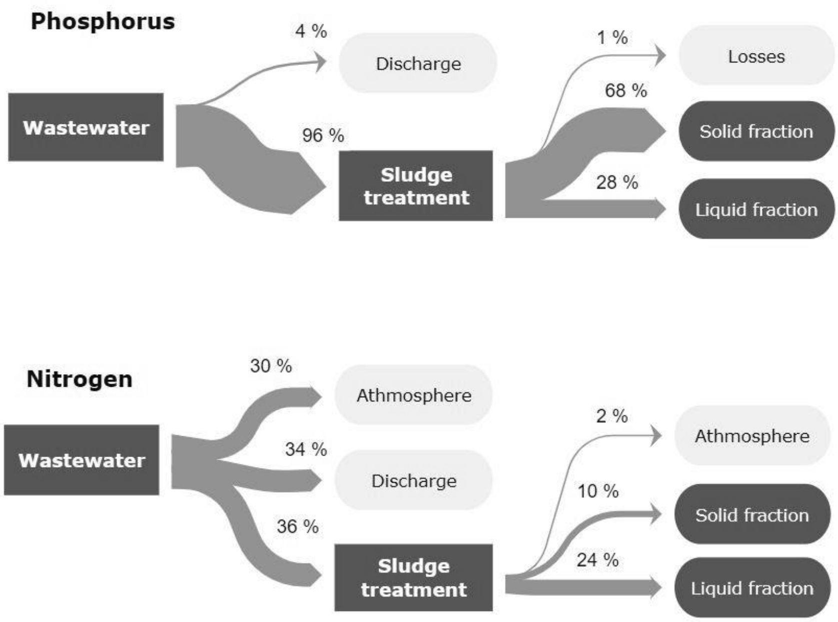


Figure 1 The average phosphorus and nitrogen flows in wastewater treatment in Finland. Sludge treatment includes pretreatment and digestion of sludge. Composting is not included. According to current Finnish practices, the liquid fraction is led to the beginning of the process. Figure modified from Lehtoranta et al. (2021a).

The sewage sludge formed in the wastewater treatment process is typically post-treated by anaerobic digestion and the digestate is composted (Lehtoranta et al., 2021a). Most of the remaining soluble nitrogen evaporates during composting and the remaining nitrogen is in an organic form that is slowly available to plants (Nacry et al., 2013).

Currently, treated sewage sludge is most commonly utilized in agriculture (47%) in Finland, and less than half (40-45%) ends up in landscaping (Vilpanen and Seppälä, 2021). When sludge is used for landscaping, multiple times higher amounts are used per area compared to agricultural use. Nutrients end up in the soil excess the need of plants' and, some leach into surface and groundwater, making landscaping in urban areas not a way to recycle nutrients and organic matter, but rather a way to dispose sludge. Indeed, diffuse pollution in urban areas has been reported high (Valtanen et al., 2015) which may deteriorate the ecological quality of receiving waters.

In rural areas, in addition to the fact that legal requirements for household wastewater management are not always complied with, the commonly used septic tank retains only a small fraction of nutrients, about 5-25% of phosphorus and 5-15% of nitrogen, in the sewage sludge (Olshammar et al., 2015; Malila et al., 2019a). The sludge from septic tank or other type of on-site wastewater treatment system is collected from households and transported to wastewater treatment plants, where it is diluted and mixed with other wastewater for further treatment. Therefore, most of the nutrients produced in residential areas end up in the environment causing a risk of pollution. Also, the on-site wastewater treatment system requires maintenance by its resident to ensure it works properly. Neglecting the maintenance may deteoriate nutrient rentention in the sludge, resulting in higher emissions to the environment.

1.3 INEFFICIENCES IN NUTRIENT RECYCLING

There is a growing global demand for nutrients in food production, which is expected to increase as the population grows (FAO, 2006). The extensive mining of phosphate rocks has led to a significant depletion of known stocks, resulting in the inclusion of phosphate rock in the EU's list of critical raw materials (European Commission, 2017). However, in 2011, the amount of economically available reserves was estimated to be considerably higher than previously estimated (Jasinski, 2011). Phosphorus reserves are geologically unevenly distributed, with 77% of the world's resources located in Morocco and its share is excpected to increase in the future (Cooper et al., 2011; Jasinski, 2011). In addition, the quality of the remaining phosphate rock deteriorates, leading to an increase in its production costs (Cordell et al., 2009). Within the EU, phosphorus is only produced in Finland.

In addition to phophorus, nitrogen fertilizers are essential for plant nutrition. In Finland, the imports of nitrogen fertilizers and their raw materials have been dependent on Russia. Since the start of Russia's war of aggression against Ukraine, the availability of nitrogen fertilizers and their raw materials have collapsed. Before the war, the prices of nitrogen fertilizers had already risen to their highest levels globally by the end of 2021 (Myers and Nigh, 2021), and prices have continued to rise due to the war in Ukraine. In addition, the production of nitrogen fertilizers is responsible for about 0.8% of global greenhouse gas emissions due to its high consumption of natural gas (Brentrup, 2009). Ensuring the availability of nutrients and the sustainability of their production are a critical issue for global food production and security.

In Finland, food production and consumption systems have been studied to be clearly the largest phosphorus and nitrogen flow sectors (Antikainen, 2007). As a result of food consumption, part of the nutrients bound to the food are excreted in urine and feces into the wastewater. Both urine and feces contain vital nutrients (phosphorus and nitrogen) for food production. In the urine, the nutrients are in a water-soluble ionic form that are readily available to plants (Schönning, 2006; Udert et al., 2006; Viskari et al., 2018). Nutrients are somewhat less available in the feces due to their binding in organic material (Spångberg et al., 2014). The nutrient content of urine and feces excreted by the Finnish population before loss, is altogether approximately 27,760 tonnes of nitrogen and 4,148 tonnes of phosphorus. This corresponds to 19% of the total nitrogen and 36% of the total phosphorus in chemical fertilizers sold to farms in Finland (Natural Resource Institute Finland, 2022). However, instead of effectively recycling wastewater nutrients back into food production or other efficient utilization, the need of nutrients in agriculture is replaced by introducing more mineral fertilizers into the system.

Due to the openness of the food production and consumption system (Antikainen, 2007), nutrient leakage in the sector causes environmental problems such as eutrophication, acidification, climate change and waste of natural resources. In addition to the inefficient recovery and reuse of wastewater nutrients, biological processes in wastewater induce methane and nitrous oxide emissions, both of which are strong greenhouse gases. Despite the fact, that about 15% of the population in Finland lives outside the sewerage network, their share of the greenhouse gas emissions from wastewater treatment is 55% according to the national greenhouse gas inventory (Fig. 2) (Lapinlampi, 2021; Statistics Finland, 2021).

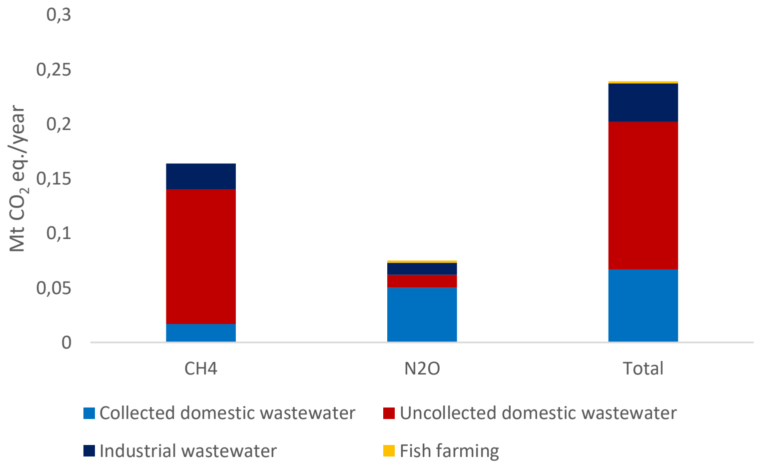


Figure 2 Reported emissions of methane (CH_4) and nitrous oxide (N_2O) in Mt CO_2 eq./year from wastewater treatment by national greenhouse gas inventory in 2019 (Statistics Finland, 2021).

1.4 SOURCE SEPARATION OF WASTEWATER

Apart from end-of-pipe solutions, source separation has emerged as an efficient way of recovering wastewater-based nutrients while enabling energy production at the same time. Furthermore, source separation of wastewaters can be accomplished either by collecting and treating separately the wastewaters generated by society (hospitals, households, industry, landfills) or more specifically, at the household level, which is the focus of this study. Household wastewater contains a mixture of nutrient-rich blackwater (containing feces, urine and toilet paper) and nutrient-poor grey water (from kitchen appliances, showers, etc.). If the nutrient-rich part of the wastewater is not mixed with grey water in the household and municipal wastewater treatment plant with other nutrient-poor waters (e.g. industrial wastewater, stormwater and infiltration), the nutrients can be recovered more easily and without mixing with harmful substances from sources other than toilet water (Jönsson et al., 2005; Tidåker et al., 2006b; Saliu and Oladoja, 2021). In addition, nutrient recovery is technically easier in systems where the nutrients are not diluted and are thus present in higher concentrations. Moreover, source separation retains the fertilizing characteristics of each fraction (Wielemaker et al., 2018).

In addition to effective nutrient recovery, the safety of recovered nutrients can be improved without mixing the nutrients with harmful substances originating from other sources. Furthermore, in source separation systems, both the risk of pathogens in the final products (Xue et al., 2016; Vidal et al., 2019) and the heavy metal content have been estimated to be very low (Lennartsson et al., 2009). However, urine contains majority of the pharmaceuticals and hormones (Udert et al., 2006), but the concentration of extractable pharmaceutical and hormones in the soil fertilized with source separated urine has been studied to remain below the detection unit (Viskari et al., 2018). Thus, source separation systems increase the potential for nutrient recovery, but also the potential for using the nutrients as fertilizers in a more soluble and contaminant-free form.

In households, the source separation can be accomplished with urine-separating toilet types or by collecting all toilet effluents in a single fraction (blackwater). Blackwater covers only 15% of the whole domestic wastewater volume in Finland (Motiva, 2020), but contains about 90% of the nitrogen and 80% of the phosphorus (Viskari et al., 2017; Saliu and Oladoja, 2021). In contrast, urine accounts for less than 2% of domestic wastewater volume and contains about 80% of the nitrogen and about half of the phosphorus and potassium (Jönsson et al., 2005). Thereby, the separation of blackwater allows highest recovery of nutrients, but urine separation maximises the amount of nutrients in the separated fractions and minimises the additional burden from faecal pathogens at the same time (Viskari et al., 2021). All in all, both blackwater and urine separation systems therefore have great potential for nutrient recovery.

The technology for source separation (especially blackwater separation) is advanced, and is already utilized e.g. in sea transportation, trains and airplanes, but the source separated wastewaters are typically afterwards mixed with other wastewaters and treated at WWTPs. In recent years, source separation systems have also been implemented in urban areas and new, larger systems are being planned (Lennartsson and Kvarnström, 2017; Skambraks et al., 2017; Stowa, 2018; Gomez et al., 2020).

1.5 POLICY MEASURES AND DECISION MAKING IN NUTRIENT RECYCLING

Nutrient recovery is one of the key drivers promoting sustainability and circular economy in wastewater management (Hoffmann et al., 2020; Larsen et al., 2009) and food systems (Tseng et al., 2019). Increasingly, wastewater is seen as a valuable source of nutrients, energy and water (Van der Hoek et al., 2002; Sutton et al., 2011; Van der Hoek et al., 2016; Salgot and Floch, 2018; Rodriguez et al., 2021). Recently, interest in nutrients in wastewater has also increased as a means to improve the security of nutrient supply. Given the growing need for nutrients in food production, such as phosphorus and nitrogen, more attention should be paid to the efficient recovery and safe circulation of wastewater nutrients to agriculture.

In the European Union, the legislation framework supporting the recovery and recycling of wastewater nutrients is currently partially outdated, but under renewal. The framework has been found to be unclear and nutrient recycling has been governed by fragmented decision-making (Hukari et al., 2016). In Finland, nutrient recycling has been actively promoted since 2010, when the government issued a Baltic Sea commitment (Working Group Memorandum, 2011). According to the Nutrient Recycling Action Plan of Finland (2019-2030), the aim is to utilize recycled nutrients, including those in sewage sludge, mainly as fertilizers by 2030 (Ministry of the Environment, 2019). Currently, the legislation on fertilizers and their origin is being reformed in Finland, and the acceptance of wastewater-based nutrients is one of the key issues for stakeholders.

To meet the demands of the society and the global and national sustainability and nutrient recycling targets, wastewater treatment systems need to be developed to achieve safe and efficient recycling of nutrients and organic matter. In addition, Finland's wastewater treatment infrastructure is aging with a hundreds of millions of euros reparation dept in the maintenance of water and sewer networks and wastewater treatment systems (ROTI, 2021). Moreover, failures to meet the legislated treatment requirements and difficulties in transportation and treatment of collected sludge in rural areas support the need for transition. It is therefore topical to explore alternatives alongside the renewal of the current system to meet the targets of nutrient recycling and the circular economy in the future in both urban and rural context.

1.6 OBJECTIVES OF THE STUDY

In previous research, Life Cycle Assessment (LCA) has been applied to source separation systems mostly in urban concepts (Remy and Jekel, 2012; Spångberg et al., 2014; Kjerstadius et al., 2015; 2017) while rural concepts have received less attention (Tidåker et al., 2006a; Benetto et al., 2009). Therefore, there exists a lack of research data on the potentials and differences between these concepts. The objective of this thesis is to analyze, from a life cycle perspective, source separation systems in urban, peri-urban and rural settings. More precisely, the main objective of this thesis is to analyze the life cycle environmental impacts, advantages and drawbacks, of wastewater management pursuing improved nutrient recovery with source separation of nutrient rich wastewaters from households. A specific methodological goal is to reveal the limitations of life cycle assessment (LCA) in assessing environmental impacts of nutrient recovery and recycling.

The specific research questions of this thesis are:

1. How source separation of wastewater contributes to sustainability in terms of life cycle environmental impacts (climate, freshwater eutrophication, acidification) and nutrient recovery potential compared to the conventional system in urban, peri-urban and rural areas of Finland?

2. How the recovery and recycling of nutrients contributes to the climate impacts of wastewater management and what are the most critical uncertainties and methodological limitations regarding the assessment?

3. What is the contribution and relevance of source separation systems in pursuing change towards circular economy in wastewater management?

2 MATERIALS AND METHODS

2.1 THEORETICAL FRAMEWORK

In the scientific field of industrial ecology, the interactions between human activities and the environment are analyzed (Socolow et al., 1994). The aim of industrial ecology is to reduce resource consumption as well as environmental impacts through closed loop of materials and substances. The closed-loop economy is the core of industrial ecology (Ayres and Ayres, 2002).

The concept of circular economy combines a wide range of sustainability sciences and industrial ecology. Korhonen et al. (2017) defines circular economy as ‘an economy constructed from societal production-consumption systems that maximizes the service produced from the linear nature-society-nature material and energy throughput flow’. This can be achieved with cyclical material flows, renewable energy resources and cascading-type energy flows. Circular economy has been promoted by the European Union and several national governments and businesses around the world (Korhonen et al., 2017). At the European Union level, targets on circular economy are ambitious and according to the recently accepted Circular Economy Action Plan, EU aims to have a carbon neutral, sustainable, toxic-free, and fully circular economy by 2050 (European Commission, 2020).

Alongside with circular economy, technological change is one of the key elements of industrial ecology. Socio-technical transitions and long-term transformation processes are fundamental in shifting to more sustainable modes of production and consumption (Geels and Schot, 2010). Socio-technical change requires a change in infrastructure and technologies, as well as in people’s practices and attitudes (Geels, 2005; Shove, 2014).

In food systems, where nutrient recycling plays an important role, the socio-technical change towards circularity has been studied for example by Jurgilevich et al. (2016). They suggest that the regulation of nutrient flows should be conducted using a cross-sectoral approach and that a coordinated and comprehensive policy should be developed. Jurgilevich et al. (2016) calls for more focus and incentives on nutrient recovery and recycling practices and the strengthening of local food systems.

Several system analysis tools are utilized to analyze the interactions and impacts between natural environment and technosphere. In this thesis Substance Flow Analysis (SFA) and Life Cycle Assessment (LCA) are applied. These methods are presented in the following sections.

2.2 METHODOLOGICAL FRAMEWORK

2.2.1 SUBSTANCE FLOW ANALYSIS

Substance Flow Analysis (SFA) is a generic method to quantify flows and stocks of substances or material in various contexts and scales (Van der Voet, 2002). The main principle of SFA is mass balance; the inflow into the system is equal to the outflows (plus stock changes). With SFA, it is possible to detect potential leaks and inefficiencies.

Antikainen (2007) has analyzed the total nitrogen and phosphorus flows in Finland with SFA. SFA has also been widely applied to assess engineering processes and in several applications and models in wastewater and nutrient recovery studies (Breen, 1990; Fan et al., 1996; Puig et al., 2008; Tervahauta et al., 2013; Khiewwijit et al., 2015; Venkatesean et al., 2016; Hong et al., 2019; Cai et al., 2020). According to Antikainen (2007), SFA is typically not a sufficiently detailed decision-making tool, and should be supported by other methods, such as LCA.

Papers II-IV of this study used SFA to analyze the nitrogen (N) and phosphorus (P) recovery potential in household wastewater management systems. The nutrient recovery potential was calculated to describe the proportion of nutrients, that are (reference system) or could be recovered (scenarios) out of the total amount of nutrients excreted in household wastewater. The nutrient recovery potential was calculated from bottom up, starting from the excretion of nutrients to wastewater treatment to the management and storage of fractions. For each activity, the fractions of volatile compounds and nutrients discharged were determined. As a result, the recovered nutrient potential for households per capita per year (kg N and P/year/person) of source separating systems and conventional system was determined (*Papers II* and *III*). In *Paper IV*, calculations were made by region for Northern Finland and Sweden, and separately for urban, peri-urban and rural areas.

2.2.2 LIFE CYCLE ASSESSMENT

Life Cycle Assessment (LCA) is a method for assessing the potential environmental impacts during the entire life cycle of a product, process, or service, based on ISO 14000 series (ISO 14040, 2006). The two main approaches of LCA have been defined as Attributional LCA (ALCA) and Consequential LCA (CLCA) (Curran at al., 2005; Finnveden et al., 2009). The ALCA analysis aims to describe the systems as it is – more specifically, it is an estimate of the share of the global environmental impact of the system or product under investigation (Weidema, 2003). According to Finnveden et al. (2009) ALCA is defined as following: 'LCA aiming to describe the environmentally relevant physical flows to and from a life cycle and its subsystems'. This approach is typically used to assess the carbon footprint of products. In CLCA, on the contrary, the aim is to describe how the environmental impacts are affected by the production and use of the product, or how the impacts would change as a consequence of a decision or action (Ekvall and Weidema, 2004; Finnveden et al., 2009, Ekvall et al., 2016). As a result, the main differences between these approaches lie within the system boundary definition and the input data used for calculations. Hence, the ALCA uses allocation and average data, while the CLCA uses system expansion and marginal data. (Ekvall, 2019.) However, a consensus on the differences between the methods are still lacking (Schaubroeck et al., 2021) and varying interpretations are common in the actively evolving LCA field (Ekvall et al., 2016; Ekvall, 2019).

In practice, the assessment consists of four phases (ISO 14040, 2006): goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA) and interpretation. The goal and scope of the study define the appropriate system boundary and other methodological choices, such as allocation methods and functional unit. The functional unit describes the function of the product or process under study whose environmental impacts are being assessed (Finnveden et al., 2009). Uncertainties in the assessment are recommended to be analyzed and several methods can be applied to it, such as Monte Carlo analysis.

LCA has been seen as a useful tool for the systematic investigation of the life cycle environmental impacts of wastewater management systems (Corominas et al. 2013; Huegel, 2000) and has been applied since 1990s (Tillman et al., 1998). Several studies have compared the sustainability of wastewater treatment systems with LCA, for example, Renou et al. (2008), Emmerson et al. (1995), Brix (1999), and Balkema et al. (2001). Moreover, the source separation of wastewater has been analyzed with LCA both in urban and rural concepts by Tidåker et al. (2006a), Benetto et al. (2009), Remy and Jekel (2012), Spångberg et al. (2014), Thibodeau et al. (2014) and Kjerstadius et al. (2015; 2017). However, for the first time, this thesis brings together urban, peri-urban and rural concepts.

In this study, LCA was carried out to analyze the environmental impacts of household wastewater management and the role of source separation in nutrient recovery both in rural and urban context (Table 1). In *Paper I*, an ALCA was constructed to describe the carbon footprint of the available alternative on-site wastewater management systems, where avoided products (carbon storage of nutrient rich biomass) were credited in product systems. Calculations were made by using the matrix method in Microsoft Excel for solving the set of linear equations representing the processes (Suh and Huppes, 2005). In *Papers II* and *III*, the aim of the LCA was to analyze, how the environmental burden will change in rural on-site wastewater treatment systems (rural systems) and urban wastewater treatment systems (urban systems) due to improved nutrient recovery of source separation. Therefore, the approach chosen was CLCA as Heimersson et al. (2019) recommends. Calculations for *Papers II* and *III* were made with Microsoft Excel and by using SimaPro-program.

For impact categories, climate change (CO_2 eq.), freshwater eutrophication (PO_4 eq.) and acidification (SO_2 eq.) were chosen for study. Freshwater eutrophication and acidification are both impact categories strongly connected to nutrient recycling. Eutrophication is a process in which water ecosystem becomes enriched with nutrients, which increases production and the risk of oxygen depletion. Acidification is a process in which the decrease in pH in the soil, due to e.g. ammonia emissions, reduce the availability of nutrients to plants and increase the solubility of toxic aluminium and heavy metals, causing damage to ecosystems.

Table 1. *A summary table on applied methodologies of Papers I-IV.*

	Paper I	**Paper II**	**Paper III**	**Paper IV**
Region	Rural	Rural	Urban	Regional (urban, peri-urban, rural)
Scale	Household of three persons	Household of three persons	City district for 26,000 inhabitants and 6,510 jobs (including schools and day care)	Northern Finland and Sweden; urban, peri-urban and rural areas
Methodo-logical framework	ALCA	CLCA, SFA	CLCA, SFA	SFA, CLCA as synthesis from *Papers II* & *III*
Functional unit	One year's use of an on-site system functioning according to the requirements set for in the On-site Decree	The amount of nutrients and wastewater produced in one year per person	The amount of nutrients and wastewater produced in one year	The amount of nutrients and wastewater produced in one year
Impact categories	climate change, freshwater eutrophication	climate change, freshwater eutrophication, acidification	climate change, freshwater eutrophication, acidification	climate change, freshwater eutrophication, acidification

Emission character-rization	Climate change: IPCC 2007 characterization factors (Solomon et al., 2007) Freshwater eutrophication: Seppälä et al., 2004.	Climate change and acidification: ReCiPe Midpoint H, completed with the GWP characterization factors for CH_4 and N_2O (IPCC, 2014). Freshwater eutrophication: Seppälä et al., 2004.	Climate change and acidification: ReCiPe Midpoint H, completed with the GWP characterization factors for CH_4 and N_2O (IPCC, 2014). Freshwater eutrophication: Seppälä et al., 2004.	-
Normali-zation and weighting	Not considered	Normalization was performed using European level normalization factors (ReCiPe Midpoint H method). No weighting was taken into account.	Not considered	-
Nutrient recovery considered	Not considered	Yes	Yes	Yes
Methods for uncertainty analysis	Scenario analysis (one-at-a-time (OAT) sensitivity analysis)	Not considered	Scenario analysis (one-at-a-time (OAT) sensitivity analysis)	Parameter variation (SFA)

2.3 DESCRIPTION OF CASE STUDIES AND SYSTEM BOUNDARIES

2.3.1 CASE STUDIES IN PAPER I

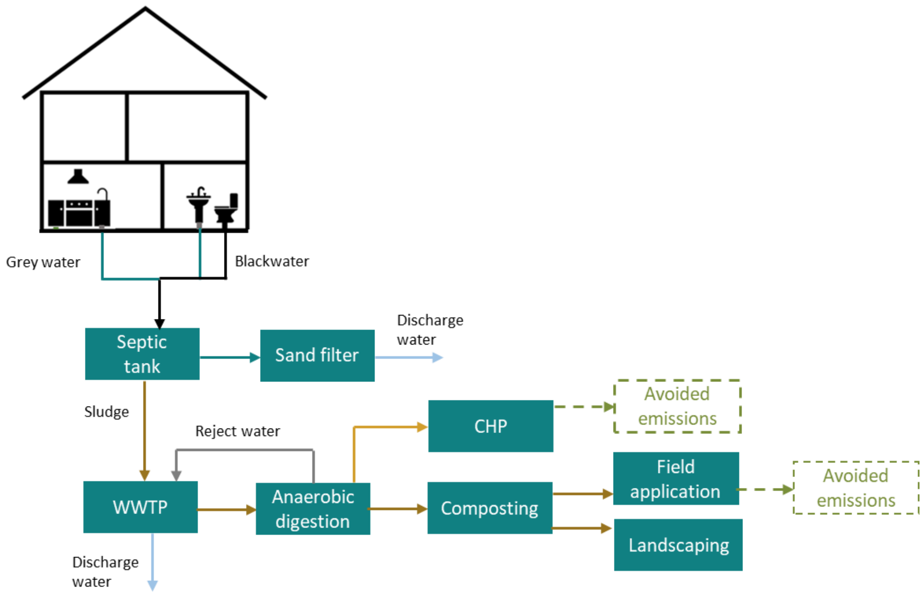
In *Paper I*, the goal of the study was to analyze the commonly available on-site wastewater treatment options for rural areas of Finland which meet the requirements of the On-Site Decree (The Government Decree on Treating Domestic Wastewater in Areas Outside Sewer Networks (209/2011)) in reaching the purification performance (85% reduction in phosphorous, 40% in nitrogen and 90% in biological oxygen demand (BOD)). Six alternatives were chosen for comparison: A) sequencing batch reactor (SBR), B) biofilter, C) soil infiltration, D) buried sandfilter, E) holding tank for blackwater and soil infiltration for grey water, and F) dry toilet with grey water treatment. Also, two sub-alternatives were used for grey water treatment: soil infiltration and pre-fabricated grey water filter. In the latter two systems, the grey and blackwater were sewered separately. In this thesis, the buried sand filter (D) was chosen as a reference and compared to alternatives E (blackwater separation) and F (urine separation with dry toilet and soil infiltration for grey water).

The sludge and blackwater formed at the on-site treatment (E, F) were transported to a municipal WWTP for further treatment. The sludge from WWTP was composted and peat was used as a supporting material. The composted sludge was utilized in landscaping. Both urine and compost from a separating dry toilet were used for fertilization on the property.

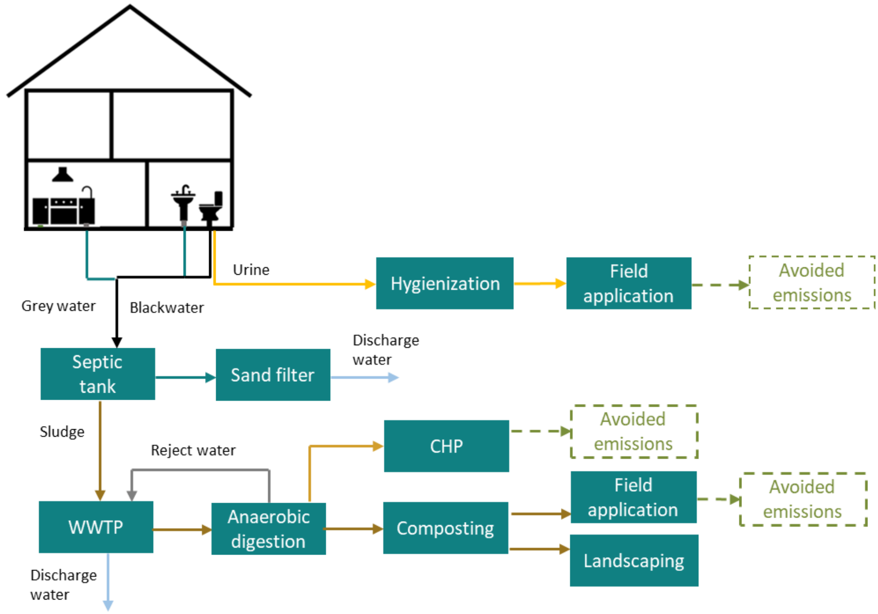
2.3.2 CASE STUDIES IN PAPER II

In *Paper II*, the goal of the study was to analyze two alternative source separation systems designed for nutrient recovery (blackwater and urine separation systems) and to compare them with the current rural wastewater treatment system (Fig. 3). The rural reference system was defined as a system in which all household wastewater was treated together in an on-site three-chamber septic tank followed by a sand filter. The sludge from the septic tank was transported to a WWTP for further treatment (including anaerobic digestion and composting). In the urine separation system, urine and feces were collected separately in a separating dry toilet. The collected urine was stored on the property in a closed tank and transported to the field once a year to be utilized as fertilizer. Feces was composted in a composter and the compost used at the property. The grey water was treated in the same way as in the reference system, except that the septic tank sludge was transported to the WWTP once a year. In the blackwater separation system, urine and feces were collected together in a vacuum toilet system in order to minimize the water consumption and the need for transportation. The blackwater was transported to a local anaerobic digestion plant. The biogas produced was utilized in combined heat and power production (CHP). The digestate was used for fertilizer purposes and the grey water was treated in the same manner as in the urine separation system. The reject water from the AD-plant was assumed to be recycled back into the process.

Reference system



Urine separation



Blackwater separation

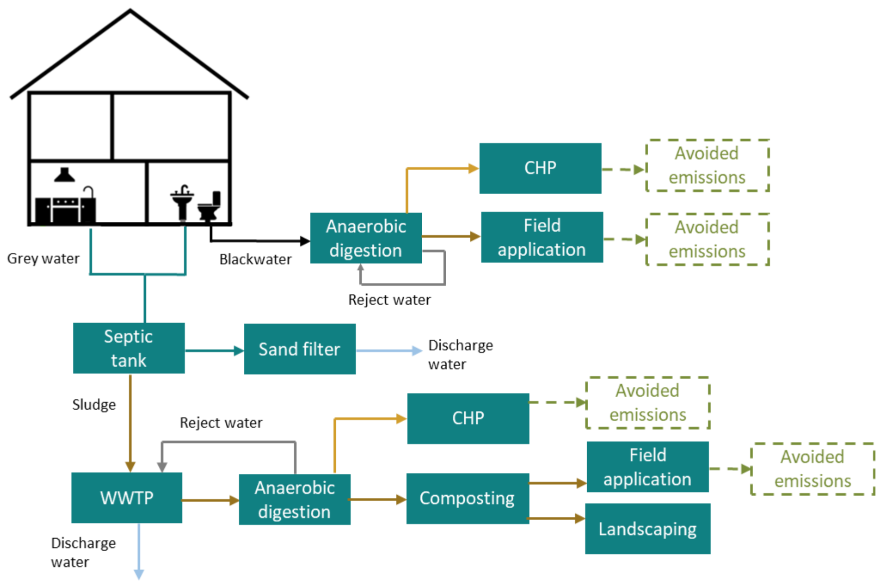


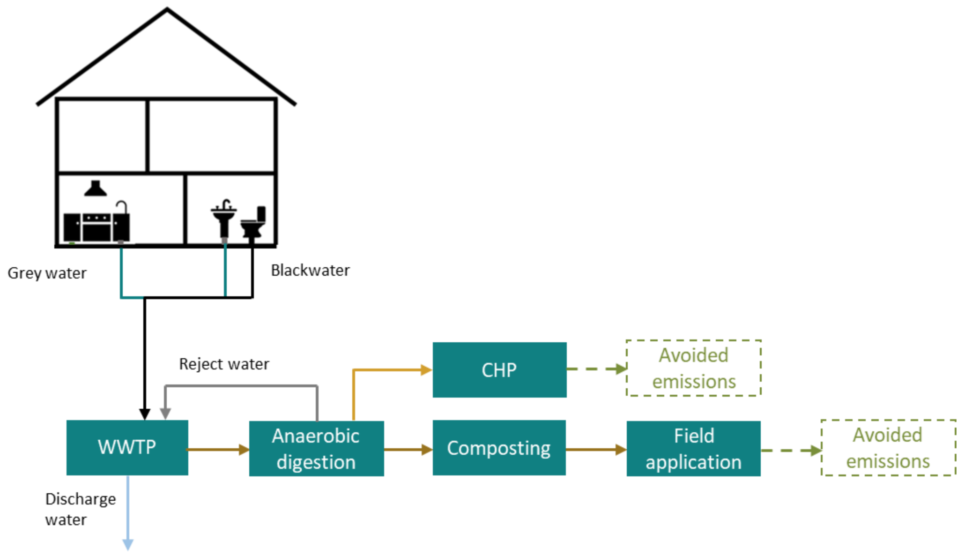
Figure 3 Summary of SFA and LCA system boundaries for reference system and source separation scenarios in *Paper II*.

2.3.3 CASE STUDIES IN PAPER III

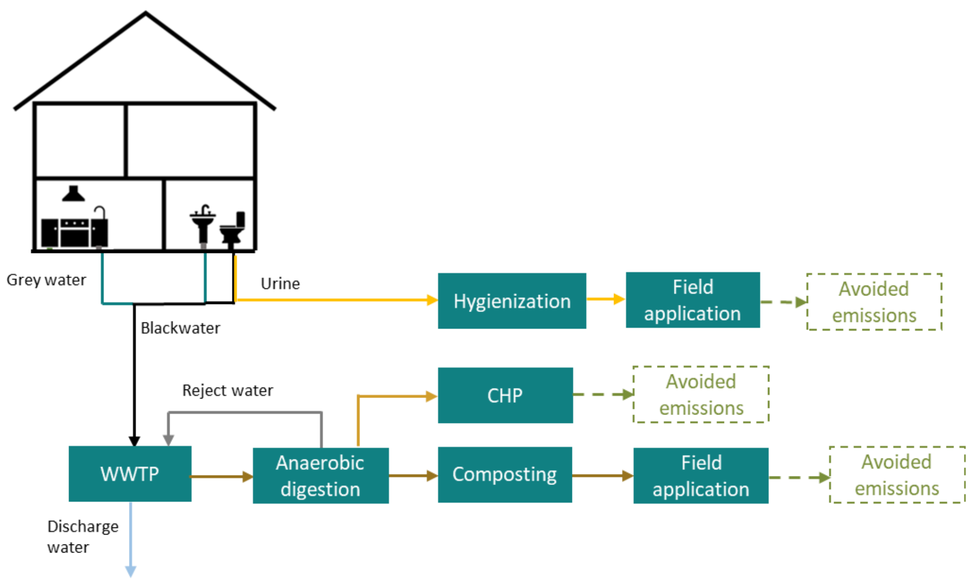
In *Paper III*, the goal of the study was to analyze two alternative source separation systems designed for nutrient recovery (blackwater and urine separation systems) and to compare them with the conventional wastewater treatment system in urban context at Hiedanranta, Tampere. The analysis covered all wastewater generated in the Hiedanranta district, which has 26,000 inhabitants and 6,510 jobs once all the planned houses have been built. 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. Thus, only mature technologies were chosen. Two scenarios for source separation were developed for comparison: A) urine separation with separating water-flush toilets, and B) blackwater separation with vacuum toilets.

The generated wastewater fractions in the reference system and source separating scenarios (Fig. 4) were assumed to be either fully or partially treated at the Sulkavuori WWTP and the produced sludge to be digested and composted. The composted sludge was utilized in agriculture. In the separating systems, grey water and feces (A), or only grey water (B) were assumed to be treated at the Sulkavuori WWTP in the same way as in the Reference system. In urine separation (A), urine was hygienized in the basement of block houses for six months, and then collected and transported for field application. Blackwater (B) was treated at the local AD plant in Hiedanranta and the reject water from the AD plant was recycled back to the AD unit in accordance with current Finnish practices. The produced biogas at Hiedanranta was upgraded to transport fuel. Digestate was used as a fertilizer and applied to fields.

Reference system



Urine separation



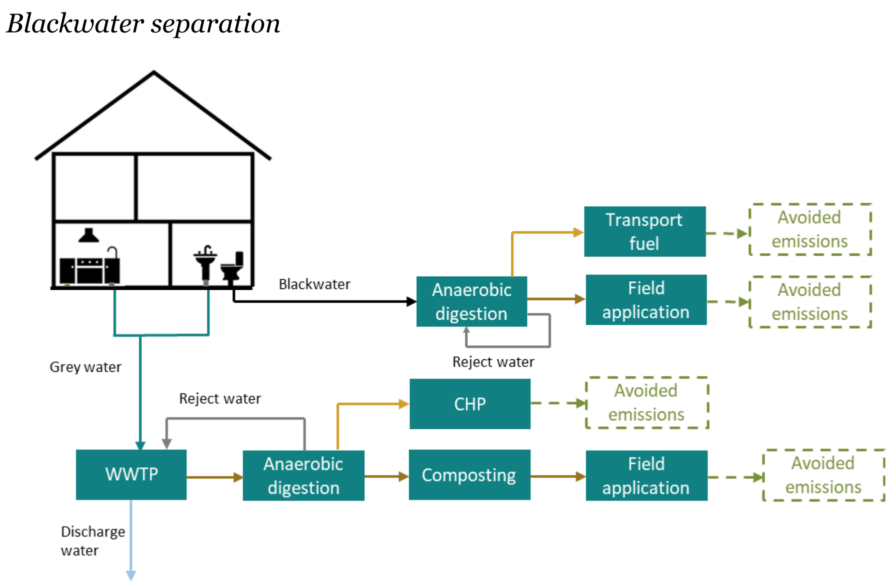


Figure 4 Summary of SFA and LCA system boundaries for reference system and source separation scenarios in *Paper III*.

2.3.4 CASE STUDIES IN PAPER IV

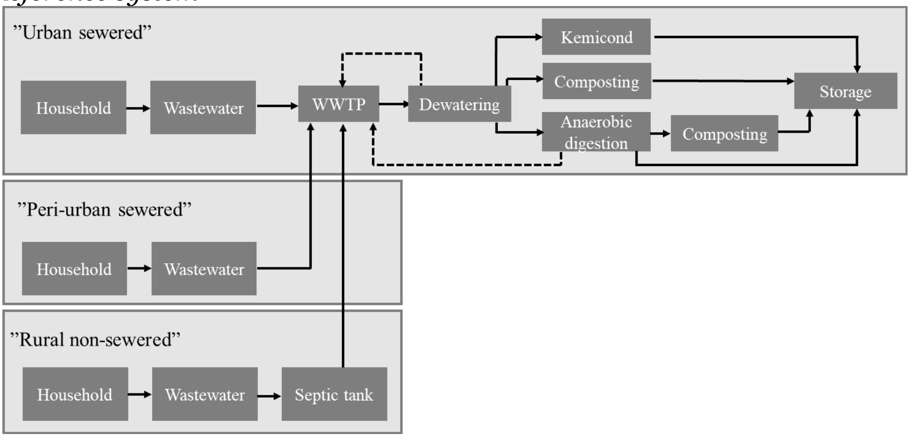
Paper IV compared the implementation of source separation sanitation systems (urine separation and blackwater separation) in the whole area of northern Finland and Sweden to the existing, conventional wastewater management in the peri-urban and rural areas (Fig. 5). The study area covered North Ostrobothnia, Lapland and Norrbotten.

In this region, 42–47% of the population of the study area lives in urban areas. The peri-urban areas account for 11–12% of the population and the rural areas for 41-47 % of the population. In the studied region, 100% of urban and peri-urban households are connected to the sewage network. The rural areas consist of areas where houses are connected (17–34%) or not connected (13–24%) to sewage networks. Therefore, households were allocated into three groups based on population density and coverage of sewage collection: urban areas connected to the sewage network ('urban sewered'); rural and peri-urban areas connected to sewage network ('peri-urban sewered'); and rural areas not connected to the sewage network ('rural non-sewered'). Summer houses were not included in the study due to the lack of available data. In urban areas, the implementation of source separation systems was not assessed since retrofitting of such systems in built-up areas is more difficult due to the lack of space and costs, for example (McConville et al., 2017; Hoffmann et al., 2020). Therefore, only peri-urban areas and rural areas were considered for urine and blackwater separation systems. The calculations were made separately for these three groups: ‘urban sewered’, ‘peri-urban-sewered’ and ‘rural non-sewered’ and for the three study areas (North Ostrobothnia, Lapland and Norrbotten). Due to the large size of the region studied and the variety of practices in the area, a streamlined approach was selected for the SFA to describe the practices in the region in general.

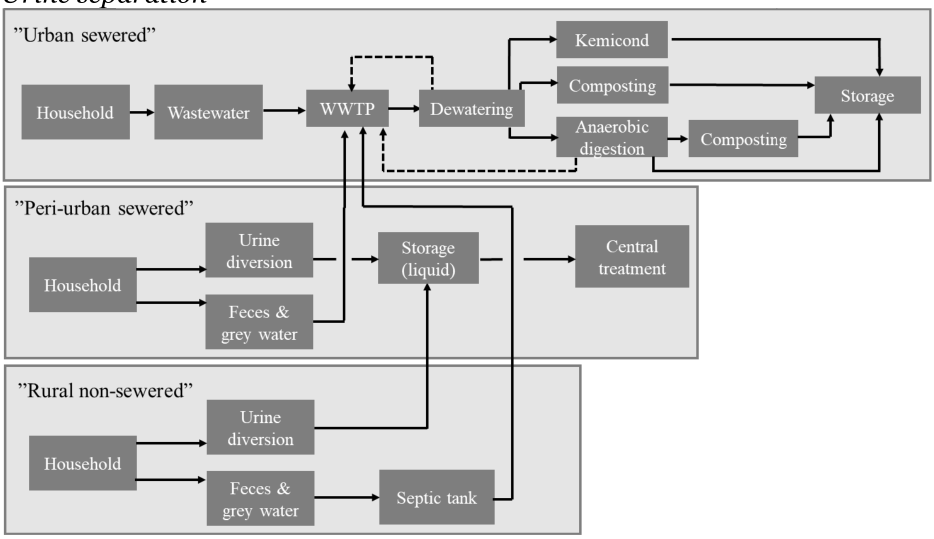
Under current wastewater management practices, wastewater produced in ‘urban sewered’ and ‘peri-urban sewered’ is treated in municipal WWTPs, and wastewater produced in ‘rural non-sewered’ areas is treated in on-site treatment systems and the sludge produced is transported to WWTPs. The formed sludge is treated either with anaerobic digestion (with or without composting), composting or kemicond in accordance with current sludge treatment practices in the study area (presented in more detail in the *Paper IV*). The solid fraction from the anaerobic digestion of the wastewater sludge is utilized and the liquid fraction (reject water) is recycled back to the WWTP for treatment, which is a common practice in Finland and Sweden. The sludge is utilized in both agriculture and landscaping in accordance with current practices in the study area (presented in more detail in the *Paper IV*). However, nutrient losses from field application were not included in the SFA system boundaries due to simplification.

In the scenarios, the conventional system was assumed to be unchanged in urban areas. In urine separation, urine from ‘peri-urban sewered’ and ‘rural non-sewered’ was collected and treated centrally in local facilities using different technologies. Feces and grey water from ‘peri-urban sewered’ communities were treated in WWTPs, as was sludge produced in on-site systems in ‘rural non-sewered’ households. In blackwater separation, blackwater from ‘peri-urban sewered’ and ‘rural non-sewered’ was treated at centralised anaerobic digestion plants. Both solid and liquid fractions from the solid-liquid separation of the digestated sewage sludge were taken into account. Grey water from ‘peri-urban sewered’ households as well as sludge produced in grey water treatment systems in ‘rural non-sewered’ households were treated at WWTPs instead of centralised anaerobic digestion plants, with the aim to reduce the amounts of harmful substances in the nutrient-rich end products.

Reference system



Urine separation



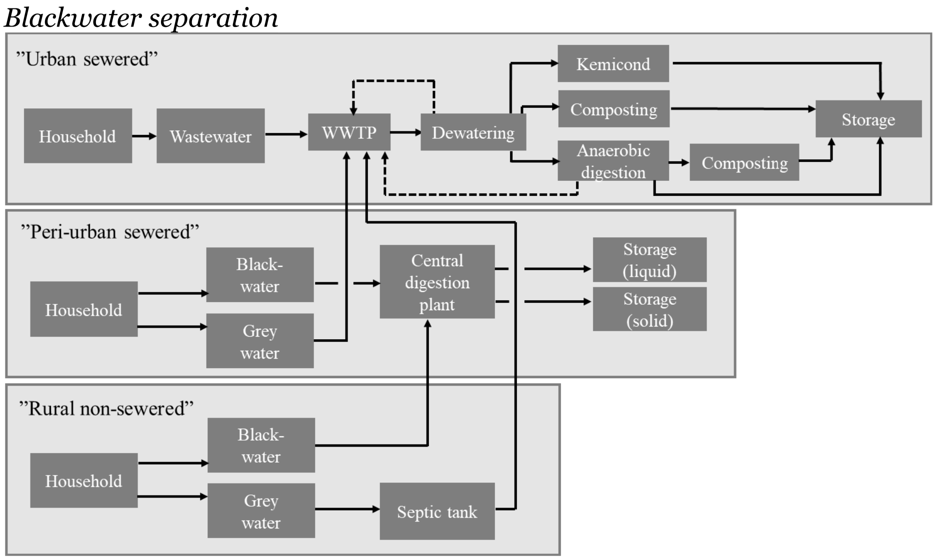


Figure 5 Summary of SFA system boundaries for reference system and source separation scenarios in *Paper IV*.

2.3.5 SYSTEM BOUNDARIES IN LIFE CYCLE ASSESSMENTS

The system boundaries and the processes considered in the LCAs of *Papers I-III* are summarized in Table 2. *Paper IV* is a synthesis of *Papers II* and *III*.

The system boundaries between LCAs and SFAs were consistent in *Papers II-III*. In *Paper IV*, the life cycle impacts were estimated based on emission factors determined in *Papers II* and *III*. However, there were some differences in the system boundaries and in the technologies selected compared to those used in the SFA (description of the system in section 2.3.4), which needs to be considered.

The differences within the LCA system boundaries to SFA boundaries in *Paper IV* are as follows; the nutrient potential of rejected water was not included in the LCAs basic assumptions in *Papers II* and *III*. In the SFA of *Paper IV*, the nutrients in reject waters were includedin the nutrient potential of the scenarios. Furthermore, the *Paper II* considered dry toilets instead of separating water toilets in urine diversion system. In addition, no sophisticated nutrient recovery technologies were applied for urine treatment in the *Paper II* and *III*. The sophisticated technologies applied in *Paper IV* are presented in section 2.4.5.

Table 2. *Summary table of system boundaries and processes included in Papers I-III.*

Fraction	Paper I	Paper II	Paper III
Daily use at the property			
Emissions from toilet system	x	x	x
Energy consumption of the toilet system	x	x	x
Litter for dry toilet	x	x	
Infrastructure			
Infrastructure at the property	x	x	x
Infractructure at WWTP			x
Infrastructure at AD-plant			x
Construction (sewer systems, facilities)	x	x	x
Production of equipments (toilets, on-site treatment plants, pipes etc.)	x	x	x
Transportation			
Transportation of the equipments	x	x	
Transportation of the recovered fractions (e.g. sludge, fertilizers)	x	x	x
Sludge/WW treatment at WWTP			
Chemicals	x	x	x
Energy consumption	x	x	x
Emissions to water and air	x	x	x
Sludge treatment at anaerobic digestion			
Chemicals		x	x
Energy consumption and production		x	x
Emissions to water and air		x	x
Reject waters			(x)
Composting of sludge (from WWTP)			
Emissions to air	x	x	x
Energy consumption	x	x	x
Support materials	x	x	x
Storage			
Blackwater	x	x	x
Urine (hygienization)	x	x	x
Application of recovered fractions			
WWTP sludge use in landscaping	x	x	
WWTP sludge use in agriculture		x	x
Urine (and compost) use at the property	x		

Urine use in agriculture		x	x
Digested BW sludge in agriculture		x	x
Carbon storage of sludge and compost	x		
Avoided processes			
Substitution of mineral fertilizers (sludge, blackwater, urine)	x	x	x
Substitution of electricity		x	x
Substitution of heat	x	x	x
Substitution of transport fuel			x

2.4 DATA AND CALCULATION ASSUMPTIONS

This section provides a brief overview of the data and calculation assumptions for the life cycle inventories (LCI) and SFA. A more detailed description of data sources and calculation assumptions are presented in the respective *Papers*.

2.4.1 BASIC ASSUMPTIONS AND DATA FOR LCA

In *Paper I,* the calculations were made for on-site systems dimensioned for five persons but for a three-person occupancy rate of 85% annual use (overall occupancy rate of 51%). The systems life span was estimated to be 30 years. In addition, in *Paper II* calculations and wastewater treatment scaling were made for a three persons household with 85% annual use. The life span of the toilet system was estimated to be 20 years. In *Paper III,* the total time span for the system was 50 years. In addition, 67% of the wastewater from toilets and 90% of the grey waters were expected to be generated at home. In addition, in the urine separation system, 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 in the flush water of the WWTP. In *Paper II*, respectively, 95% of the urine was assumed to be recovered separately, the rest ended up in the feces.

LCA in *Papers I* and *II* included that the old wastewater treatment system of the house, built between the 1950s and 1970s, including new toilet facilities and sewer pipes, would be completely renovated. This describes the majority of situations where the sewage treatment needs to be improved. The *Paper III* did not have existing wastewater treatment systems, since Hiedanranta is a new urban city district where it was considered to build source separation sanitation instead of the conventional system.

The primary data used in LCA and SFA were collected mainly from the literature in all the *Papers*, but also from urban plans and environmental permit applications. The environmental impact assessment report was utilized especially in *Paper III*. Moreover, some of the data was received directly from companies (*Papers I* and *II*). The secondary data were obtained mainly from Ecoinvent (Swiss Centre for Life Cycle Inventories, 2007) and the LIPASTO databases (VTT, 2011).

2.4.2 NUTRIENTS IN WASTEWATER

The estimation of the amounts of nutrients produced per person slightly varied between studies (Table 3). The estimated amount of nutrients produced was similar in *Papers II* and *III*. In *Paper IV*, lower amounts of excreted nutrients were used to reduce the risk of overestimation of available nutrients. In addition, it was assumed that children and teenagers generate 50% less urine and 30% less feces than adults (Laak, 1974; Almeida et al., 1999; Karak and Bhattacharyya, 2011). The share of children and teenagers in the total population was defined as 15.8% in Finland (Statistics Finland, 2019) and 23.3% in Sweden (Statistics Sweden, 2021). In *Papers II-III,* only total nitrogen (Ntot) was calculated for SFA, but in *Paper IV* also the share of ammonium nitrogen (NH_4-N) was considered.

Table 3. *The amount (kg) of nutrients and BOD produced annually by a person.*

Fraction	P (kg/a/person)	Ntot (kg/a/person)	NH_4-N (% of Ntot)	BOD_7 (kg/a/person)
Urine				
Papers II & III*	0.4	4.13	-	1.83
Paper IV**	0.32	4.02	100	-
Feces				
Papers II & III*	0.21	0.52	-	5.48
Paper IV**	0.18	0.55	-	-
Other				
Papers II & III*	0.15	0.37	-	10.95
Paper IV**	0.15	0.37	-	-
Total				
Papers II & III*	0.75	5.02	-	18.25
Paper IV**	0.66	4.93	81.4	-

**Papers II & III* (Weckman, 2005; Udert et al., 2006; Ministry of the Environment Finland, 157/2017)

***Paper IV* (Jönsson et al., 2005; Government Decree on Treating Domestic Wastewater in Areas Outside Sewer Networks (209/2011); Simha et al., 2017)

2.4.3 DAILY USE AT THE PROPERTY

In on-site treatment systems, nutrient recovery and emissions to water were calculated based on the minimum treatment efficiency requirements under Finnish legislations (Ministry of the Environment Finland, 157/2017). Instead, emissions to air from on-site systems were derived from centralized WWTP emissions from *Papers I* and *II*. In *Paper II,* 50% of the nitrous oxide emissions from a centralised WWTP's were taken into account.

The proportion of nutrients discharged into water bodies from on-site systems (*Papers I-II*), was estimated by the generated eutrophication emission factor. The eutrophication factor is a rough estimate of the variability of the emissions into the freshwater ecosystem after discharges into the soil. Actual emission factors are difficult to measure, since the weather conditions in Finland vary during the year (e.g. soil frost, snow cover and melting) and between years. Majority of the emissions occur in the spring when the snow melts and during the autumn rains.

The energy consumption of the toilet system was calculated based on literature and data obtained from equipment manufacturers.

2.4.4 WASTEWATER AND SLUDGE TREATMENT

For all the *Papers,* data on the conventional WWTP and sludge treatment were obtained from environmental permits, environmental reports and WWTPs planning documents. For LCA and SFA, data were used to calculate emissions, sludge production and energy consumption and production by relating the input of BOD and nutrient loads with the data collected. With regard to conventional wastewater management, all *Papers* included the following steps commonly used in Finnish WWTPs; pretreatment, i.e. grit removal and screening, primary sedimentation, activated sludge process with simultaneous precipitation and drying of the sludge. For *Paper I*, the treatment efficiency was based on Klaukkala WWTP in Finland, and in *Paper II* the average treatment efficiency was calculated based on five medium-sized Finnish WWTP plants.

In *Paper III*, wastewater was treated at the Sulkavuori WWTP, which is currently under construction and will be introduced in 2025. The Sulkavuori WWTP represents the latest technology, but no efforts have been made to improve nutrient recycling. The data used for the Sulkavuori WWTP was supplemented by negotiations with the urban planners of the City of Tampere and the expert assessments of the consultant involved in the design of the Sulkavuori WWTP. Some of the input data, such as the assessment of air emissions and sludge production, utilized the emissions, operation, and environmental permit data of a similar sized Helsinki Viikinmäki WWTP in Finland. The calculations took into account the differences in the input flows and nutrient loads of the Sulkavuori WWTP and their impacts on sludge production, energy consumption and emissions.

For SFA in *Paper IV*, the nutrient recovery in WWTP was estimated based on data gathered in *Papers II* and *III* and in Lehtoranta et al. (2021a). As the characteristics of wastewater and sludge treatment processes are similar in Finland and Sweden, the same nutrient recovery factors were used in both countries.

In *Papers I-III*, it was assumed that the sludge from WWTP is composted together with peat (and other biomasses). The calculations take into account the carbon dioxide emissions from the degradation of peat, but only in *Paper I* the carbon storage formed by sludge-based compost (70 kg C/tonne) was also considered. The emissions from composting were estimated based on the literature in all the respective *Papers*.

2.4.5 URINE AND BLACKWATER TREATMENT

In *Papers I* and *II*, urine was separated with dry toilet and peat was used as litter. Emissions from peat degradation were taken into account. In *Paper I*, the carbon storage from home compost was conducted similarly as for WWTP sludge (section 2.4.5). In *Paper I*, it was assumed that 10% of the nitrogen content of urine will evaporate from the toilet. In *Papers II* and *III*, urine was hygienized and 0.5% of total urine nitrogen was assumed to volatilize as ammonia mainly as a result of uncontrolled leakage (Udert et al., 2006). Urine was not treated further in *Papers I-III*. In urine separation, further treatment was included only in the SFA of *Paper IV,* where average values were used to describe variations in processing technologies applied in the study region, such as membrane technologies, ammonia stripping, Sanitation 360, NPHarvest and Vuna process (Etter et al., 2015; Mönkäre et al., 2016; Marttinen, et al. 2017; Kaljunen et al., 2021; Simha et al., 2021) (Table 5, section 2.5).

In *Paper I*, blackwater was treated in a WWTP, but in *Papers II-IV* it was digested separately to reduce the amounts and variability of harmful substances in the end products. The sludge digestion data from the WWTP were utilized in the process and the solid-liquid separation was included. In *Paper IV*, nutrients from the liquid fractions of blackwater digestion were also taken into account in the nutrient balance calculations, and the same treatment as for urine was assumed for reject waters. No further processing for the dry fraction of blackwater sludge was assumed in any of the *Papers*.

2.4.6 TRANSPORTATIONS

The emissions of the vehicle used were obtained from VTT's (Technical Research Centre of Finland) LIPASTO databases (VTT, 2011) (Finnish traffic exhaust emissions and energy consumption calculation system). The estimated transportation distances and the frequency of transports varied between studies. The frequency of blackwater transportation from the on-site systems were calculated based on the volume of blackwater and the size of the tank used. The sludge from the septic tanks was assumed to be transported twice a year if all wastewater was treated in the soil systems, and once if only grey water was treated. The need for transportation in the construction of soil systems was estimated based on average transportation loads and the need for soil material (e.g. sand). In the urban system, transportations took place by means of blackwater sludge pipelines (*Paper III*). The distances used are described in Table 4.

Table 4. *Transportation distances in kilometers in Papers I-III.*

Transport type	Paper I	Paper II	Paper III
Soil	50	50	-
Septic tank sludge	50	250	-
BW sludge to WWTP	50 (30*)		
Urine to hygienization	-	50	-
WWTP sludge to digestion	-	50	-
BW sludge to digestion	-	50	-
Digestate to composting	-	20	
Sludge to composting	-		-
Nutrients to application	-	100	20

* The value in parentheses describes the distance considered in the sensitivity analysis.

2.4.7 STORAGE AND FIELD APPLICATION OF NUTRIENTS

For the land application of composted sludge, *Paper I* used literature data on nutrient leaching (1.5% for total phosphorus and 5.5% for total nitrogen) and no volatile emissions were assumed. *Paper I* did not assume urine storage emissions, but for household fertilizer use, it was assumed that 60% of urinary ammonium nitrogen evaporates as ammonia and 1.25% of the total nitrogen as nitrous oxide.

Papers II and *III* assumed that advanced spreading techniques were used to apply urine, digested blackwater and digested and composted WWTP sludge. In *Paper II,* a value of 15% was used for ammonia volatilization from soluble nitrogen and 1% for nitrous oxide from total nitrogen. In *Paper III,* the gaseous emissions (ammonia, nitrous oxide) from storage and field application of recycled nutrients were calculated based on international animal manure emission calculation guidelines (EMEP/EEA, 2016; Grönroos et al., 2017) and IPCC (2006) guidelines. No storage of fractions was assumed in *Paper II*.

For eutrophic emissions, the emission factors were mostly similar in *Papers*. *Paper III* used emission factors for nitrogen and phosphorus leaching from manure field application developed by the Baltic Manure- project (INTERREG) (Grönroos et al., 2013a;b) (1.5% of phosphorus and 10% of soluble nitrogen). In *Paper II*, the emission factor for soluble nitrogen was higher, being 15% of the soluble nitrogen.

2.4.8 AVOIDED EMISSIONS

Avoided emissions from the fertilizer use of urine, digested blackwater and digested and composted sludge were calculated in *Papers II-III* assuming that they replace mineral fertilizers. Emissions from both the use and manufacture of mineral fertilizers were considered (*Papers II* and *III*). For nitrogen, only the amount of soluble nitrogen was considered in all the *Papers*. According to the terms of the Finnish farmer's support system, the assumed avoided values for phosphorus in *Paper II* were 100% for urine and 40% for both digested and composted sludge, but in *Paper III* 60% of the phosphorus in sludge was considered due to the changes in the Finnish farmer's support system.

Avoided emissions from energy production were included in the *Papers II-III*. In *Paper II*, the excess biogas produced from the digestion of blackwater was assumed to be utilized in the CHP unit replacing the Finnish electricity and heat mix. In *Paper III*, the biogas produced from the blackwater digestion was upgraded to transport fuel replacing the use of petrol. In addition, for heat production, biogas was assumed to replace heat produced by natural gas (39%), oil (5%) and wood (56%) reflecting the average heat production in Tampere during the study.

2.5 UNCERTAINTY ANALYZES

Several assumptions have to be made for LCA and SFA. Due to the uncertainties within the selected parameters and scenario assumptions, the sensitivity of the results was tested by a simple scenario analysis in *Papers I, III* and *IV*.

In *Paper I*, the LCA evaluated different raw materials, energy consumptions and transport distances. The sensitivity analysis was constructed as one-at-a-time (OAT) sensitivity analysis. The factors in the sensitivity analysis were chosen based on the uncertainties encountered when using a single parameter value. Moreover, in the sensitivity analysis, the fluctuation of eutrophication factor (the fraction of nutrients ending up in the water bodies) was examined in two alternative scenarios. In scenario 1, it was assumed that all the purified wastewater (100%) will end up in the freshwater ecosystem. In scenario 2, it was assumed that only 15% would do the same (rest retaining in the soil).

In *Paper III*, the consequences of improved nutrient and energy recovery were examined in more detail in the LCA in order to map the climate impacts of alternative management processes in urban systems. For the conventional system, it was calculated what climate benefits would be achieved if the digestate was utilized as such without composting. In blackwater separation, it was studied how the impacts would change if biogas from blackwater digestion was utilized as CHP instead of being used as a transport fuel. In addition, it was studied how the recovery and utilization of nutrients in the AD reject water from both the WWTP and local AD plants would reduce the use of mineral fertilizers.

In the SFA of *Paper IV*, a baseline for nutrient recovery was defined to describe the typical values in northern Finland and Sweden. However, some treatment methods are not currently in use (e.g. urine treatment) and the variation within the processes (such as solid-liquid separation) is high. Thus, an average value was defined for the baseline, and the variation in recovery rates associated with these technologies were considered in the uncertainty analysis. The respective values are presented in the Table 5.

Table 5. *Phosphorus, total nitrogen, and soluble nitrogen (in percentages) recovered from the input material of different treatment processes. Variation range is shown in parentheses.*

Treatment process	P % (recovered)	N % (recovered)	N (soluble) % (recovered)	Reference
Septic tank (sludge)	15 (5–25)	10 (5–15)	3	Olshammar et al., 2015; Malila et al., 2019a
WWTP (sludge)	96	36	22	Malila et al., 2019a; SYKE, 2019
Dewatered sludge	40	40	30	Møller et al., 2000; Hjorth et al., 2010; Luostarinen et al., 2019
Anaerobic digestion (solid and liquid fraction)	100	100	130 (114–161)*	Magdoff and Chromec, 1977
Solid-liquid separation of digestate (solid fraction)	71 (40–90)	32 (17–60)	20 (15–30)	Wäger-Baumann, 2011; Al Seadi, 2013; Borowski and Weatherley, 2013; Ruuhela, 2017; Luostarinen et al., 2019; Malila et al., 2019a
Solid-liquid separation of digestate (liquid fraction)	29 (10–60)	68 (40–83)	80 (70–85)	Wäger-Baumann, 2011; Al Seadi, 2013; Ruuhela, 2017; Malila et al., 2019a
Sludge composting/ chemical treatment + composting	100	30 (25–50)	2	Myllymaa, 2008; Ruuhela, 2016
Digestate/dewatered sludge storage (non-covered)	100	90	84	IPCC, 2006, EMEP/EEA, 2016, Grönroos et al., 2017 (emission factors for bovine manure were applied)
Urine/liquid fraction storage (closed tank)	90 (80–100)	95	95	Karlsson and Rodhe, 2002; Maurer et al., 2006
Urine/liquid fraction central treatment	95 (90–100)	90 (80–100)	90 (80–100)	Marttinen et al., 2015; Etter et al., 2015; Mönkäre et al., 2016; Kaljunen et al., 2021; Simha et al., 2021b

**for WWTP sludge. For blackwater sludge it was assumed to be 110.*

3 RESULTS

3.1 PHOSPHORUS AND NITROGEN RECOVERY

3.1.1 POTENTIAL IN RURAL AND URBAN AREAS

Conventional rural on-site wastewater treatment systems are less effective in nutrient recovery than urban systems (Figs. 6 and 7). This is due to the fact, that in on-site systems in rural areas (*Paper II*), most of the phosphorus accumulates in the soil in soil systems (conventional management) and part of the nutrients retains in the sludge. In urban systems, on the other hand, higher proportion of phosphorus remains in the sludge due the precipitation. However, the current practices of the WWTP are ineffective, especially regarding nitrogen recovery. Source separation systems recover higher amounts of nutrients and in plant applicable form in both rural and urban context, but the increase in nutrient recovery potential per capita is higher in rural systems than in urban systems, mostly due to differences in conventional systems. In addition, the technical solutions studied for the source separation systems were different on an urban and rural scale, which is also reflected in the results. (Figs. 6 and 7).

According to *Paper II*, approximately 48-80% of the excreted phosphorus (excluding the potential in reject waters) and 25-66% of the nitrogen can be recovered when implementing source separation systems in rural areas. This corresponds to four times more phosphorus and over thirty times more nitrogen compared to current practices. In urban areas, the phosphorus recovery potential was 74-84% and nitrogen 14-58%, respectively (excluding the potential in reject waters). This corresponds to 3-13 times higher nitrogen recovery compared to the conventional wastewater management *(Paper III*). (Fig. 6.)

Regarding the total amounts of phosphorus recovered, there was no significant difference between source separation systems and conventional system in urban areas (*Paper III*). The most significant difference is, however, that phosphorus can be recovered in a more valuable and plant-usable form with source separation. The total phosphorus recovery is greatest in the urine separation of rural context (*Paper II*), since the urine separation includes the home composting of feces. In the blackwater separation of urban systems, more than half of the phosphorus could be recovered without utilizing the potential of reject water as such. In both rural and urban areas, the difference in phosphorus recovery between blackwater and urine separation is small, if the potential in reject water is considered (without recycling it back to a WWTP for further treatment).

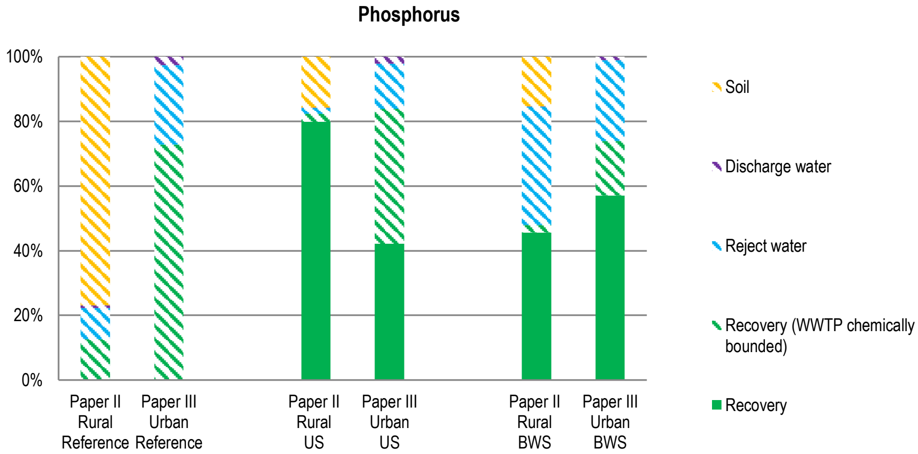


Figure 6 Path of total phosphorus (as a percentage) in conventional and source separating systems of rural and urban areas according to *Papers II* and *III*. Reference = conventional, US = urine separation, BWS = blackwater separation.

Less than a tenth of the nitrogen is recovered in conventional urban systems, while the rest of the nitrogen ends up in water bodies and air as a result of biological nitrogen removal (*Paper III*) (Fig. 7). In rural systems, the recovery is even lower because most of the nitrogen ends up in the soil. Moreover, most of the recovered nitrogen in the composted AD sludge is in organic form and slowly becomes available to plants. If the reject water (both in the source separation and the conventional system) was utilized as such or the nutrients were recovered by implementing new technologies, the recovery of nitrogen would be greater.

The potential to recover nitrogen is clearly higher in source separation systems (Fig. 7). However, the differences between rural and urban areas are small. With urine separation, it is possible to recover more than half of the nitrogen contained in the wastewater. With the blackwater separation, the amount would be even higher, if the capacity of the reject water were utilized. However, if the nutrients in reject waters in WWTPs and ADs were recovered more efficiently, the difference between source separation and conventional practices would be smaller, especially regarding nitrogen.

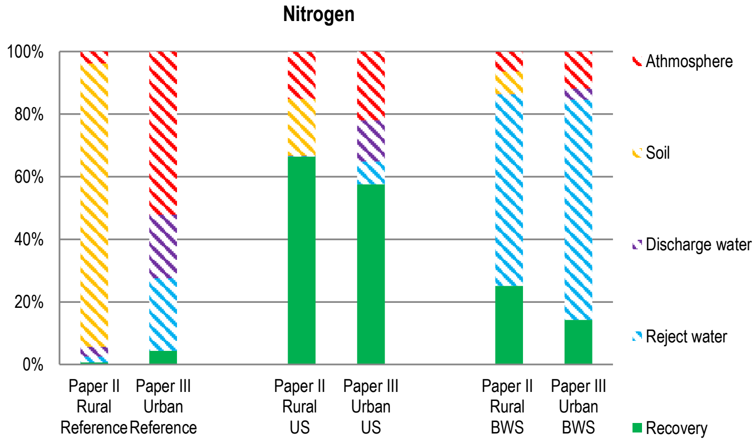


Figure 7 Path of total nitrogen (as a percentage) in conventional and source separating systems of rural and urban areas according to *Papers II* and *III*. Reference = conventional, US = urine separation, BWS = blackwater separation

3.1.2 REGIONAL POTENTIAL

The implementation of source separation systems leads to more efficient nutrient recovery in the studied regions of Northern Finland and Sweden (*Paper IV*) (Fig. 8). Both phosphorous and nitrogen recovery potentials increased significantly (41–81% for phosphorus, 690–860% for nitrogen) in source separation systems in comparison to the reference (conventional) system.

The total phosphorus recovery potential of the reference system was 164 tonnes P/year. Of this, only a relatively small fraction (3.9%) came from the 'rural non-sewered' areas while about 45% came from the 'peri-urban sewered' areas. The phosphorus recovery potential is low in rural areas due to on-site treatment, where most of the phosphorus is discharged and accumulated in the soil. The phosphorus recovery potential of source separating scenarios was 297 tonnes P/year (blackwater separation) and 233 tonnes P/year (urine separation). The highest increase in phosphorus recovery was achieved in rural areas. Moreover, 44–62% and 51–74% of recovered phosphorus was in more plant-available form in northern Finland and in northern Sweden, respectively.

In the conventional system, the total nitrogen recovery potential in northern Finland and Sweden was 195 tonnes N/year, from which 24% was in plant-available (soluble) form. In source separation systems, the recovery potential of nitrogen was 1,880 tonnes N/year (blackwater system) and 1539 tonnes N/year (urine separation). Also, the fraction of soluble nitrogen from the total nitrogen was higher (average 93%), reflecting the substantial potential for recovery of easily plant-available nitrogen.

The variation in the results (error bars in Fig. 8) are due to the range of minimum and maximum rates used for nutrient recovery in the treatment processes (sensitivity analysis). The variation (in percent) is greatest in the nitrogen recovery in the reference system, where the total nitrogen recovery yield would result in a 24% lower, or 57% higher recovery yield compared to the average value used. The high variation is a consequence of the high variability in solid-liquid separation technologies and sludge composting. Instead, in the blackwater and urine separation scenarios, the variation is smaller in percentage because the nutrients in the liquid fraction are considered a resource: blackwater separation, min. -0.4%, maximum, +1.5%; and urine separation, min. -11%, maximum, +14%. For phosphorus recovery, the variation in the reference system is min. -26%, max. +12%, for blackwater separation, min. -16%, max +5% and for urine separation, min. -26%, max. +14%.

In addition to nutrient potential, it was estimated how the recovered nutrients would theoretically replace the use of mineral nutrients in the region. In Lapland, wastewater-derived nutrients could replace 38–49% of mineral phosphorus fertilizers, in North Ostrobothnia 13–17% and in Norrbotten 49–65%. In the case that soluble nitrogen would replace the use of mineral nitrogen fertilizers in agriculture, source separation could cover 13–16% of the need for mineral nitrogen in Lapland, 5–7% in North Ostrobothnia and 50–60% in Norrbotten.

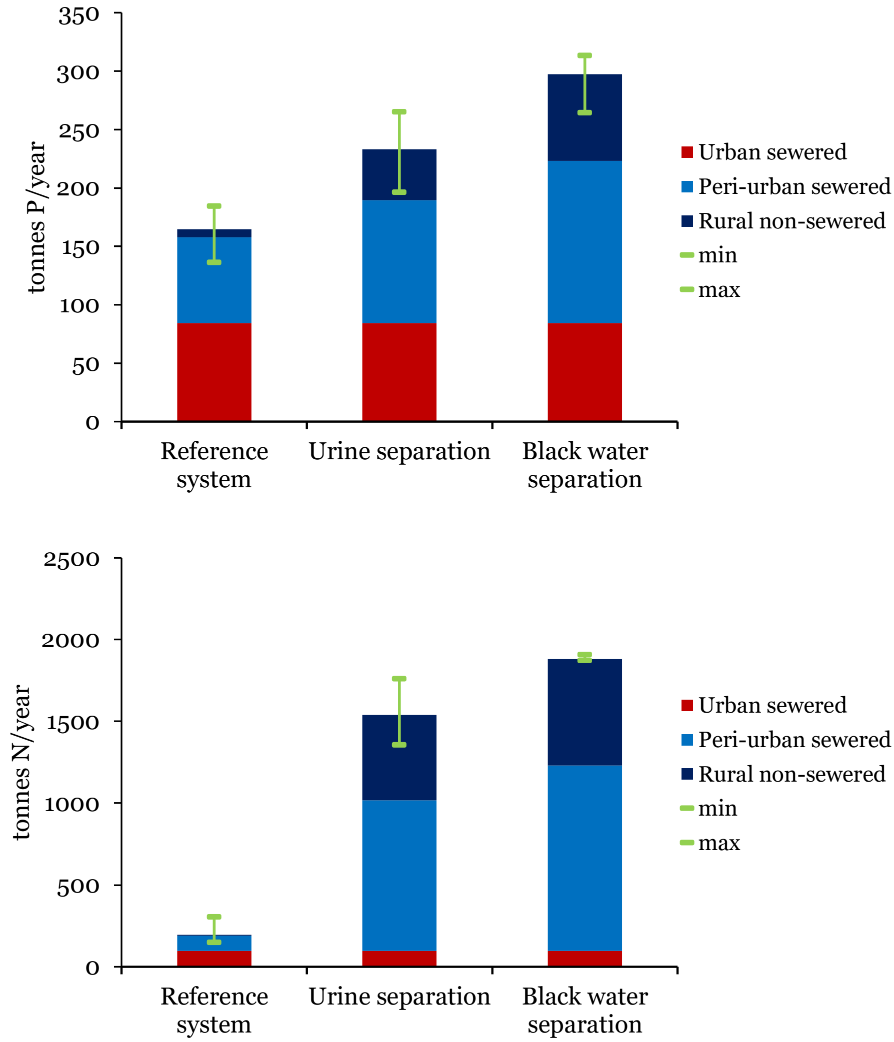


Figure 8 The recovery potential of phosphorus and nitrogen (P/N tonnes/year) in the three studied regions in northern Finland and Sweden. The fluctuation in error bars reflects the variation in the technologies implemented.

3.2 LIFE CYCLE IMPACTS OF SOURCE SEPARATION

3.2.1 CLIMATE CHANGE IMPACTS IN URBAN AND RURAL SYSTEMS

According to the results, the climate benefits achieved are higher in urban areas when source separating systems are implemented compared to conventional systems. In rural areas, the impacts remain the same or are even higher (Fig. 9). However, if the avoided emissions of fertilizer and energy use are not considered, the difference between the source separation and the conventional system fades in most of the cases studied. Nonetheless, in urban areas climate impacts of blackwater scenario are lower, even if the avoided emissions are not considered.

The highest variation between the treatment systems was in rural areas in *Paper I*. When nutrient recovery was in the focus of the study (*Paper II*) the differences between the studied systems were small. In the rural on-site systems, the main contributors to the climate impacts were the transportation distances of sludge (in the Fig. 9: sludge/WW treatment at WWTP included the transportation of sludge) and the direct emissions from the use of on-site systems. Moreover, infrastructure accounted for a large share of emissions due to the transportation of sand to soil systems and the assumed service life of the systems. On the contrary, in urban systems, the main contributor was the treatment of wastewater and sludge in a municipal WWTP.

In *Paper I* the treatment and transportation of sludge from an on-site property to a municipal WWTP caused a significant portion of the climate impacts, and therefore the urine separation in dry toilet was clearly the best of all alternatives. In rural areas, transportation distances vary, which contribute to the climate impacts of the household. When applying blackwater separation in low flush or vacuum toilets, the water consumption is significantly lower, which reflects to the capacity and emptying interval of the holding tank. The results in *Paper I* show that the transportation of blackwater (implemented without a vacuum or a low-flush toilet) had the highest climate impact due to the transportation of blackwater and its conventional treatment at a municipal WWTP. If the sludge had been processed in the vicinity to recycle nutrients, the overall environmental impacts would have been substantially lower, as *Paper II* shows.

In urban areas, vehicle transportation of sludge or wastewater was not included as the fractions ran along the pipeline. In the urban urine separating system, the transportation of hygienized urine had a significant effect on emissions, as no further processing of urine was included in the study (Fig. 9; use of fertilizer includes the transportation of nutrients). However, reducing the volume of urine on site should be considered by implementing new technologies, such as membranes.

The use of recovered fertilizers was lower in blackwater systems than in urine separation systems (*Papers II* and *III*). This difference is due to the assumption, that the reject water from blackwater digestion was circulated to the WWTP, and its potential as a fertilizer was not included. However, if the liquid fraction of anaerobic digestion (reject water) or the nutrients contained therein were utilized as such or in processed form, the impacts from fertilizer use would be greater as well as the avoided emissions.

Overall, the production, raw materials and installation of the equipments had only a relatively small impact on the climate in *Paper I*. In *Paper II*, on the other hand, the emissions from on-site treatment facility (toilet system and sand filter) caused altogether 58-80% of the climate emissions (Fig. 9; infrastructure). However, in *Papers I-III,* there is not much relative fluctuation in the role of infrastructure, and the difference between the *Papers* are due to differences in the assumptions and data used.

Climate emission from day-to day use of the property are higher in on-site systems since no treatment is located in urban households. However, the emissions from on-site systems are not well known and in *Papers I* and *II* the values were calculated based on literature and expert opinions. In both papers, emission calculations were based on emissions from municipal WWTPs. Even though the data used for nitrogen removal efficiency has improved resulting in higher nitrous oxide emissions, emissions are smaller in *Paper II* than in *Paper I.* This is due to the assumption made in *Paper II,* that on-site emissions were assumed to be half of the corresponding emissions from the WWTP due to the assumed ineffiencies at the soil on-site systems. However, nitrous oxide emissions were not expected to be formed from nitrogen accumulation in the soil system, making the on-site systems climate impact assessment incomplete in both *Papers (I* and *II).*

In urban areas, emissions from WWTP and sludge processing had the highest contribution to the climate impacts. The GHG emissions of the WWTP are largely derived from nitrous oxide emissions from the nitrogen removal process. With source separation, climate emissions are reduced by about a quarter, when taking into account recycled fertilizer products and energy produced, as well as avoidable emissions related to their utilization, such as the production and use of mineral fertilizers and fossil fuels. The reduction is largely due to the reduced need for wastewater treatment, especially the energy-intensive nitrogen removal from wastewater, but also to the more efficient nutrient management, as they are not mixed and diluted with other wastewaters. As a result of the recovery of nutrients in the wastewater, the nitrogen load to the municipal WWTP is reduced to less than one-fifth and the phosphorus load to about one-third, decreasing the climate emissions from wastewater treatment.

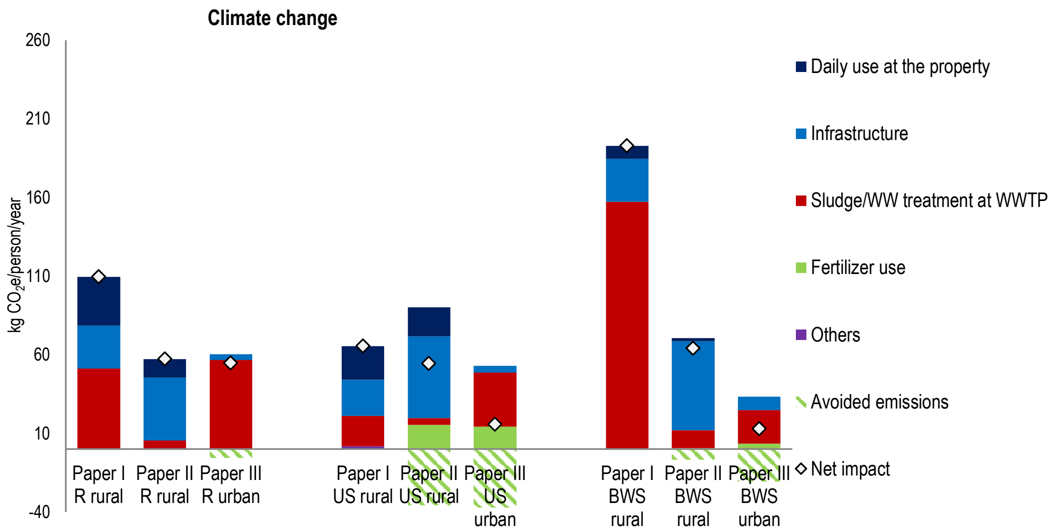


Figure 9 Climate change impacts (kg CO_2 eq/person/year) in *Papers I-III*. Explanations for abbreviations: R = reference (conventional), US = urine separation, BWS = blackwater separation. Daily use at the property includes emissions from the toilet and treatment system and its energy consumption. Infrastructure includes the construction of the systems and materials needed. Sludge/WW treatment at WWTP includes the transportation of WW or sludge to treatment, digestion, and composting. Fertilizer use includes the use of recovered fertilizers and their transportation. Others include other type of processes. Avoided emissions include substitutions in energy and fertilizer use.

3.2.2 EUTROPHICATION FROM URBAN AND RURAL SYSTEMS

With regard to eutrophication, it is clearly shown, that the impacts of source separation systems, where nutrient-rich fractions are separated and utilized instead of being treated in on-site systems or/and WWTPs, are substantially lower (Fig. 10). Wastewater from the treatment of sludge or blackwater from a municipal WWTP discharged directly into water systems is likely to cause a greater eutrophication impact than source separation. Moreover, treating wastewaters at municipal WWTP shows clear benefits compared to rural on-site systems (Fig. 10; reference). The daily use at the property (*Papers I* and *II*) had the largest contribution to eutrophication impacts due to the assumed eutrophication impacts of soil systems, which at the same time has substantial uncertainty.

The eutrophication impacts were lowest in blackwater separation systems where the source separated blackwater was digested in local plants (*Papers II* and *III*). The blackwater separation scenario in *Paper I* had relatively high emissions from blackwater treatment in WWTP. Also, the eutrophication impact is even higher than in conventional urban systems (*Paper III*). This is because *Paper I* assumed a higher occupancy rate (85%) in the settlement, compared to the urban system (67%). Also in the urban system, the efficiency of the new modern wastewater treatment process was substantially higher in nitrogen removal (80%) compared to that used for WWTP in *Papers I* and *II* (61-63,5%).

The fertilizer use of urine separation had higher eutrophication impacts compared to the utilization of blackwater. This is because in both *Papers*, the nutrient content recovered and utilized was higher in urine separation. The impacts of fertilizer use and the avoided emissions almost overturned each other due to the avoided emissions from the use of mineral fertilizers.

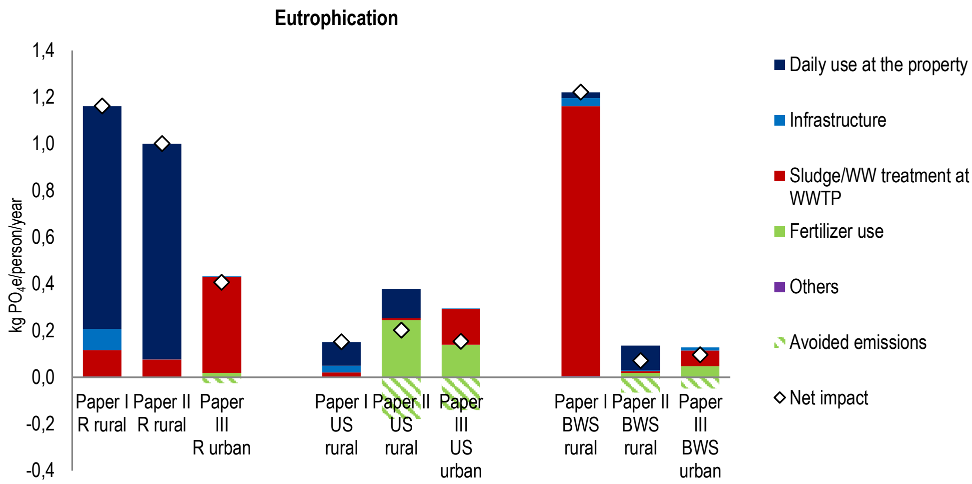


Figure 10 Eutrophication impacts (kg PO_4 eq/person/year) in *Papers I-III*. Explanations for abbreviations: R = reference (conventional), US = urine separation, BWS = blackwater separation. Daily use at the property includes emissions from the toilet and treatment system and its energy consumption. Infrastructure includes the construction of the systems and materials needed. Sludge/WW treatment at WWTP includes the transportation of WW or sludge to treatment, digestion, and composting. Fertilizer use includes the use of recovered fertilizers and their transportation. Others include other type of processes. Avoided emissions include substitutions in energy and fertilizer use.

3.2.3 ACIDIFICATION FROM URBAN AND RURAL SYSTEMS

Acidification was considered only in *Papers II* and *III*. The results show clearly, that with source separation systems, the risk of acidification is higher (Fig. 11). The evaporation of ammonia from applying urine and blackwater as fertilizers contributed mostly to the impact. The ammonium nitrogen in urine and blackwater evaporates easily as ammonia, causing acidification emissions from the daily use at the property (rural areas) and their fertilizer use. Both *Papers* used advanced spreading techniques, but the emission factors used for evaporation were different. As a result of lower nitrogen losses connected to advanced spreading techniques, such as deep injection, the amount of soluble nitrogen remains higher in the soil, which in turn may increase the risk of nutrient leaching and cause eutrophication.

In *Paper III*, the acidification impacts of blackwater separation are higher compared to urine separation due to differences in storage and field application practices of liquid (urine) and solid (blackwater sludge) fractions and is not related to urban and rural context. These results may better reflect reality compared to results in *Paper II,* which used the same emission factors for the application of urine and blackwater and did not include storage of the blackwater based solid fraction. The risk of ammonia evaporation from the storage of solid fractions is high due to the high content of soluble nitrogen after digestion.

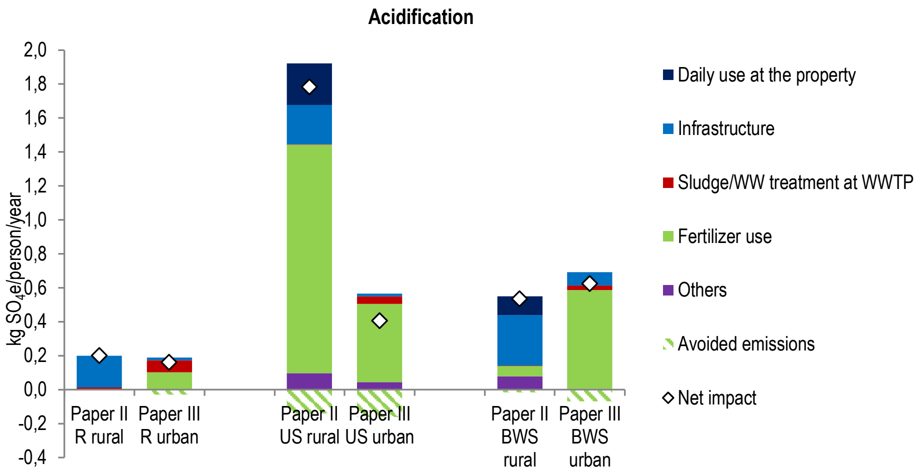


Figure 11 Acidification impacts (kg SO_4 eq/person/year) in *Papers I-III*. Explanations for abbreviations: R = reference (conventional), US = urine separation, BWS = blackwater separation. Daily use at the property includes emissions from the toilet and treatment system and its energy consumption. Infrastructure includes the construction of the systems and materials needed. Sludge/WW treatment at WWTP includes the transportation of WW or sludge to treatment, digestion, and composting. Fertilizer use includes the use of recovered fertilizers and their transportation. Others include other type of processes. Avoided emissions include substitutions in energy and fertilizer use.

3.2.4 REGIONAL IMPACTS

In *Paper IV*, the whole region of Northern Finland and Sweden was studied by applying the LCA results of *Papers II* and *III*. According to the results, the estimated climate change impacts of the conventional system were 9700 tonnes of CO_2 eq./ year in Lapland, 22,800 tonnes in North Ostrobothnia and 13,700 tonnes in Norrbotten (Fig. 12). Implementing blackwater and urine separation would result in 23.8% and 25.6% lower CO_2 eq./ year (respectively) emissions in total of the three regions studied.

The impact of source separation on climate change was greatest in the Norrbotten region. This is due to the fact, that in Norrbotten 45% of the population lives in peri-urban areas, while in North Ostrobothnia and Lapland 28% and 31% respectively. Rural areas accounted for 20% of climate emissions in the conventional system, 30% in blackwater separation and 26% in urine separation. In peri-urban areas, on the other hand, about 34% of the climate change impacts were due to the conventional system, 10% to blackwater and 13% to urine separation. The highest share of climate emissions originated from urban households (45–60%).

Overall, climate change emissions could be reduced by 18–32%, depending on the region, if source separation systems were implemented in both rural and peri-urban areas. However, the results indicate a risk of increased climate impacts in sparsely populated rural areas if source separation systems (both urine separation and blackwater) are implemented, but substantially lower emissions for peri-urban areas. Moreover, the separation of blackwater caused higher climate impacts than urine separation. This is because in areas with urine separation (rural areas), the use of dry toilet reduced the transportation and treatment of brown and grey water at the WWTP.

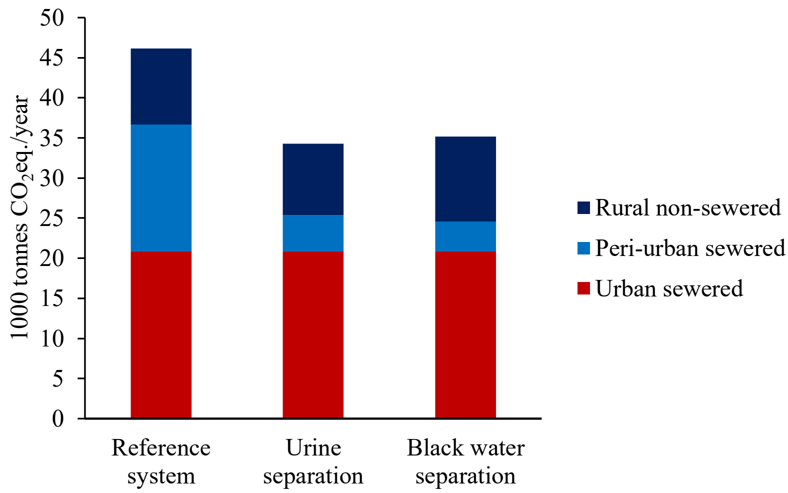


Figure 12 Climate change (1000 tonnes CO_2 eq./year) in urban, peri-urban and rural areas of northern Finland and Sweden.

The total eutrophication impact of the conventional system was estimated to be 435 tonnes of PO_4 eq./ year (Fig. 13). By implementing source separation systems in peri-urban and rural areas, eutrophication impacts can be reduced by 46% and 54% in Lapland, 45 and 54% in North Ostrobothnia and 41% and 52% in Norrbotten for blackwater and urine separation respectively. The reduction in eutrophication is due to improvements in rural on-site wastewater systems and improved nutrient recovery and reuse (*Papers II-III*).

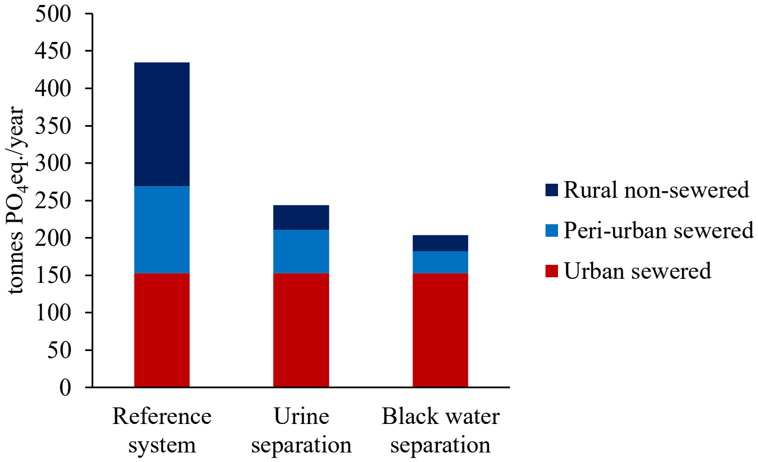


Figure 13 Eutrophication (tonnes PO_4 eq./year) in urban, peri-urban and rural areas of northern Finland and Sweden.

The total acidification impact in the three regions was 168 tonnes of SO_2 eq./year in the conventional system (Fig. 14). If implemented, the source separation scenarios would increase acidification impacts (101% in blackwater separation and 190% in urine separation) compared to the conventional system, with the largest contributors being ammonia emissions from storage and field application of sludge in (*Papers II* and *III*).

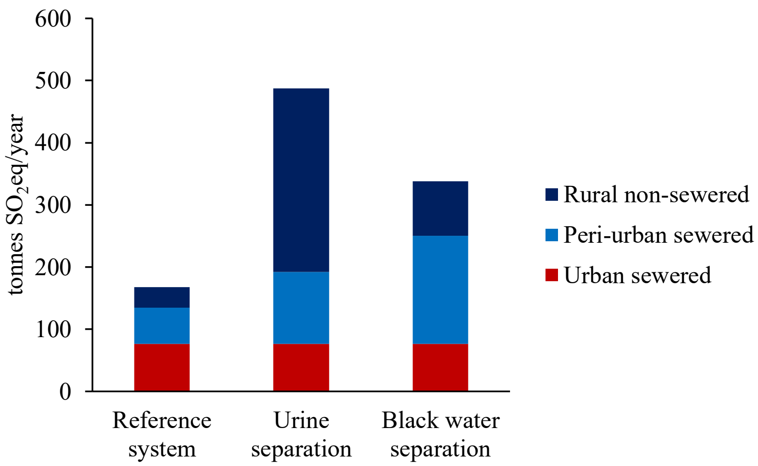


Figure 14 Acidification (tonnes SO_2 eq./year) in urban, peri-urban and rural areas of northern Finland and Sweden.

3.2.5 UNCERTAINTY OF RESULTS

In *Papers I* and *III*, the uncertainties of the LCA results were analyzed in respect to the assumptions made in the calculations. *Paper IV* also included the impact of different technologies on nutrient recovery, and the results are presented in section 3.1.2.

In *Paper I*, the sensitivity of the eutrophication impact was analyzed as well as the chosen factors (different raw materials, energy consumptions and transport distances) for on-site systems. The eutrophication impact was found to vary the least with blackwater separation (0-0.13 kg PO_4eq./person/year) and the variation being highest in on-site soil systems (0-1.3 kg PO_4eq./ person/year). This describes well the importance of the circumstances at the property in relation to the eutrophication emission factors used. In addition, the carbon footprint was smaller in the blackwater system, when the low flush toilet, tank size and transport distance of sludge were changed. These changes would reduce the carbon footprint of blackwater separation by 19-30%, when considered separately. The option of looking at all these changes together was not evaluated in *Paper I*.

The results of the uncertainty analysis in *Paper III* show that the assumptions about avoided emissions have a significant effect on climate impacts and that even higher avoidable impacts can be achieved through careful planning and management of the system (Table 6). The highest avoided impacts could be achieved with blackwater separation if biogas was utilized as a transport fuel and the reject water from the blackwater digestion was used as such (not pumped to a WWTP for further treatment).

If the nutrients in reject waters were recovered and reused, the overall climate impact could be further reduced in all systems. On the other hand, if the nutrients in 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 in producing more easily applicable nitrogen fertilizers would replace the use of mineral fertilizers. Utilizing biogas as a transport fuel, leads to higher emission savings compared to CHP production, as the offsetting benefits of CHP production are smaller.

Table 6. *The total impact of avoided processes (default and alternative avoided processes) in the reference system, blackwater separation and urine separation.*

Default avoided processes	Avoided emissions (t CO_2 eq./a)	Impact on total emissions (%)	Alternative (and default) avoided processes	Avoided emissions (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 %	-2 %
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 %**
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			Reject waters from	-71	-5 %	-5 %

from Sulkavuori AD not utilized			Sulkavuori AD used for fertilization			
In total	**-969**	**-71 %**		**-1 047**	**-76 %**	**-6 %**
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 %**		**-1 101**	**-127 %**	**-65 %**

4 DISCUSSION

4.1 ENVIRONMENTAL IMPACTS

4.1.1 NUTRIENT RECOVERY POTENTIAL

The results of this study indicate a substantial nutrient recovery potential with source separation systems. Per capita, highest increase in nutrient recovery can be reached in rural areas. In urban areas, considerable improvements in nitrogen recovery can be achieved by source separation, especially if all treatment fractions are efficiently utilized. Both urine and blackwater separation achieve substantial improvements in nutrient recovery, the potential being largest with blackwater separation combined with advanced practices.

The results of this study strongly support previous studies that suggest that the highest potential for nutrient recovery in urban and peri-urban context could be achieved by source separating systems, either by blackwater or urine separation (Kjerstadius et al., 2015; Kjerstadius et al., 2017; Wielemaker et al., 2018; Turlan, 2019). According to *Paper III*, nutrient recovery is many times greater in both source separating systems compared to conventional system, even if the nutrient potential in reject waters of blackwater digestion is not utilized.

With improved recovery of wastewater nutrients, the use of mineral fertilizers could be decreased in agriculture. In Finland, approximately 11,400 tonnes of mineral phosphorus fertilizers and 147,000 tonnes of mineral nitrogen fertilizers were sold to farms in 2021 (Natural Resources Institute Finland, 2022a). By using source separating techniques, approximately 18,000-22,000 tonnes of nitrogen and 2,100-2,900 tonnes of phosphorus per year could be recovered (calculated roughly on a Finnish scale based on *Paper IV)*. This corresponds to the total amount of nitrogen produced by pigs and 60-80% of the phosphorus produced annually by poultry in Finland (calculated by the Natural Resources Institute Finland, 2022b and Luostarinen et al., 2017). However, despite the already substantial phosphorus potential of municipal sewage sludge (in the conventional system), this potential has not been realised. Overall, 47% of sewage sludge is used in agriculture, but regional differences are large (Vilpanen and Seppälä, 2021). In some areas, such as Lapland, sewage sludge is not currently used in agriculture (Vilpanen and Seppälä, 2021). This same contradiction applies to the nutrient potential presented in *Papers II* and *III,* as although the nutrient potential also exists in conventional systems, it does not realize in practice, since only part of the sludge is utilized as plant nutrients. Instead, nutrients are applied in large amounts per area for land improvement and landscaping purposes, where nutrients are at risk of leaching into water bodies. Thus, attempts to improve nutrient recovery will not be enough to improve environmental performance if the nutrients recovered are not used efficiently, for example in agriculture.

In addition to improved nutrient recovery source separation decreases the total amount of harmful substances, such as heavy metals and microplastics, ending up in the soil during application. For example, grey water contains majority of the heavy metals found in wastewater (Simha, 2021), and separating the blackwater or urine excludes the input of the metal fraction from the grey water. However, the concentration of harmful substances, such as hormones in the end products may be high. For example, urine contains most of the pharmaceuticals and hormones found in wastewater (Udert et al., 2006), but for example, Viskari et al. (2018) found that, apart from progesterone, the concentrations of all extractable pharmaceuticals and hormones in soil fertilized with source separated urine remained below the detection limit (Viskari et al., 2018).

The concentrations of most metals in source separated blackwater are generally much lower than in sewage sludge, and hence the utilization of blackwater is more effective in reducing metals in agriculture (Tervahauta et al., 2014). However, the blackwater fraction contains most of the pathogens in the wastewater and the risk of pathogens in the end products depends on the treatment technologies of the separated blackwater. Significant inactivation of pathogens can be achieved, for example, by urea treatment (Nordin et al., 2009; Fidjeland et al., 2015) which has proven to be a robust option to the safe recycling of plant nutrients, as well as thermophilic anaerobic digestion applied to both blackwater (Moerland et al., 2020) and sewage sludge (Zhao and Liu, 2019).

However, the direct impacts of harmful substances on humans via sludge-fertilized crops have been reported to be minor (Viskari et al., 2018; Ylivainio et al., 2020), but the impacts of harmful substances on soil or biota and their reflection on carbon sequestering, for example, are not well known. Harmful substances may harm the environment by accumulating in soil or biota or by leaching into groundwater (Dolar et al., 2021; Selonen et al., 2020). Once persistent and accumulating harmful substances enter the soil, it is no longer possible to remove them from the environment. Also, changes in the species composition of soil invertebrates are known to affect soil nutrient fluxes (Kremen, 2005). Moreover, some studies suggest that the ecotoxicity of digestate (Teglia et al., 2010; Pivato et al., 2016; Tigini et al., 2016) may affect the carbon sequestration of soil biota.

Nevertheless, the risk of harmful substances needs to be investigated further. Instead of utilizing the source separated fractions as such, new processing and recovery technologies can reduce the risk of harmful substances (Lehtoranta et al. 2021a) and at the same time, reduce the volume of fractions to a more easily transportable and field applicable form. Processing of nutrients also allows them to be mixed with other recycled nutrients and with manure. In Finland, part of the recyclable nutrients, such as manure and sewage sludge, are produced in relatively small areas and beyond the needs of the regions. For example, in the case of wastewater most of the sludge is generated in densely populated Uusimaa (Vilpanen and Seppälä, 2021), where there is little agricultural land for its application. Therefore, the efficient recycling of nurtients, requires the processing of nutrient fractions to ensure the safe and efficient recycling of nutrients and enable their use in places, where nutrients are actually needed.

Even though this dissertation has not examined the techniques to improve the nutrient recovery in the current practice of centralized municipal WWTPs, it must not be ignored. In recent years, wastewater nutrient recovery technologies have been studied and new technologies have been diligently developed also in Finland, for example, the RAVITA and NPHarvest process methods (Rossi et al., 2019; Kaljunen et al., 2021). With new technologies and their combinations, even 1.3 times greater phosphorus recovery and three times greater nitrogen recovery could be achieved in centralized WWTPs, compared to current technologies (Lehtoranta et al., 2021a). At the same time, new technologies can increase the usability of nutrients and reduce the accumulation of hazardous substances.

4.1.2 CLIMATE IMPACTS

The climate change impacts addressed in *Papers I-III* show the possibilites to reduce the climate change impacts in urban areas, but in rural areas the impacts were smaller or higher compared to conventional systems. The results indicate that climate impacts and nutrient recovery potential, especially, are strongly related to the system design and the decisions made in the society. Moreover, source separation does not in itself decrease climate impacts, but allows for more efficient nutrient recycling. Achieving the climate benefits of improved nutrient recovery requires the realization of avoided impacts of fertilizers and energy.

The results of this study are in line with previous studies on source separation reviewed in Lam et al. (2020). For example, Kjerstadius et al. (2015; 2017) concluded that in urban areas, the carbon footprint decreased with source separation, mainly due to increased biogas production, increased replacement of mineral fertilizers in agriculture, and reduced nitrous oxide emissions from wastewater treatment, which is in line with this study. Moreover, other studies imply that, increased climatic benefits can also be achieved with combined collection and treatment of food waste and blackwater (Remy and Jekel, 2012; Kjerstadius et al., 2015; Kjerstadius et al., 2017). Therefore, it can be assumed that the possibilities to utilize food waste generated in the area would result in additional climate benefits, but only if it would lead to a higher rate of food waste collection and anaerobic digestion.

However, a literature review of LCA studies on source separation systems reveal variability in the estimated climate impacts. While some studies report that source separation systems have lower carbon footprints than conventional wastewater treatment (Remy and Jekel, 2012; Spångberg et al., 2014; Lam et al., 2015; Kjerstadius et al., 2015; 2017), others show the opposite (Thibodeau et al., 2014) or no difference (Tidåker et al., 2006a).

The main contributors to climate change in rural applications have been reported to be sludge transportation, nitrous oxide emissions from storage in collection systems and field application of recovered blackwater sludge or urine, as well as the construction and maintenance of on-site systems. In conventional wastewater treatment, on the other hand, the largest contributions originate from purification processes aimed at removing nutrient and organic matter.

In total, the climate impacts in sparsely populated northern Finland were smaller in source separating systems compared to the conventional system, according to a streamlined analysis in *Paper IV*. However, the assessment of the climate impacts of regional source separation systems and improved nutrient recycling would require a more systematic and comprehensive life cycle impact analysis. For example, transportation distances and logistical solutions (e.g. location of regional treatment facilities) are key variables that should be further studied in sparsely populated areas, as *Paper I* reflects. The contribution of transportation to climate impacts has also been shown by Dixon et al. (2003), Benetto et al. (2009) and Turlan (2019). The transportation of sludge and the location of its processing facilities is one of the critical questions for rural wastewater management that needs to be investigated more throughoutly. Therefore, in order to assess the transport distances required for source separation and the need for new regional treatment facilities, the issue should be addressed through comprehensive regional logistics modeling. Thus, the question of sludge treatment in centralised versus decentralised plants is essential.

In this study, the climate impacts of conventional WWTPs were not further analyzed, except for the uncertainty analysis in *Paper III*. The results reflect similar findings to those of Maktabifard et al. (2022) has reported from the Baltic Sea region. According to Maktabifard et al. (2022), the carbon footprint of WWTPs could be reduced by up to 27% by selling biofuel, electricity and fertilizers. They also found out, that direct emissions had the highest contribution (70%) to the total carbon footprint of WWTPs, while energy consumption dominated more than 30% of total indirect emissions. In conclusion, the decrease in the nutrient load of the WWTP results in lower climate emissions due to the reduced use of chemicals and nitrous oxide emissions, which supports the source separation of wastewater.

However, it is important to note that avoided emissions from energy and substituted mineral fertilizers (*Paper III*) will not be achieved unless their use is reduced in the same proportion (IPCC, 2014). The actual emission benefits depend on how avoidable emissions will realize, which depends heavily on societal policies and decision-making processes. For example, if the ongoing reform of national fertilizer legislation (Ministry of the Agriculture and Forestry, 2020) ends up restricting the future use of wastewater-based nutrients in Finland, the benefits of the assumed replacement of mineral fertilizers will be lost. The same applies to the energy produced. The full realization of benefits usually requires the introduction of new policies and policy instruments, as well as good planning and management. Moreover, the time frame for avoided emissions is relevant to the results, as the benefits achieved are expected to change over time as society changes.

4.1.3 EUTROPHICATION AND ACIDIFICATION

Decreased eutrophication and increased acifidication impacts in the implementation of source separation systems were found in *Papers I-III*, which have also been recognized in some other studies (Tidåker et al., 2006; Spångberg et al., 2014). These impacts were mainly due to improved nurtient recovery and reuse, with less nutrients ending up directly in water bodies. Moreover, compared to sludge from WWTPs, the risk of acidification from the storage and application of source separated nutrients is higher due to the higher soluble nitrogen content of the fractions.

In general, eutrophic and acidific emissions associated with the use of fertilizers are difficult to assess, due to the variation in weather conditions, soil features and surface forms, proximity to watersheds, and the techniques and cultivated plants used. Both impacts rely heavily on several agricultural pressures and the physical attributes of the catchments (Dupas et al., 2015). Moreover, the impacts of the use of mineral fertilizers may differ from those of recycled fertilizers due to their features and application methods. Studies comparing the use of mineral fertilizers and organic fertilizers did not show statistical differences in yields, but the use of organic fertilizers had lower emissions and energy consumption compared to mineral fertilizers (Horn et al., 2020; Kyttä et al., 2021). All in all, LCA results regarding eutrophication and acidification impacts should be critically viewed (Morelli et al., 2018).

In *Paper I*, the eutrophication emission factor from on-site plants was based on an assessment on what proportion of the nutrients end up in freshwater ecosystem from the discharge water. This was an estimation based on expert opinions, as no measured data was available. Sensitivity analysis of the share of nutrients entering the water bodies showed that the main conclusion did not change, and the urine-separating dry toilet was superior compared to the other systems studied. Moreover, if the impacts of implementing source separation at the regional level are at the core of the study, a catchment scenario model would further contribute to the evaluation of the eutrophication impacts and changes in sludge and wastewater nutrient use. In addition to model changes in nutrient input and leaching, the effect on the input of harmful substances into soil systems could be included.

In general, wastewater treated in non-sewered areas puts more pressure on water bodies than wastewater treated in centralized systems (Vienonen, 2007). If source separed nutrients are used for fertilization (for the actual need of the plants), the impacts on eutrophication can be expected to decrease if the use of mineral fertilizers is reduced. Especially in rural areas, the introduction of source separation systems will reduce the load of organic matter, nutrients, and pathogens on receiving waters, as only grey waters would be treated on-site. However, reducing eutrophic emissions requires appropriate storage and application of recovered nutrients to achieve the benefits.

The acidification emission factors used for storage and field application in *Papers II* and *III* may overestimate the ammonia emissions of blackwater digestate and urine. The use of manure-based conservative values for blackwater digestate has led to similar results found, for example, in Thibodeau 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 blackwater require further processing and appropriate storage and spreading practices to keep ammonia evaporation to a minimum (Webb et al., 2005).

4.2 RESTRICTIONS ON DATA IN THE ENVIRONMENTAL IMPACT ASSESSMENT

4.2.1 SYSTEM BOUNDARIES AND DATA QUALITY

In wastewater management systems, the life cycle environmental impacts can differ greatly depending on the treatment techniques of different fractions, their use as fertilizers e.g. field or land application, and their ability to replace nutrients in the farmland. In this study, the assumptions made for system boundaries were based on expert opinions on well planned, adequately sized, and maintained on-site and urban systems. However, in practice, this might not be the case and the systems may, for example, be undersized and/or have maintenance deficiencies. This would result in shortcomings in the systems and cause environmental effects that have not been assessed in this study. Furthermore, the results of the reference systems in this study cannot be directly generalized to describe the state of wastewater management.

The input data used for raw materials and transportation etc. were mainly based on the literature and the ecoinvent database. One of the biggest uncertainties related to data lies within the estimations of on-site emissions. Research data on emissions from on-site systems are limited, and in this study, the emissions were assessed based on the emissions from WWTP (see section 2.4.3). In Finland's national greenhouse gas inventory, the role of uncollected wastewater in sparsely populated areas is more than ten times higher per person, compared to sewered systems (calculated from Statistics Finland, 2021 and Lapinlampi, 2021). The estimate is highly uncertain, as the default values used have been defined for Europe (IPCC, 2006) without considering the weather conditions in Finland or distinguishing between different on-site wastewater management systems. Therefore, it is likely that the GHG emissions reported in the national GHG inventory from sparsely populated areas are overestimated. To gain a more accurate understanding on the impacts of sparsely populated areas, more measured data are needed.

The emissions related to the storage and land application of nutrients, such as leaching and emissions to air (eutrophic and acidific emissions), are strongly related to the actual environmental circumstances, such as climate conditions, soil properties and distances from water bodies, as well as spreading techniques and types of storage used. In this study, the data used for emissions from storage and spreading are based on manure related data, since no accurate data were available. This caused uncertainties in all of the impact categories studied. In addition, if the source separating wastewater systems were to become more common and the recovered nutrients were to be utilized in agriculture, more advanced techniques would be required to process nutrient-rich fractions to ease their storage, transports and spreading practices. The environmental impacts of these actions were not included in the LCA assessments of the study.

In *Paper IV*, the environmental impacts were synthesized by utilizing the information from *Papers II* and *III*. The results of the study are clearly a rough estimation and need to be reviewed critically. In order to obtain a more comprehensive and precise analysis, detailed information especially on on-site systems and transport distances would have needed to be applied for LCA. However, the results indicate that the role of sparsely populated area is important for nutrient recovery, but the life cycle impacts of recovery and reuse would require more detailed analysis to decide on optimal implementation.

This study did not consider the potential for improving the nutrient recovery of current processes at the centralised wastewater treatment plant. In order to address the overall potential of decentralised source separation systems on a Finnish scale, the possibilities to improve the processes of centralized systems should be considered.

4.2.2 METHODOLOGICAL ASPECTS

In this study, SFA and LCA were used to analyze the nutrient recovery potential and the environmental impacts of source separation systems compared to conventional system in rural and urban areas in Finland. As stated by Antikainen (2007), SFA itself is not a sufficient method for environmental impact analysis and it needs to be accomplished with LCA, as has been done in this study. However, LCA and LCA-based studies are highly dependent on the boundaries of the study, the assumptions made, and the data used, which makes it complicating to generalize the results.

This thesis combined the results of four studies. There were differences in the design of the scenarios and their system boundaries, as well as in the data and assumptions made. Therefore, care must be taken in the interpretation of the results, as the results of the studies are not directly comparable, although they are presented together in same figures. Hence, the emissions of the life cycle stages are not comparable between the *Papers*, but the relationships between them in each *Paper* are.

There were also differences in the research questions presented in the *Papers,* resulting in different definitions of system boundaries and methods used. In *Paper I*, the environmental impacts of on-site systems were compared and the impacts on nutrient recovery and reycling were not included. Instead, in *Papers II-III*, improving nutrient recovery and its impacts were the main question to be answered. Therefore, the impacts of improved nutrient recovery were analyzed by CLCA and the avoided impacts were included in the study to reflect the potential for nutrient recovery to reduce the impacts. The CLCA is typically recommended as an approach in decision making and policy (Weidema et al., 2018) and has been found useful, for example, in situations where a change is introduced to a WWTP and there is an interest to show the environmental impact of the change (Heimersson et al., 2019). The ALCA should be considered as an approach when there is an interest in knowing how much of the global environmental impacts is the responsibility of that activity (Ekvall, 2019). Eventually, the research question defines the frames and approach for how LCA should be performed. However, there is still an ongoing debate on methological aspects of ALCA and CLCA and the differences related to them (Ekvall et al., 2016; Ekvall, 2019; Schaubroeak et al., 2021).

The marginal data are commonly used in CLCA and average data in ALCA, but there is no clear consistency in their use (Ekvall et al., 2016) or in how this data should be calculated and presented, especially in the case of electricity (Curran et al., 2005). As a result, there are methdological inconsistencies in CLCA studies due to the complexity of estimating marginal effects. Neihter in ALCA is the definition of electricity production mix straight forward. (Ekvall, 2009; Soimakallio et al., 2011). In *Papers II* and *III,* electricity and heat produced with biogas were credited as an average energy production mix and marginal data were not used for simplification. This is, however, in line with the ILCD handbook (2010) guidance, which states that average data can be used to model small changes and marginal data can be used to model changes that have a large-scale effect on system production capacity.

One of the main methodological challenges in LCA studies related to nutrient recycling is the inclusion of carbon content of organic matter, its impacts to the environment as well as its degradation. The climate impacts related to these are typically excluded from LCA studies due to incomplete methods, lack of research data and uncertainty in impact assessments (Brandão et al., 2011; Petersen et al., 2013; Arzoumanidis et al., 2014;

Soimakallio et al., 2015; Celestina et al., 2019; Lam et al., 2020). In this dissertation, these impacts were not included, although in *Paper I* the carbon storage formed by wastewater sludge and home compost was estimated very roughly. The benefits achieved were insignificant due to the use of peat in composting.

In general, nutrient recycling leads to improved environmental performance by decreasing the need for mineral fertilizers and by helping to restore the soil organic matter, which improves soil structure and micro-organism activity, and reduces nutrient leaching (Liang et al., 2017; Wiesmeier et al., 2019). In contrast, mineral fertilizers do not contain organic matter. However, the common practice in recent LCA studies is still to completely exclude the role of organic matter in recycled nutrients or nutrient rich biomasses (Havukainen et al., 2020; Kyttä et al., 2021).

Furthermore, any changes in wastewater nutrient recovery, sludge production and processing and use affects the region's carbon balance and should be considered. When sludge or other organic material is utilized in soil improvement or fertilization, the carbon in the organic matter starts to decompose. A varying proportion of carbon in sludge or digestate is in a more stable form and decompose more slowly than the carbon in a readily degradable form (Heikkinen et al., 2021). Processing of these recycled biomasses affects the properties and content of carbon. The slowly degradable carbon is important to maintain the soil carbon storage. The readily degradable carbon of the organic matter ending in the soil, i.e. the labile substance, is important for the soil biota, in the decomposition activity of which the nutrients bound to the organic matter are released back to the plants. Soil microbes have an important effect on carbon storage as they break down carbon into a more permanent form (Liang et al., 2017, Wiesmeier et al., 2019). However, the role and mechanisms of soil microbes in the carbon cycle are still poorly understood (Liang et al., 2017; Chenu et al., 2019).

Some simplified attempts to include the changes in slowly degradable carbon related to nutrient recycling and processing of biomass in LCA studies has been accomplished by introducing Yasso-modeling (Liski et al.) and/or assumptions on degradation measures (Heinonsalo et al., 2020; Paavola et al., 2020; Lehtoranta et al., 2020; Lehtoranta et al., 2021c). Moreover, the temporal occurrence of greenhouse gas emissions and sinks is also a key factor in assessing the climate change impacts of organic materials and their processing. Dynamic indicators can be used for evaluation, that take into account the temporal occurrence of greenhouse gas emissions and sinks. For example, the REFUGE3-model (Pingoud et al., 2012; Helin et al., 2016; Koponen and Soimakallio, 2015) can be applied to estimate the impact of temporal emissions as Lehtoranta et al. (2022) and Koponen and Soimakallio (2015) have done.

Even though the methods are incomplete and under development, some indicative information can be obtained. Therefore, in future studies, when assessing changes in sludge production, the impacts on soil and carbon storage should be modelled at the regional level and the effects of changes in wastewater management, sludge processing and spreading to soil should be assessed. In addition, the role of using the sludge in landscaping compared to agricultural use, should be examined.

4.3 SOURCE SEPARATION IN PROMOTING CIRCULAR ECONOMY

4.3.1 FEASIBILITY IN RURAL AND URBAN AREAS

The results of this study indicate that, source separation has environmental benefits, and it should be therefore introduced more widely. As the technology is already in use on ships and aircrafts, as well as at outdoor festivals, for example, some source separated wastewaters could be utilized more efficiently in the short term already. The technology is viable and sensible option, especially in isolated areas, such as the Finnish archipelago and ski resorts in Lapland, where water availability can also be a challenge or occupancy rates vary greatly throughout the year, causing difficulties in municipal WWTPs wastewater treatment resulting in risk of overflows. By source separating wastewater, the design challenges of sizing the wastewater treatment plant, for example, can be reduced.

The implementation of source separating systems should be considered, especially when renovating or improving current systems in both rural and urban context. Source separation systems are more easily applicable in rural areas, but, in cities and urban areas, the switch to alternative systems would need more infrastructural changes. In both cases, the challenges are mainly related to technical and logistical issues, like transportation and processing of the collected urine and blackwater.

In general, blackwater systems perform better than urine systems in both urban and rural context in terms of nutrient recovery and feasibility (Kjerstadius et al., 2015; Skambraks et al., 2017; Bisschops et al., 2019; Besson et al., 2021). Practical issues related to urine separation include, e.g. that urine forms spontaneously struvite, which might clog pipes and compromise collection and pre-treatment systems (Doyle and Parsons, 2002; Altinbas, 2009; von Bahr and Kärrman, 2019). In addition, blackwater separation offers the opportunity to utilize other local organic feedstocks in anaerobic digesters. For example, kitchen or/and garden waste can be collected and treated anaerobically together with blackwater to produce biogas for energy and digestate for fertilizer use, increasing the nutrient and energy recovery potential even further (Kjerstadius et al., 2015; Kjerstadius et al., 2017; Skambraks et al., 2017; Stowa, 2018; Gomez et al., 2020; Xu et al., 2021).

Urine separation can be feasible in rural areas, especially if treated on-site to reduce the volume for collection and transportation (e.g. Malila et al., 2019b; Turlan, 2019). In urban and peri-urban areas, the collection of urine would require a separate sewage system or on-site treatment to reduce the need for transportation. On the other hand, the relatively easy processing of urine into fertilizer products, such as struvite, favors a urine separation system (Ganrot et al., 2007). The advantage of both systems is that blackwater or urine is not mixed with other wastewaters that might contain, aside from pharmaceuticals and hormones, larger amounts of harmful substances and micropollutants.

Source separation could reduce the diffuse pollution load of on-site wastewater treatment systems in households' in rural areas. Furthermore, source separation would decrease the amount of septic tank sludge collected and treated in municipal WWTPs. This would facilitate the operation of small treatment plants that report difficulties in treating sludge collected separately from septic tanks under their environmental permits due to the increased load on their systems, leading to long-distance transport of sludge to larger units (Tarkka and Leppänen, 2019). In addition, on-site wastewater treatment systems do not always work ideally in cold winter conditions due to reduced biological activity (Luostarinen et al., 2007; Kauppinen et al., 2014; Kinnunen et al., 2021; Vidal et al., 2021). Source separation partially addresses this problem, as only grey waters are treated on-site.

According to the results of this study, especially in rural areas, an on-site source separation system could be a better option for eutrophication than conventional on-site soil systems, provided that the on-site systems function properly and are appropriately maintained. However, logistics is one of the key issues related to the implementation of source separation sanitation and nutrient recycling, especially in sparsely populated rural areas. To alleviate logistical challenges, nutrients in urine and blackwater could be processed into easily transportable fertilizer products either in households or centrally. Tidåker et al. (2007) suggested handling septic tank sludge from rural areas as fertilizer on farmlands as an alternative to its treatment in a municipal WWTP. Farmers had a generally positive attitude toward handling (collection, storage, and spreading) of sewage sludge, on the assumption that the economic conditions are favorable, and the food industry accepts the practice (Tidåker et al., 2004). However, organizing the entire system from collection to reuse has been found to be challenging (McConville et al., 2017.)

In urban areas, source separation systems might be an attractive option for urban renewal or new city districts, especially given the cost of renovating outdated sewer networks and WWTPs and the potentially emerging market for recycled nutrients. If source separating systems were implemented in cities and urban areas already connected to the WWTP, the incoming nutrient and BOD loads to the WWTP would decrease significantly. For example, if the urine of all Finns were collected separately, the nitrogen load in WWTPs would be reduced by a quarter and the phosphorus load would be halved (Säylä, 2015; Malila et al., 2019a). This would result in lower sizing requirements for WWTPs, which in turn would reduce energy and chemical consumption as well as sludge production. However, reduced nutrient flow to WWTPs may affect the activated sludge process, which requires an appropriate C:N:P ratio. However, WWTPs and their input flows are site- specific, and it is possible that a reduction in nitrogen inflow could cause operational problems. The impacts of source separation on the operation of WWTPs in general have not been studied in more detail in this study.

In the recent years, a lot has happened at European level, especially in the implementation of source separation in urban planning. In Sweden, the Netherlands and Germany, there are a several development projects implementing source separation in the urban environment, some of which are already in use and new large- scale pilot areas are being planned (Stowa, 2014; Skambraks et al., 2014; Skambraks et al., 2017; Lennartsson and Kvarnström, 2017; Lennartsson et al., 2019; Gomez et al., 2020). The implementation of source separation systems is, however, hindered by several problems, which are related to administrative issues, weak interactions between knowledge development and entrepreneurship, and responsibilities (Lennartsson and Kvarnström, 2017; Lennartsson et al., 2017; McConville et al., 2017; Lehtoranta et al., 2021). Also, decision making in sanitation planning has been reported to be complex, including trade-offs between sociopolitical, environmental, technical, and economic factors (Bao et al., 2012).

One of the main questions in source separating systems is related to their economic feasibility. In sparsely populated rural areas, the expansion of sewage networks is rarely an economically feasible option due to long distances, which supports the implementation of source separating systems. In urban areas, some studies report the total costs of source separating systems being higher than in the conventional system at current market prices (*Paper III*) while others show the opposite (Wood et al., 2015; Schoen et al., 2017) especially if compensations for energy and fertilizers produced are recognized (Xue et al., 2016). However, the market for separating systems is marginal compared to the mainstream and the market for recycled nutrients is still evolving, which is reflected in their overall price level. In the future, water and nutrient scarcity may trigger the need for alternative separating sanitation solutions, making them more common and likely lowering their prices. However, the life cycle cost of source separation should be studied and compared to the possible improvements and investments required in municipal WWTPs. This would clarify the context and scale in which source separation would be an economically and environmentally viable alternative (McConville et al., 2017).

4.3.2 TOWARDS CIRCULARITY

Although the benefits of source separation are undeniable in terms of nutrient recovery, the implementation of such systems would largely require a systemic change in the wastewater treatment sector towards a circular economy (Larsen et al., 2009). Among Finnish water experts, nutrient recovery has been acknowledged as a highly significant aspect of the circular economy, but the probability of realization is low (Laitinen et al., 2019). This may reflect the fact, that water services are rather conventional and rigid in terms of change (Heino, 2016). Moreover, while rational planning and conventional habits prevail in the education and management of water services (Innes and Booher, 2010; Kurki, 2016), management problems are increasingly interconnected with political and social domains (Teisman et al., 2013; Linton and Budds, 2014). Thus, in order to exploit the potential for source separation, technical and financial investments are not sufficient, but a wide systemic change towards a circular economy is needed in the water services sector.

Traditionally, source separation has been considered a viable option only in rural areas (Nelson and Murray, 2008), but the results of this study show their potential in the urban area as well. According to Larsen et al. (2009), it is often assumed, that the acceptance of a source separating system is objected because of the anticipated feeling of revulsion, but some studies indicate the opposite; most people actually like source separating systems (Lienert and Larsen, 2010). Moreover, the attitudes of farmers using human originated nutrients are generally positive, as long as they are safe and convenient to use (Lienert et al., 2003; Simha et al., 2017). In addition, Simha et al. (2021a) concludes, that the acceptance among food consumers is not, at least, the major social barrier to the utilization of human urine.

Currently, many factors, among the negative attitudes and suspicions related to the use of human urine and feces in crop cultivation, prevent nutrient recovery from wastewaters on a larger scale. These include undeveloped logistical and management chains, as well as legislative barriers (Magid et al., 2006; Lienert and Larsen, 2010). The key bottlenecks in the profitability of the nutrient cycle of human origin as well as other recycled nutrients, are the low prices of mineral fertilizers and the lack of knowledge on hazardous substances. However, in the past few years, the prizes of mineral fertilizers have begun to rise and just recently concerns regarding the security of nutrient supply in unstable conditions have increased, making nutrient recycling more attracting and topical. There are also a need for policy instruments and incentives to enable the use of human originated recycled fertilizers in Finland. This would require the setting of threshold values for hazardous substances to ensure the safe reuse of nutrients.

In practice, the implementation of source separation, especially on an urban scale, requires major structural changes in infrastructure and practices, and according to Swedish experience, the process of implementing research results in practice is not straight forward (Kvarnström et al., 2000; Lennartsson et al., 2019). Although technical solutions are available and ready to be implemented with relatively low risks (Kvanström et al., 2006; Skambraks et al., 2014; Gomez et al., 2020; Larsen et al., 2021a;b), the biggest challenge is to change current business and service models. However, successful projects, such as in Sweden, can reduce the technical, cost and regulatory uncertainties of future projects and thus support the implementation of future projects (Skambraks et al., 2017). Moreover, experiences with implementing smaller-scale source separating systems has already made it possible to scale up and it is already relevant to discuss the introduction of the concept instead of piloting (Bisschops et al., 2019).

5 CONCLUSIONS AND RECOMMENDATIONS

The results of this study indicate that source separation would result in higher nutrient recovery in urban, peri-urban and rural areas compared to conventional systems. Source separation systems in rural areas have a higher increase in nutrient potential per capita compared to urban and peri-urban systems. Especially in rural areas, source separation would decrease the risk of diffuse pollution and provide a viable solution to current practices.

In urban areas, source separation shows clear benefits in terms of climate change and eutrophication. Blackwater treatment also provides a local resource of energy. The risk of higher acidification impacts is present in source separating systems requiring special attention in the handling, storage, and application of nutrients. Processing nutrients into a more transportable form would ease their use over a wider area and improve the safety and acceptance of recovered nutrients but would increase energy use.

All in all, with source separation, climate and eutrophication impacts could be decreased, but drawbacks in acidification impacts may occur. However, the actual environmental benefits of improved nutrient recovery and recycling require the realization of avoidable emissions, which rely strongly on the decisions made in the design of the system and the societal policies. For example, the benefits of improved nutrient recovery with source separation will not be realized if it does not lead to replacement of inorganic, energy-intensively produced fertilizers and fossil fuels. The full realization of the benefits usually requires the introduction of new policies and policy instruments, as well as good planning and management.

The CLCA offers an excellect tool to support planning, decision- making, and policy related to nutrient recycling by showing the potential environmental consequences of decisions. Moreover, LCA can be used to identify additional information needs and knowledge gaps. Although some principles have been developed, the LCA methodology still needs further development and accepted rules are needed for nutrient recycling to include the carbon content of organic matter, its impacts to the environment as well as its degradation. In the future, as the aim of the policy is to increase the efficiency of nutrient recycling, instead of studying only small or local-scale impacts, research efforts should also be made to analyze the impacts of changes in the processing and utilization of sludge or other recycled nutrient-rich biomasses (such as manure) in regional level and include impacts on carbon fluxes and harmful substances.

The results of this study reveal the importance of careful planning and management of wastewater treatment systems from a life cycle perspective in pursuit of sustainability. Sustainable and efficient recycling of wastewater nutrients requires successful management and planning of the whole chain, from site planning, collection, transportation and storage to processing and end use, to achieve cost-effective, acceptable, and environmentally sustainable end products. In addition to the development of technologies, logistics and regulations, a change in attitudes is needed to make human-originate nutrients socially acceptable for use in agriculture. This socio-technical change is inevitable to meet the growing nutrient demand of food production and to increase self-sufficiency and promote security of supply. To achieve this, the safety of nutrients of human origin for the environment must be ensured.

In the society, the recent rise in prices and the change in the availability of nitrogen fertilizers are supporting efforts to recover and recycle nutrients from wastewater. With source separation sanitation, the greatest increase in nutrient recovery is reached especially for nitrogen. Source separation is technically simpler than adding new recovery technologies to municipal WWTPs. However, the utilization of source separated nutrients may require the introduction of new technologies to process the nutrients for transport and field application. In this context, an overall assessment of the environmental life cycle impacts and costs of nutrient recovery, either by source separation or in WWTPs are worth considering.

Source separation allows for more efficient recycling of nutrients and would support the self-sufficiency of fertilizers. Thus, to realize the potential, policy support for the agricultural use of wastewater-based nutrients is needed. Moreover, in order to achieve the goals of nutrient recycling and the circular economy in Finnish environmental policy, the use of wastewater-based nutrients should be supported by legislation. This requires that the nutrients contained in the effluent are recovered and processed into safe end products. Source separation of wastewaters itself would offer a great opportunity to recover nutrients in safer, and plant available form.

Tackling the inefficiencies of nutrient recovery and recycling promotes the change towards circular economy and carbon neutrality in wastewater management. Source separation of wastewaters offers one way to accomplish these. However, source separation is not a short-term solution to improve self-sufficiency of nutrients due to significant changes required in infrastructure and technologies. It requires systemic change to transform current sanitation systems to support nutrient reclaim, recovery and reuse at their full capacity. Moreover, the need to repair existing and outdated wastewater systems is obvious and thus, careful consideration is needed to evaluate which solutions are worth implementing in the future. This represents a great opportunity to replace conventional and ineffective treatment systems with source separation units in suitable areas.

6 REFERENCES

Almeida, M. C., Butler, D. & Friedler, E. 1999. At-source domestic wastewater quality. Urban Water 1(1): 49–55.

Al Seadi, T. 2013. Biogas digestate quality and utilization. Published in: Wellinger, A., Murphy, J. & Baxter, D. (edit.). The biogas handbook. Science production and applications. Woodhead Publishing Limited, Cambridge. S. p. 267-301.

Altinbas, M. 2009. Nitrogen recovery via struvite production. Cervantes, F. (edit.). Environmental technologies to treat nitrogen pollution: Principles and engineering. IWA Publishing, London. p. 239-268.

Antikainen, R. 2007. Substance flow analysis in Finland – four case studies on N and P flows. Academic dissertation. University of Helsinki. Monographs of the Boreal Environment Research 27/2007.

Arzoumanidis, I., Fullana-i -Palmer, P., Raggi, A. & Gazulla, C. 2014. Unresolved issues in the accounting of biogenic carbon exchanges in the wine sector. Journal of Cleaner Production 82: 16–22.

Ayres, R.U. & Ayres, L W. 2002. A handbook of industrial ecology. Edward Elgar, Cheltenham.

Balkema A.J., Preisig, H.A., Otterpohl, R. Lambert, A.J. & Weijers S.R. 2001. Developing a model based decision support tool for the identification of sustainable treatment options for domestic wastewater. Water Science and Technology 43 (7): 265-270.

Benetto E., Nguyen D., Lohmann T., Schmitt B. & Schosseler, P. 2009. Life cycle assessment of ecological sanitation system for small-scale wastewater treatment. Science of the Total Environment 407: 1506-1516.

Besson, M., Berger, S., Tiruta-barna, L., Paul, E. & Sperandio, M. 2021. Environmental assessment of urine, black and grey water separation for resource recovery in a new district compared to centralized wastewater resources recovery plant. Journal of Cleaner Production 301: 126868.

Bisschops, I., Kjerstadius, H., Meulman, B. & van Eekert, M. 2019. Integrated nutrient recovery from source-separated domestic wastewaters for application as fertilisers. Current Opinion in Environmental Sustainability 40: 7-13.

Brandão, M., Lavasseur, A., Kirschbaum, M., Weidema, B., Cowie, A., Jorgensen, A., Hauschild, M., Pennington, D. & Chomkhamsri, K. 2013. Key issues and options in accounting for carbon sequestration and temporary storage in life cycle assessment and carbon footprinting. International Journal of Life Cycle Assessment 18: 230–240.

Brandão, M., Milà i Canals, L. & Clift, R. 2011. Soil organic carbon changes in the cultivation of energy crops: implications for GHG balances and soil quality for use in LCA. Biomass Bioenergy 35: 2323–2336.

Brentrup, F. 2009. The impact of mineral fertilizers on the carbon footprint of crop production. Conference paper: International Plant Nutrition Colloquium XVI, at University of California, Davis/US.

Breen, P. 1990. A mass balance method for assessing the potential of artificial wetlands for wastewater treatment. Water Research 24(6): 689-697.

Brix, H. 1999. How green are aquaculture, constructed wetlands and conventional wastewater treatment systems? Water Science and Technology 32(3): 13-20.

Bulkeley, H., S. Marvin, Y. Palgan, K. Mccormick, M. Breitfuss-Loidl, L. Mai, T. Von Wirth, & N. Frantzeskaki. 2019. Urban living kaboratories: conducting the experimental city? European Urban and Regional Studies 26 (4): 317–335.

Cai, J., Liang, P., Qiu, Y., Cao, J. & Huang, X. 2020. A mass balance framework to evaluate the new pathways of biological nutrient removal in wastewater treatment. Journal of Water Process Engineering 37: 101523.

Celestina, C., Hunt, J.R., Sale, P.W.G. & Franks, A. E. 2019. Attribution of crop yield responses to application of organic amendments: A critical review. Soil and Tillage Research 186: 135–145.

Chenu, C., Angers, D.A., Barre, P., Derrien, D., Arrouays, D. & Balesdent, J. 2019. Increasing organics stocks in agricultural soils: Knowledge gaps and potential innovations. Soil and Tillage Research 188: 41-52.

Curran, M.A., Mann, M. & Norris, G. 2005. The international workshop on electricity data for life cycle inventories. Journal of Cleaner Production 13(8): 853–862.

Cordell, D., Drangert, J.O. & White, S. 2009. The story of phosphorus: global food security and food for thought. Global Environmental Change 19: 292-305.

Cordell, D. 2010. The story of phosphorus. Sustainability implications of global phosphorus scarcity for food security. Linköping University.

Corominas, L., Foley, J., Guest, J. S., Hospido, A., Larsen, H. F., Morera, S. & Shaw, A. 2013. Life cycle assessment applied to wastewater treatment: state of the art. Water Resources 47(15): 5480-92.

Díaz-Cruz, M.S., García-Galán, M.J., Guerra, P., Jelic, A., Postigo, C., Eljarrat, E., Farré, M., López de Alda, M.J., Petrovic, M. & Barceló, D. 2009. Analysis of selected emerging contaminants in sewage sludge. TrAC Trends in Analytical Chemistry 28(11): 1263-1275.

Dixon, A., Simon, M. & Burkitt, T. 2003. Assessing the environmental impact of two options for small-scale wastewater treatment: comparing a reed bed and an aerated biological filter using a life cycle approach. Ecology Engineering 20: 297-308.

Dolar, A., Selonen, S., van Gestel, C. A. M., Perc, V., Drobne, D. & Kokalj, A. J. 2021. Microplastics, chlorpyrifos and their mixtures modulate immune processes in the terrestrial crustacean Porcellio scaber. Science of the Total Environment 772: 144900.

Doyle, K. & Parsons, S. 2002. Struvite formation, control and recovery. Water Research 36: 3925-3940.

Dupas, R., Delmas, M. Dorioz, J-M., Garnier, J. Moatar, F. & Gascuel-Odoux, C. 2015. Assessing the impact of agricultural pressures on N and P loads and eutrophication risk. Ecological Indicators 48: 396-407.

Ekvall, T. & Weidema, B. P. 2004. System boundaries and input data in consequential life cycle inventory analysis. The International Journal of Life Cycle Assessment. 9: 161-171.

Ekvall, T., Azapahic, A., Finnveden, G., Rydberg, T., Weidema, B. P. & Zamagni, A., 2016. Attributional and consequential LCA in the ILCD handbook. International Journal of Life Cycle Assessment 21: 293-296.

Ekvall, T. 2019. Attributional and consequential life cycle assessment. Chapter in Sustainability Assessment at the 21st century, eds. Bastatante-Ceca, M. J.

EMEP/EEA 2016. EMEP/EEA air pollutant emission inventory Guidebook 2016. 3.B Manure management. European Environment Agency. https://www.eea.europa.eu/publications/emep-eea-guidebook-2016.

Emmerson R.H.C., Morse G.K., Lester, J.N. & Edge, D.R . 1995. Life cycle analysis of small scale sewage treatment processes. Water Environment Journal 9 (3): 317-325.

Etter, B., Udert, K.M. & Gounden, T. 2015. Valorisation of urine nutrients: Promoting sanitation and nutrient recovery through urine separation. VUNA Final Report. Eawag, Dübendorf, Switzerland.

European Commission 2017. Communication from the commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions on the 2017 List of Critical Raw Materials for the EU.

COM/2017/0490 final. Brussels, 13.9.2017. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52017DC0490

European Commission. 2020. Circular economy action plan. For a cleaner and more competitive Europe. Available at: https://ec.europa.eu/environment/pdf/circular-economy/new_circular_economy_action_plan.pdf

FAO (The Food and Agriculture Organization). 2006. Food security. Policy Brief. FAO's Agriculture and Development Economics Division. June 2006.

Fan., J-X., Urbain, V. & Manem, J. 1996. Nitrification and mass balance with a membrane bioreactor for municipal wastewater treatment. Water Science and Technology 34(1-2): 129-136.

Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D. & Suh, S. 2009. Recent developments in life cycle assessment. Journal of Environmental Management 91(1): 1-21.

Fidjeland, J., Svensson, S.E. & Vinnerås, B. 2015. Ammonia sanitization of blackwater for safe use as fertilizer. Water Science and Technology 71: 795–800.

Finnish Environment Institute. 2019. Yhdyskuntien jätevesien kuormitus vesiin. (Pollution of municipal wastewater into water) Available at: https://www.ymparisto.fi/fi-fi/kartat_ja_tilastot/vesihuoltoraportit/Yhdyskuntien_jatevesien_kuormitus_vesiin#:~:text=Yhdyskuntien%20j%C3%A4teveden%20puhdistamoilta%20johdettiin%20vesist%C3%B6%C3%B6n,typpe%C3%A4%20noin%2011%20090%20tonnia. [In Finnish]

Ganrot, Z., Dave, G., & Nilsson, E. 2007. Recovery of N and P from human urine by freezing, struvite precipitation and adsorption to zeolite and active carbon. Bioresource Technology 98(16): 3112-3121.

Geels, F.W. & Schot, J. 2010. The dynamics of transitions: A socio-technical perspective. In transitions to sustainable development: new directions in the study of long term transformative change; Grin, J., Rotmans, J., Schot, J., Eds.; Routledge: New York, NY, USA. p. 11–104

Geels, F.W. 2005. The dynamics of transitions in socio-technical systems: A multi-level analysis of the transition pathway from horse-drawn carriages to automobiles (1860–1930). Technology Analysis and Strategic Management. 17: 445–476.

Gomez, C., Medina, B., Sabucedo, J.M., Smith, D. & Obed, J. 2020. Recovery and utilization of nutrients 4 low impact fertilizer. Run4Life. European Union Horizon 2020. Available at: https://run4life-project.eu/wp-content/uploads/2020/06/Run4Life-Deliverable-6.1-Initial-Mapping-of-Stakeholder-and-Expectations.pdf

Government Decree on Treating Domestic Wastewater in Areas Outside Sewer Networks 157/2017 (209/2011). Ministry of the Environment (Finland).

Grönroos, J., Munther, J. & Luostarinen S. 2017. Calculation of atmospheric nitrogen and NMVOC emissions from Finnish agriculture. Description of the revised model. Reports of the Finnish Environment Institute, 37.

Grönroos, J., Rankinen, K., Cano-Bernal, J. E., Larvus L. & Alakukku, L. 2013a. Life cycle inventory & assessment report: source separation of dairy cattle manure, Finland. Knowledge report. Baltic Forum for Innovative Technologies for Sustainable Manure Management.

Grönroos, J., Rankinen, K., Cano-Bernal, J. E., Larvus L. & Alakukku, L. 2013b. Life cycle inventory & assessment report: cooling of manure, applied to fattening pig slurry, Finland. Knowledge report. Baltic Forum for Innovative Technologies for Sustainable Manure Management.

Harris, B., & Helgertz, J. 2019. Urban sanitation and the decline of mortality. The History of the Family 24 (2): 207-226.

Havukainen, J., Väisänen, S., Rantala, T., Saunila, M. & Ukko, J. 2020. Environmental impacts of manure management based on life cycle assessment approach. Journal of Cleaner Production 264: 121576.

Heijungs, R., Settanni, E. & Guinée, J. 2013. Toward a computational structure for life cycle sustainability analysis: unifying LCA and LCC. International Journal of Life Cycle Assessment 18 (9): 1722-1733.

Heikkinen, J., Ketoja, E., Seppänen, L., Luostarinen, S., Fritze, H., Pennanen, T., Peltoniemi, K., Velmala, S., Hanajik, P. & Regina, K. 2021. Chemical composition controls the decomposition of organic amendments and influences the microbial community structure in agricultural soils. Carbon Management 12 (4): 359-376.

Heimersson, S., Svanström, M. & Ekvall, T. 2019. Opportunities of consequential and attributional modelling in life cycle assessment of wastewater and sludge management. Journal of Cleaner Production 222: 242-251.

Heino, O. 2016. Paradigman jäljillä. Tutkimus vesihuollon ajattelumalleista. (Exploring the åaradigm. Research on thought patterns of water services). Doctoral dissertation, Tampere University of Technology. https://trepo.tuni.fi/handle/10024/114521 [In Finnish]

Heinonsalo, J. (edit.) 2020. Hiiliopas. Katsaus maaperän hiileen ja hiiliviljelyn perusteisiin. https://carbonaction.org/wp-content/uploads/2020/01/BSAG-hiiliopas-1.-painos-2020.pdf [In Finnish]

Helin, T., Salminen, H., Hynynen, J., Soimakallio, S., Huuskonen, S. & Pingoud, K. 2016. Global warming potentials of stemwood used for energy and materials in Southern Finland: differentiation of impacts based on type of harvest and product lifetime. GCB Bioenergy 8: 334-345.

HELCOM. 2013. Baltic Sea Action Plan, Nutrient Reduction Scheme. Baltic Marine Environment Protection Commission. Available at: https://helcom.fi/baltic-sea-action-plan/nutrient-reduction-scheme/targets/

Hjorth, M., Christensen, K.V., Christensen, M.L. & Sommer, S.G. 2010. Solid-liquid separation of animal slurry in theory and practice. A review. Agronomy for Sustainable Development 30: 153–180.

Hoffmann, S., Feldmann, U., Bach, P.M., Binz, C., Farrelly, M., Frantzeskaki, N., Hiessl, H., Inauen, J., Larsen, T.A., Lienert, J., Londong, J., Lüthi, C., Maurer, M., Mitchell, C., Morgenroth, E., Nelson, K.L., Scholten, L., Truffer, B. & Udert, K.M. 2020. A research agenda for the future of urban water management: Exploring the potential of nongrid, small-grid, and hybrid solutions. Environmental Science & Technology 54: 5312–5322.

Hong, Y., Lee, I., Lee, W. & Kim, H. 2019. Mass-balance-model-based evaluation of sewage treatment plant contribution to residual pharmaceuticals in environmental waters. Chemosphere 225: 378-387.

Horn, S., Seppänen, A-M., Winquist, E., Lehtoranta, S. & Luostarinen, S. 2020. Biokaasulaitoksen mädätysjäännöksen hyödyntämisvaihtoehdot – vaihtoehtojen ilmastovaikutukset ja taloudellisuus. Suomen ympäristökeskuksen raportteja 42/2020. [In Finnish]

Huegel, K. 2000. Ökobilanzen in der Siedlungswasserwirtschaft (Life Cycle Assessment in sanitary engineering). PhD thesis, Institut für Umweltingenieurwissenschaften, ETH Zurich.

Hukari, S., Herrman, L. & Nättorp, A. 2016. From wastewater to fertilizers – technical overview and critical review of European legislation governing phosphorus recycling. Science of the Total Environment 542: 1127-1135.

Hurley, R. R. & Nizzetto, L. 2018. Fate and occurrence of micro(nano)plastics in soils: Knowledge gaps and possible risks. Current Opinion in Environmental Science & Health 1: 6-11.

ILCD handbook (International Reference Life Cycle Data System). 2010. General guide for life cycle assessment – detailed guidance. JRC European Commission. Luxembourg.

Innes, J. & Booher, D. 2010. Planning with complexity. An introduction to collaborative rationality for public policy. New York, NY: Routledge.

IPCC 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia, L., Miwa, K., Ngara, T. & Tanabe K. (eds.). IGES, Japan.

IPCC 2014. Climate Change 2014. Mitigation of Climate Change. Intergovernmental panel on climate change. WGIII. Working group III contribution to the fifth assessment report of the intergovernmental panel on climate change. Cambridge University Press 2014.

ISO 14040. 2006. Environmental management - Life cycle assessment - Principles and framework.

Jasinski, S. M. 2011. Phosphate Rock, Mineral Commodity Summaries. U. S. Geological Survey.

Jurgilevich, A., Birge, T., Kentala-Lehtonen, J., Korhonen-Kurki, K., Pietikäinen, J., Saikku, L. & Schösler, H. 2016. Transition towards circular economy in the food system. Sustainability 8(1), 69.

Jönsson, H., Baky, A., Jeppsson, U., Hellström, D. & Kärrman, E. 2005. Composition of urine, faeces, greywater and biowaste. Urban Water - Chalmers University of Technology. Urban Water Report 6: 49.

Kaljunen, J. U., Al-Juboori, R. A., Mikola, A., Rihgetto, I. & Konola I. 2021. Newly developed membrane contactor-based N and P recovery process: Pilot-scale field experiments and cost analysis. Journal of Cleaner Production 281: 125288.

Kallio, J. 2020. Jätevesineuvonta haja-asutusalueilla 2011-2019 – Loppuraportti. (On-site wastewater treatment guidance 2011-2019 – final report). Finnish Environment Institute, Helsinki. Suomen ympäristökeskuksen raportteja 48/2020. [In Finnish]

Kallio, J. & Suikkanen, J. 2019. Muistio haja-asutusalueiden jätevedenkäsittelyn toimeenpanon alueellisesta tilanteesta 2019. (Memo of the regional situation in implementing on-site sanitation in 2019). Finnish Environment Institute 25.6.2019. [In Finnish]

Karak, T. & Bhattacharyya, P. 2011. Human urine as a source of alternative natural fertilizer in agriculture: A flight of fancy or an achievable reality. Resources, Conservation and Recycling 55(4): 400–408.

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

Kauppinen, A., Martikainen, K., Matikka, V., Veijalainen, A-M., Pitkänen, T., Heinonen-Tanski, H. & Miettinen, I. 2014. Sand filters for removal of microbes and nutrients from wastewater during a one-year pilot study in a cold temperate climate. Journal of Environmental Management 133: 206-213.

Khiewwijit, R., Temmink, H., Rijnaarts, H & Keesman, K. 2015. Energy and nutrient recovery for municipal wastewater treatment: How to design a feasible plant layout? Environmental Modelling & Software 68: 156-165.

Kinnunen J., Rossi P.M., Herrmann I., Ronkanen A-K. & Heiderscheidt E. 2021. Factors affecting effluent quality in on-site decentralized wastewater treatment systems in cold climate regions. Nordic Wastewater Conference, NORDIWA 2021 (ONLINE), 28/09-01/10/2021.

Kjerstadius, H., Haghighatafshar, S. & Davidsson, Å. 2015. Potential for nutrient recovery and biogas production from blackwater, food waste and greywater in urban source control systems. Environmental Technology 36(13): 1707-1720.

Kjerstadius, H., Saraiva, B. A., Spångberg, J. & Davidsson, Å. 2017. Carbon footprint of urban source separation for nutrient recovery. Journal of Environmental Management 197: 250-257.

Koponen, K. & Soimakallio, S. 2015. Foregone carbon sequestration due to land occupation—the case of agro-bioenergy in Finland. The International Journal for Life Cycle Assessment 20: 1544–1556.

Korhonen, J., Honkasalo, A. & Seppälä, J. 2017. Circular economy: The concept and its limitations. Ecological Economics 143: 37-46.

Kremen, C. 2005. Managing ecosystem services: what do we need to know about their ecology? Ecology letters 8: 468–479.

Kurki, V. 2016. Negotiating groundwater governance. Lessons from contentious aquifer recharge projects. Doctoral dissertation, Tampere University of Technology.

Kuster, M., López de Alda, M.J., D. & Barceló. D. 2005. The Handbook of Environmental Chemistry, Vol. 5, Part 0, Springer, Berlin, Germany, pp. 1-24

Kvarnström, E., Emilsson, K., Richert Stintzing, A., Johansson, M., Jönsson, H., af Petersens, E., Schönning, C., Christensen, J., Hellström, D., Qvarnström, L., Ridderstolpe, P. & Drangert, J.-O. 2006. Urine diversion: one step towards sustainable sanitation. Stockholm Environment Institute, Stockholm.

Kyttä, V., Helenius, J. & Tuomisto, H. J. 2021. Carbon footprint and energy use of recycled fertilizers in arable farming. Journal of Cleaner Production 287: 125063.

Laak, R. 1974. Relative pollution strengths of undiluted waste materials discharged in households and the dilution waters used for each. In: J.H.T. Winneberger, Manual of grey water treatment practice, 68-78. Michigan, USA: Ann Arbor.

Laamanen, M. 2016. Programme of measures for the development and implementation of the marine strategy in Finland 2016–2021. Reports of the Ministry of the Environment 5/2016. [In Finnish]

Laitinen, J., Antikainen, R., Hukka, J. & Katko, S. 2019. Water supply and sanitation in a green economy society: the case of Finland. Public Works Management & Policy 25(1): 33-50.

Lam, L., Kurisu, K. & Hanaki, K. 2015. Comparative environmental impacts of source separation systems for domestic wastewater management in rural China. Journal of Cleaner Production 104: 185-198.

Lam, K. L., Zlatanovic, L. & van der Hoek, J. P. 2020. Life cycle assessment of nutrient recycling from wastewater: A critical review. Water Research 173: 115519.

Lapinlampi, T. 2021. Vesihuoltolaitokset 2014. Viemärilaitokset. (Water utilities 2014. Wastewater). Suomen ympäristökeskuksen raportteja 23/2021. Finnish Environment Institute, Helsinki. [In Finnish]

Larsen, T. A., Alder, A. C., Eggen, R.I.L., Maurer, M. & Lienert, J. 2009. Source separation: Will we see a paradigm shift in wastewater handling? Environmental Science & Technology 43: 6121-6125.

Larsen, T.A., Gruendl, H. & Binz, C. 2021a. The potential contribution of urine source separation to the SDG agenda-a review of the progress so far and future development options. Environmental Science and Water Research Technology.

Larsen, T. A., Riechmann, M. E. & Udert, K. M. 2021b. State of the art of urine treatment technologies: A critical review. Water Research 13: 100114.

Lehtoranta, S., Malila, R., Fjäder, P., Laukka, V., Mustajoki, J. & Äystö, L. 2021a. Jätevesien ravinteet kiertoon turvallisesti ja tehokkaasti. Suomen ympäristökeskuksen raportteja 18/2021. [In Finnish]

Lehtoranta, S., Malila, R., Kämäri, T., Johansson, A., Särkilahti, M. & Viskari, E-L. 2021b. Vesihuollon kiertotalouden edistäminen kaupunkiympäristössä. Suomen ympäristökeskuksen raportteja 6/2021. [In Finnish]

Lehtoranta, S., Johansson, A., Myllyviita, T., Grönroos, J. & Manni, K. 2021c. Turvetta korvaavien kuivikemateriaalien ilmastovaikutukset. Suomen ympäristökeskuksen raportteja 51/2021. [In Finnish]

Lehtoranta, S., Johansson, A., Malila, R., Rankinen, K., Grönroos, J., Luostarinen, S. & Kaistila, K. 2020. Vaihtoehtoja kestävämpään turkiseläinten lannan hyödyntämiseen. Suomen ympäristökeskuksen raportteja 35/2020. [In Finnish]

Lennartsson M., McConville J., Kvarnström E., Hagman M. & Kjerstadius, H. 2019. Investment in innovative urban sanitation – decision-making processes in Sweden. Water Alternatives 12(2): 588-608.

Lennartsson, M. & Kvarnström, E. 2017. Food and energy in a circular economy, 2017. Final Report. Stockholm June 20, 2017. https://carbonneutralcities.org/wp-content/uploads/2018/05/Food-and-Energy-in-the-Circular-Economy.pdf

Lennartsson, M., McConville, J., Kvarnström, E., Hagman, M. & Kjerstadius, H. 2019. Investments in innovative, urban sanitation- Decision-making processes in Sweden. Water Alternatives 12(2): 588-608.

Lennartsson, M., Kvarnström, E., Lundbreg, T., Buenfil, J. & Sawyer, R. 2009. Comparing sanitation systems using sustainability criteria. Stockholm Environment Institute, Stockholm. Ecological Sanitation Research Series, 2009-1.

Liang, C., Schimel, J.P. & Jastrow, J.D. 2017. The importance of anabolism in microbial control over soil carbon storage. Nature Microbiology 2, 17105.

Lienert, J., Haller, M., Berner, A., Stauffacher, M., & Larsen, T. A. 2003. How farmers in Switzerland perceive fertilizers from recycled anthropogenic nutrients (urine). Water Science and Technology 48 (1), 47–56.

Lienert, J. & Larsen, T. A. 2010. High acceptance of urine source separation in seven European Countries: A Review. Environmental Science & Technology 44(2): 556-566.

Linton, J. & Budds, J. 2014. The hydrosocial cycle: defining and mobilizing a rational-dialectical approach to water. Geoforum 57: 170-180.

Liski, J., Viskari, T., Järvenpää, M., Repo, A., Kulmala, L., Akujärvi, A. & Kaasalainen, M. Soil Carbon model Yasso15. Unpublished manuscript.

Luostarinen, S., Grönroos, J., Hellstedt, M., Nousiainen, J. & Munther J. 2017. Suomen normilanta – laskentajärjestelmän kuvaus ja ensimmäiset tulokset. Luonnonvara- ja biotalouden tutkimus 47/2017. [In Finnish]

Luostarinen, S., Grönroos, J., Lemola., R., Lehtonen, E., Lehtoranta, S., Tampio, E., Turtola, E. & Uusitalo, R. 2019. Ravinnelaskuri – Alueellisen ravinnekierron suunnittelun työkalu. Luonnonvara- ja biotalouden tutkimus. https://www.luke.fi/projektit/ravinnelaskuri/ [In Finnish]

Luostarinen, S., Sanders, W., Kujawa-Roeleveld, K. & Zeeman, G. 2007. Effect of temperature on anaerobic treatment of black water in UASB-septic tank systems. Bioresource Technology 98(5): 980-986.

Magid, J., Eilersen, A. M., Wrisberg, S. & Henze, M. 2006. Possibilities and barriers for recirculation of nutrients and organic matter from urban to rural areas: A technical theoretical framework applied to the medium-sized town Hillerod, Denmark. Ecological Engineering 28(1): 44-54.

Magdoff, F. R. & Chromec, F. W. 1977. Nitrogen mineralization from sewage sludge. Journal of Environmental Science and Health. Part A: Environmental Science and Engineering. 12: 191-201.

Magnusson, K. & Norén, F. 2014. Screening of microplastic particles in and down-stream a wastewater treatment plant. IVL Swedish Environmental Research Institute, Number C 55 August 2014 Report.

Malila, R., Lehtoranta, S. & Viskari, E.-L. 2019a. The role of source separation in nutrient recovery e Comparison of alternative wastewater treatment systems. Journal of Cleaner Production 219: 350-358.

Malila, R., Viskari, E-L. & Kallio, J. 2019b. Virtsan ravinteet kiertoon – MORTTI-hankkeen loppuraportti. Suomen ympäristökeskuksen raportteja 29/2019. [In Finnish]

Marttinen, S., Venelampi, O., Iho, A., Koikkalainen, K., Lehtonen, E., Luostarinen, S., Rasa, K., Sarvi, M., Tampio, E., Turtola, E., Ylivainio, K., Grönroos, J., Kauppila, J., Koskiaho, J., Valve, H., Laine-Ylijoki, J., Lantto, R., Oasmaa, A. & Castell-Rüdenhausen, M. 2017. Kohti ravinteiden kierrätyksen läpimurtoa. Nykytila ja suositukset ohjauskeinojen kehittämiseksi Suomessa. Luonnonvarakeskus, Helsinki. Luonnonvara- ja biotalouden tutkimus 45/2017. [In Finnish]

Maktabifard, M., Awaitey, A., Merta, E., Haimi, H., Zaborowska, E., Mikola, A. & Makinia, J. 2022. Comprehensive evaluation of the carbon footpront components of wastewater treatment plants located in the Baltic Sea region. Science of the Total Environment 806(1): 150436.

Maurer, M., Pronk, W. & Larsen, T.A. 2006. Treatment processes for source-separated urine. Water Research 40(17): 3151-3166.

McConville, J.R., Kvarnström, E., Jönsson, H., Kärrman, E. & Johansson, M. 2017. Source separation: challenges & opportunities for transition in the swedish wastewater sector. Resources, Conservation and Recycling 120: 144-156.

Ministry of the Agriculture and Forestry (Finland). 2020. Law drafting. Available at: https://mmm.fi/en/project?tunnus=MMM072:00/2020 [In Finnish]

Ministry of the Environment, Finland. 2019. Ravinteiden kierrätyksen toimenpideohjelma 2019-2030. (Operational program for nutrient recycling 2019-2030). Available at: Ravinteiden-kierratyksen-toimenpideohjelma-2019-2030-allekirjoitettu-D7F9043A_0090_4785_B029_9C119B566BDD-146284.pdf (ym.fi) [In Finnish]

Moerland, M.J., Borneman, A., Chatzopoulos, P., Fraile, A.G., van Eekert, M.H.A., Zeeman, G. & Buisman, C.J.N. 2020. Increased (Antibiotic-resistant) pathogen indicator organism removal during (hyper)thermophilic anaerobic digestion of concentrated black water for safe nutrient recovery. Sustainability 12: 1–13.

Møller H.B., Sommer S.G. & Ahring B.K. 2002. Separation efficiency and particle size composition in relation to manure type and storage conditions. Bioresource Technology 85: 189–196.

Morelli, B., Hawkins, T. R., Niblick, B., Henderson, A. D., Golden, H.E., Compton, J. E., Cooter, E. J. & Bare, J. C. 2018. Critical review of eutrophication models for life cycle assessment. Environmental Science & Technology 52(17): 9562-9578.

Motiva. 2020. Vedenkulutus (Water consumption). Available at: https://www.motiva.fi/koti_ja_asuminen/hyva_arki_kotona/vedenkulutus [In Finnish].

MSFD. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy. Marine Strategy Framework Directive. Official Journal of the European Union.

Myers, S. & Nigh, V. 2021. Too Many to Count: Factors Driving Fertilizer Prices Higher and Higher. Market Intel. The American Farm Bureau Federation. December 13, 2021. Cited 13.1.2022. Available at: https://www.fb.org/market-intel/too-many-to-count-factors-driving-fertilizer-prices-higher-and-higher

Mönkäre, T., Kinnunen, V., Tampio, E., Ervasti, S., Lehtonen, E., Kettunen, R., Rasi, S. & Rintala, J. 2016. Ravinnevisio - Selvitys Pirkanmaan puhdistamolietteiden ja biojätteiden ravinteista ja niiden potentiaalisesta käytöstä. Pirkanmaan elinkeino-, liikenne- ja ympäristökeskus, Helsinki. Raportteja 74/2016. [In Finnish]

Nacry, P., Bouguyon, E., & Gojon, A. 2013. Nitrogen acquisition by roots: physiological and developmental mechanisms ensuring plant adaptation to a fluctuating resource. Plant Soil 370: 1–29.

Natural Resources Institute Finland. 2022a. Sales of fertilizers to farms. Statics database.

Natural Resources Institute Finland. 2022b. Number of Livestock. Statics database.

Nelson, K. L. & Murray, A. 2008. Sanitation for unserved populations: technologies, implementation challenges and opportunities. Annual Review of Environment and Resources 33: 119– 15.

Nordin, A., Ottoson, J.R. & Vinnerås, B. 2009. Sanitation of faeces from source-separating dry toilets using urea. Journal of Applied Microbiology 107: 1579–1587.

Olshammar, M., Rosenquist, M. Ek, L., Ejhed, H., Sidvall, A & Svanström, S. 2015. På uppdrag av Havs-och vattenmyndigheten Uppdatering av kunskapsläget och statistik för små avloppsanläggningar (The state of knowledge and statistics for small sewage plants. Swedish Agency for Marine and Water Management). Report 166, Norrköping. [In Swedish]

Paavola, T., Lehtoranta, S., Luostarinen, S., Akujärvi, A. & Grönroos, J. 2019. Agrisymbiooseilla kohti kestävämpää sianlihantuotantoa. Suomen ympäristökeskuksen raportteja 50/2019. [In Finnish]

Petersen, M. B., Knudsen, M. T. & Hermansen, J. E. 2013. An approach to include soil carbon changes in life cycle assessment. Journal of Cleaner Production 52: 217–224.

Pingoud, K., Ekholm, T. & Savolainen, I. 2012. Global warming potential factors and warming payback time as climate indicators of forest biomass use. Mitigation and Adaptation Strategies for Global Change 17:369–386.

Pivato, A., Vanin, S., Raga, R., Lavagnolo, M. C., Barausse, A., Rieple, A., Laurent, A. & Cossu, R. 2016. Use of digestate from a decentralized on-farm biogas plant as fertilizer in soils: An ecotoxicological study for future indicators in risk and life cycle assessment. Waste Management 49: 378-389.

Puig., S., van Loosdrecht, M., Colprom, J. & Meijer, S. 2008. Data evaluation of full-scale wastewater treatment plants by mass balance. Water research 42(18): 4645-4655.

Remy, S. & Jekel, M. 2012. Energy analysis of conventional and source-separation systems for urban wastewater management using life cycle assessment. Water Science and Technology 65(1):22-29.

Renou, S., Thomas, J.S., Aoustin, E. & Pons, M.N. 2008. Influence of impact assessment methods in wastewater treatment LCA. Journal of Cleaner Production 16:1098-1105.

Riva, C. Orzi, V., Carozzi, M., Acutis, M., Boccasile G., Lonati S., Tambone F., D'Imporzano G. & Adani, F. 2014. Short-term experiments in using digestate products as substitutes for mineral (N) fertilizer: Agronomic performance, odours, and ammonia emission impacts. Science of The Total Environment 547: 206-214.

ROTI. 2021. Rakennetun omaisuuden tila 2021. (Value of the built environment in 2021). Available at https://www.ril.fi/media/2021/vaikuttaminen/roti2021_low.pdf [In Finnish]

Rodriguez, D., Serrano, H., Delgado, A., Nolasco, D. & Saltiel, G. 2021. From waste to resource. Shifting paradigms for smarter wastewater interventions in Latin America and the Caribbean. International Bank for Reconstruction and Development. The World Bank.

Rogers, H. R. 1996. Sources, behavior and fate of organic contaminants during sewage treatment and in sewage sludges. Science of the Total Environment 185(1–3): 3-26.

Rossi, L., Reuna, S., Fred, T. & Heinonen, M. 2019. RAVITA Technology – new innovation for combined phosphorus and nitrogen recovery. Water Science & Technology 78(12): 2511-2517.

Ruuhela, S. 2017. Puhdistamolietteen käsittelyn hankinnan laatukriteerien kehittäminen – RANKU-hanke. Varsinais-Suomen elinkeino-, liikenne- ja ympäristökeskus, Turku. Raportteja 18/2017. [In Finnish]

Ruuhela, S. 2016. Puhdistamolietteen käsittelyn hankinnan laatukriteerien kehittäminen. Tampere University of Technology. [In Finnish]

Räike, A., Taskinen, A. & Knuuttila, S. 2020. Nutrient export from Finnish rivers into the Baltic Sea has not decreased despite water protection measures. Ambio 48:460-474.

Salgot, M. & Folch, M. 2018. Wastewater treatment and water reuse. Current Opinion in Environmental Science & Health 2:64-74.

Saliu, T.D. & Oladoja, N. A. 2021. Nutrient recovery from wastewater and reuse in agriculture: a review. Environmental Chemistry Letters 19: 2299-2316.

Schell, T., Hurley, R., Buenaventura, N, T., Mauri, P. V., Nizzetto, L., Rico, A. & Vighi, M. 2022. Fate of microplastics in agricultural soils amended with sewage sludge: Is surface water runoff a relevant environmental pathway? Environmental Pollution 293: 118520.

Schaubroeck, T., Schaubroeck, S., Heijungs, R., Zamagni, A., Brandão, M. & Benetto, E. 2021. Attributional & consequential life cycle assessment: definitions, conceptual characteristics and modelling restrictions. Sustainability 13(13): 7386.

Schoen, M.E., Xue, X., Wood, A., Hawkins, T.R., Garland, J. & Ashbolt, N.J. 2017. Cost, energy, global warming, eutrophication and local human health impacts of community water and sanitation service options. Water Research 109: 186-195.

Schönning, C. 2006. Urine diversion: Hygienic risks and microbial guidelines for reuse. Report background paper for guidelines for the safe use of wastewater, excreta and greywater of WHO. WHO, Solna.

Seleiman, M.F., Santanen, A. & Makela, P.S.A. 2020. Recycling sludge on cropland as fertilizer - Advantages and risks. Resources, Conservation and Recycling 155: 104647.

Selonen, S., Dolar, A., Jemec Kokalj, A., Skalar, T., Parramon Dolcet, L., Hurley, R. & van Gestel, C.A.M. 2020. Exploring the impacts of plastics in soil – The effects of polyester textile fibers on soil invertebrates. Science of the Total Environment 700:134451.

Seppälä, J., Knuuttila, S. & Silvo, K. 2004. Eutrophication of freshwater ecosystems: a new method for calculating the potential contributions of nitrogen and phosphorus. International Journal of Life Cycle Assessment 9 (2): 90-100.

Shove, E. 2014. Putting practice into policy: Reconfiguring questions of consumption and climate change. Contemporary Social Science 9: 415–429.

Simha, P. 2021. Alkaline Urine Dehydration. How to dry source separated human urine and recover nutrients. Doctoral thesis No. 2021:28. Faculty of Natural Resources and Agricultural Sciences. Swedish University of Agricultural Sciences, Uppsala.

Simha, P., Barton, M.A, Perez-Mercado, L.F., McConville, J.R. ... & Vinnerås, B. (2021a). Willingness among food consumers to recycle human urine as crop fertiliser: Evidence from a multinational survey. Science of the Total Environment 765: 144438.

Simha, P., Friedrich, C., Randall, D.G., Vinnerås, B., 2021b. Alkaline dehydration of human urine collected in source-separated sanitation systems using magnesium oxide. Frontiers in Environmental Science 8: 286.

Simha, P., Lalander, C., Vinnerås, B. & Ganesapillai, M. (2017). Farmer attitudes and perceptions to the re–use of fertiliser products from resource–oriented sanitation systems–The case of Vellore, South India. Science of the Total Environment 581: 885-896.

Skambraks, A.-K., Kjerstadius, H., Meier, M., Davidsson, Å., Wuttke, M. & Giese, T. 2017. Source separation sewage systems as a trend in urban wastewater management: Drivers for the implementation of pilot areas in Northern Europe. Sustainable Cities and Society 28: 287–296.

Skambraks, A.-K., Augustin, K., Meinzinger, F. & Hartmann, M. 2014. Hamburg's lead on water and energy: Implementing resource-oriented sanitation using the Hamburg water cycle. Water 21: 15-18.

Socolow, R., Andrews, C., Berkhout, F., Thomas, V. (Eds.) 1994. Industrial ecology and global change. Cambridge University Press, UK.

Soimakallio, S., Cowie, A., Brandao, M., Finnveden, G., Ekvall, T., Erlandsson, M. Koponen, K. & Karlsson, P.-E. 2015. Attributional life cycle assessment: is a land-use baseline necessary? The International Journal of Life Cycle Assessment 20: 1364-1375.

Soimakallio, S., Kiviluoma, J. & Saikku, L. 2011. The compexity and challenges of determining GHG (greenhouse gas) emissions from grid electricity consumption and conservation in LCA (life cycle assessment) – A methodological review. Energy 36(12): 6705-6713.

Solomon, S., Qin, D., Manning, M., Marquis, M., Averyt, K., Tignor, M.M.B., Miller, H. & Chen Z. (Eds.). 2007. Climate Change 2007: the physical science basis: contribution of working group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, U.K.

Spångberg, J., Tidåker, P. & Jönsson, H., 2014. Environmental impact of recycling nutrients in human excreta to agriculture compared with enhanced wastewater treatment. Science of the Total Environment 493: 209–219.

Statistics Finland. 2021. Greenhouse gas emissions in Finland 1990 to 2019. National Inventory Report Under UNFCCC and the Kyoto Protocol. Submission to the European Union 15 March 2021.

Statistics Finland. 2019. Key figures on population by region, 1990-2020. Available at: https://pxnet2.stat.fi/PXWeb/pxweb/en/StatFin/StatFin__vrm__vaerak/statfin_vaerak_pxt_11ra.px/

Statistics Sweden. 2021. Population by age and sex, 2020.

Stowa. 2014. Evaluatie Nieuwe Sanitatie Noorderhoek Sneek. Rapport. Available at: https://www.stowa.nl/sites/default/files/assets/PUBLICATIES/Publicaties%202014/STOWA%202014-38.pdf

Stowa. 2018. Evaluatie nieuwe sanitatie Noorderhoek/Watershoon 2. Rapport 63. Available at: https://www.stowa.nl/sites/default/files/assets/PUBLICATIES/Publicaties%202018/STOWA%202018-63%20NS%20Noorderhoek.pdf

Suh, S. & Huppes, G. 2005. Methods for life cycle inventory of a product. Journal of Cleaner Production 13 (7): 687-697.

Sutton, P. M., Rittmann, B. E, Schraa, O. J., Banaszak, J. E. & Togna, A. P. 2011. Wastewater as a resource: a unique approach to achieving energy sustainability. Water Science Technology 63(9): 2004-2009.

Swiss Centre for Life Cycle Inventories. 2007. Ecoinvent Database, Version 2.01.

Säylä, L. 2015. Urban wastewater treatment 2013. Reports of the Finnish Environment Institute 34/2015.

Tattari, S., Puustinen, M., Koskiaho, J., Röman, E. & Riihimäki, J. 2015. Vesistöjen ravinnekuormituksen lähteet ja vähentämismahdollisuudet. Suomen ympäristökeskuksen raportteja 35. [In Finnish]

Talvitie, J., Mikola, A., Setälä, O., Heinonen, M. & Koistinen, A. 2017. How well is microlitter purified from wastewater? – A detailed study on the stepwise removal of microlitter in a tertiary level wastewater treatment plant. Water Research 109:164-172.

Tarkka, P. & Leppänen, E. 2019. Hajajätevesilietteet hyötykäyttöön Itä-Lapissa. Hankesuunnitelma. Accessed 13 January 2022. Available at: https://www.savukoski.fi/wp-content/uploads/2020/02/hankesuunnitelma-02042019.pdf [In Finnish]

Teglia, C., Tremier, A. & Martel J.-L. 2010. Characterization of solid digestates: Part 1, Review of existing indicators to assess solid digestates agricultural use. Waste and Biomass Valorization 2 (1):43-58.

Teisman, G., van Buuren, A., Edelenbos, J. & Warner, J. 2013. Water governance: Facing the limits of managerialism, water-centricity, and technocratic problem-solving. International Journal of Water Governance 1(01-02): 1-11.

Tervahauta, T., Hoang, T., Hernandez, L, Zeeman G. & Buisman, C. 2013. Prospects of source-separation-based aanitation concepts: a model-based study. Water 5(3): 1006-1035.

Thibodeau, C., Monette, F., Bulle, C. & Glaus, M. 2014. Comparison of black water source-separation and conventional sanitation systems using life cycle assessment. Journal of Cleaner Production 67:45-57.

Tigini, V., Franchino, M., Bona, F. & Varese, G. C. 2016. Is digestate safe? A study on its ecotoxicity and environmental risk on a pig manure. Science of the Total Environment 551–552:127-132.

Tidåker, P., Sjöberg, C. & Jönsson, H. 2007. Local recycling of plant nutrients from small-scale wastewater systems to farmland — a Swedish scenario study. Resources, Conservation and Recycling 49: 388-405.

Tidåker, P., Kärrman, E., Baky, A. & Jönsson, H. 2006b. Wastewater management integrated with farming – an environmental systems analysis of a Swedish country town. Resources, Conversation and Recycling 47:295-315.

Tidåker, P., Sjöberg, C. & Jönsson, H. 2006a. Local recycling of plant nutrients from small-scale wastewater systems to farmland - A Swedish scenario study. Resources, Conservation and Recycling 49: 388-405.

Tidåker, P., Sjöberg, C & Jönsson, H. 2004. The use of sewage fertilizer products on arable land – requirements from the farmers perspective. Sustainable Sanitation Alliance.

Tillman, A.-M., Svinby, M. & Lundström, H. 1998. Life cycle assessment of municipal wastewater systems. International Journal of Life Cycle Assessment 3(3):145-157.

Tseng, M. L., Chiu, A. S. F., Chien, C. F. & Tan, R. R. 2019. Pathways and barriers to circularity in food systems. Resources, Conservation and Recycling 143: 236-237.

Turlan, L. 2019. Nutrient recovery from urine to agriculture in Northern Sweden. Luleå University of Technology, Sweden. Master's thesis.

Udert, K. M., Larsen, T.A., Gujer, W. 2006. Fate of major compounds in source-separated urine. Water Science & Technology 54(11-12): 413-420.

United Nations. 2021. The 2030 Agenda and the Sustainable Development Goals.

Valtanen M., Sillanpää N. & Setälä H. 2015. Key factors affecting urban runoff pollution under cold climatic conditions. Journal of Hydrology 529: 1578-1589.

Van der Hoek, J. P., de Fooji, H. & Struker, A. 2016. Wastewater as a resource: Strategies to recover resources from Amsterdam's wastewater. Resources, Conservation and Recycling 113: 53-64.

Van der Hoek, W., Hassan, M., Ensink, J., Feenstra, S., Raschid-Sally, L., Munir, S., Aslam, R., Ali, N., Hussain, R. & Matsuno, Y. 2002. Urban Wastewater: A valuable resource for agriculture. A case study from Haroonabad, Pakistan. Research report 63. International Wastewater Management Institute. Sri Lanka.

Van der Voet, E. 2002. Substance flow analysis methodology. Handbook of Industrial Ecology: 91–101.

Venkatesean, A., Hamdan, A-H., Chavez, V., Brown, J. & Halden, R. 2016. Mass balance model for sustainable phosphorus recovery in a US wastewater treatment plant. Journal of Environmental Quality 45(1): 84-89.

Vidal, B., Hedström, A., Barraud, S., Kärrman, E. & Herrman, I. 2019. Assessing the sustainability of on-site sanitation systems using multi-criteria analysis. Environmental Science: Water Research & Technology 9: 1599-1615.

Vidal, B., Kinnunen, J., Hedström, A., Heiderscheidt, E., Rossi, P. M. & Herrmann, I. 2021. Treatment efficiency of small-scale package plants in northern Sweden and Finland. Nordic Wastewater Conference, NORDIWA 2021 (ONLINE), 28/09-01/10/2021.

Vieno, N. 2014. Haitalliset aineet jätevedenpuhdistamoilla –hankkeen loppuraportti. (Harmful substances at wastewater treatment plant – final report). Finnish Water Utilities Association, Helsinki. Vesilaitosyhdistyksen monistesarja nro 34. [In Finnish]

Vieno N., Sarvi M., Salo T., Rämö S., Ylivoinio K., Pitkänen T., & Kusnetsov J. 2018. Puhdistamolietteiden sisältämien haitta-aineiden aiheuttamat riskit lannoitekäytössä. Luonnonvara- ja biotalouden tutkimus 58/2018. [In Finnish]

Vienonen, S. 2007. Haja-asutuksen vedenhankinnan ja jätevedenkäsittelyn tilanne vuonna 2007. Finnish Environment Institute, Helsinki. Suomen ympäristökeskuksen raportteja 24/2007. [In Finnish]

Vilpanen, M. & Seppälä, P. 2021. Yhdyskuntalietteen käsittelyn ja hyödyntämisen nykytilakatsaus vuosilta 2019-2020. Vesilaitosyhdistyksen monistesarja nro 71. Helsinki 2021. [In Finnish]

Viskari, E.-L., Grobler, G., Karimäki, K. Gorbatova, A., Vilpas, R. & Lehtoranta, S. 2018. Nitrogen recovery with source separation of human urine – preliminary results of its fertilizer potential and use in agriculture. Frontiers in Sustainable Food Systems 2(32): 1-14.

Viskari, E-L., Lehtoranta, S. & Malila, R. 2021. Urine: The potential, value chain and its sustainable management. Sanitation Value Chain 5(2): 27-43.

Viskari, E-L., Vilpas, R., Lehtoranta, S., Pakula, S. & Tuukkanen, K. 2017. BIOUREA. Erilliskerätyn virtsan lannoitepotentiaali, kokeelliset tutkimukset ja elinkaaritarkastelu. Loppuraportti. (BIOUREA. Fertilizer potential of source separated urine, experimental research and life cycle assessment). [In Finnish]

von Bahr, B. & Kärrman, E. 2019. Tekniska processer för fosforåtervinning ur avloppsslam. RISE Rapport 2019: 59. [In Swedish]

VTT (VTT Technical Research Centre of Finland). 2017. LIPASTO Emission database. Updated 7.7.2017. http://lipasto.vtt.fi/yksikkopaastot/index.htm [In Finnish]

Warman, P.R. & Termeer, W.C. 2005. Evaluation of sewage sludge, septic waste and sludge compost applications to corn and forage: Yields and N, P and K content of crops and soils. Bioresources Technology 96: 955–961.

Webb, J., Menzi, H., Pain, B. F., Misselbrook, T. H., Dämmgen, U., Hendriks, H. & Döhler, H. 2005. Managing ammonia emissions from livestock production in Europe. Environmental Pollution 135(3): 399-406.

Weckman, A., 2005. Nutrients from toilets to fields. Guide of the Finnish Garden Association. [In Finnish]

Weidema, B. P., Pizzol, M., Schmidt, J. & Thoma, G. 2018. Attributional or consequential Life Cycle Assessment: A matter of social responsibility. Journal of Cleaner Production 174: 305-314.

Weidema, B. 2003. Market information in life cycle assessment. Environmental Project No. 863. Copenhagen: Danish Environmental Protection Agency.

Wielemaker, R. C., Weijma, J., & Zeeman, G. 2018. Harvest to harvest: Recovering nutrients with new sanitation systems for reuse in urban agriculture. Resources, Conservation and Recycling 128: 426-437.

Wiesmeier, M., Urbanski, L., Hobley, E., Lang, B., von Lützow, M., Marin-Spiotta, E., van Wesemael, B., Rabot, E., Ließ, M., Garcia-Franco, N., Wollschläger, U., Vogel, H.-J. & Kögel-Knabner, I. 2019. Soil organic carbon storage as a key function of soils- A review of drivers and indicators at various scales. Geoderma 333: 149–162.

WHO & UNICEF. 2021. Progress on household drinking water, sanitation and hygiene. 2000-2020. Five years into the SDGs. WHO/UNICEF joint monitoring programme for water supply, sanitation and hygiene.

WWAP (United Nations World Water Assessment Programme). 2017. The United Nations World Water Development Report 2017. The untapped resource. Paris. Unecso. Retrieved February 6. 2021.

Wood, A., Blackhurst, M., Hawkins, T., Xue, X., Ashbolt, N. & Garland, J. 2015. Cost-effectiveness of nitrogen mitigation by alternative household wastewater management technologies. Journal of Environmental Management 150: 344-354.

Working Group Memorandum. 2011. Suomesta ravinteiden kierrätyksen mallimaa. Työryhmämuistio mmm 2011:5. [In Finnish]

Wäger-Baumann, F. 2011. Physical and biological methods for the treatment of the liquid fraction of anaerobic digester effluent. University of Natural Resources and Applied Life Sciences, Vienna.

Xu, M., Zhang, Y., Li, Y., Lv, M., Zhu, S., Qian, T. & Fan, B. 2021. Energy recovery potential in coupling of sanitation and agriculture: Techno-economic analysis on resource-oriented sanitation. Waste Management 126: 141-151.

Xue, X., Hawkins, T.R., Schoen, M.E., Garland, J. & Ashbolt, N.J. 2016. Comparing the life cycle energy consumption, global warming and eutrophication potentials of several water and waste service options. Water 8(154): 21.

Ylivainio K., Äystö L., Fjäder P., Suominen K., Lehti A., Perkola N., Ranta J., Meriläinen P., Välttilä V. & Turtola E. 2020. Jätevesilietteen pitkäkestoinen fosforilannoitusvaikutus ja yhteys ympäristö- ja ruokaturvallisuuteen. Jätevesilietteen potentiaali kasvintuotannossa ja vaikutukset ympäristöön ja elintarviketurvallisuuteen (PProduct) -hankkeen loppuraportti. Luonnonvara- ja biotalouden tutkimus 55/2020. [In Finnish]

Zhao, Q. & Liu, Y. 2019. Is anaerobic digestion a reliable barrier for deactivation of pathogens in biosludge? Science of the Total Environment 668(10): 893-902.