Small sanitation systems
Treatment efficiency, sustainability and implementation

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Preface

This doctoral thesis presents the results of my work during the past years in the Urban Water Engineering research group, at the department of Civil, environmental and Natural Resources Engineering at Luleå University of technology. The studies were financially supported by the Swedish Research Council Formas (Grant No. 942-2015-758 and 2019-01438), the Swedish Agency for Marine and Water Management (1:11 Grant No. 1634/20 and 00929-2021, and 1:12), the J Gustaf Richert Foundation, the Interreg Nord European Regional Development Fund (grant no. NYPS 550 20201833) and Region Norrbotten (grant no. NYPS 20201991).

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Brenda Vidal
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Abstract

Current socio-technical wastewater system paradigms are being challenged by emerging global issues such as climate change, environmental degradation and scarcity of resources. Thus, exploration of innovative new urban water management solutions is required that enable closure of resource loops. On-site and decentralised wastewater systems are seen as emerging solutions, partly because of their flexibility, as they increase the potential for recovery and reuse of resources like nutrients. At the smallest scale, on-site wastewater treatment systems are widely present in rural and peri-urban areas globally. However, they often do not meet acceptable levels of nutrient and bacterial discharge, thereby contributing to environmental degradation and health risks. The overall aim of this doctoral dissertation is to improve knowledge and understanding of small sanitation systems in terms of treatment processes and efficiency, their sustainability and implementation. At a greater scale, the dissertation explores the historical and current contexts that have shaped, and are shaping, the existing wastewater sector, dominated by large-scale centralised mixed wastewater collection and treatment, and barriers to implementation of alternative, innovative sanitation solutions with higher resource recovery capacities, like source separating systems.

Sixteen full scale on-site facilities for wastewater treatment in Sweden were investigated in the research the thesis is based upon, including sand filters and package plants. Sand filters followed by alkaline phosphorus filters showed good removals of organic matter and high P-removal capacity (>92%). Six (of 11) investigated package plants showed >78% phosphorus removal (including chemical precipitation and alkaline phosphorus filters as treatment steps). Nitrogen removal was generally low in the package plants, likely because of the low average temperatures, and appeared to be mainly mediated by sedimentation of particulate organic nitrogen. Denitrification was observed to a limited extent in facilities with effective water/sludge recirculation. High densities of indicator bacteria were found in the effluent of most facilities as the removal rates were low, often exceeding the EU Bathing Water Directive’s limits for excellent water quality in terms of intestinal enterococci and *Escherichia coli*. Levels of pharmaceuticals detected in the effluent of package plants were within the range or higher than previously reported in effluents from conventional wastewater treatment plants, and included anti-inflammatories, β-blockers, ACE inhibitors, anticonvulsants and antidepressants. Effluent concentrations of phthalates were below those previously reported in the literature.

A scenario analysis comparing nine on-site sanitation options suggested that source separation of greywater and blackwater and urine diversion were the most sustainable options when nutrient removal and recycling were highly prioritised. A conventional sand filter or drain field were the most sustainable options when nutrient removal and recycling were less prioritised and, (in combination with chemical P-removal) when CO₂ emissions and energy use and recovery were important aspects.

When planning wastewater services for a given area, results of interviews with water professionals confirmed the general trend to opt for a centralising approach, whenever technically feasible, by installing pumping stations and connecting sewer pipes to a main treatment plant, in contrast to building decentralised systems. Reasons mentioned by the interviewees included the robustness of the system, simplicity of operation and maintenance and protection of the receiving waters. Identified barriers hindering implementation of alternative sanitation solutions with focus on resources recovery included factors categorized as legislative factors (lack of requirements and law interpretation), technical factors (immature technologies, uncertainties), organizational factors (lack of initiative, competence and experience) and economic factors (financial limitation, lack of incentives). From a historical perspective, strong governmental control and continuous enforcement of environmental requirements drove the expansion and strengthened the domination of the large-scale centralised sanitation system. Alternative sanitation systems with higher resource recovery capacities have received less attention and institutional support, and their shortcomings were constantly highlighted during the period covered by a historical review (1974-2015).
Sammanfattning

Nuvarande sociotekniska paradigmer för avloppsvattensystem utmanas av nya globala frågor som klimatförändringar, miljöbelastning och resursbrist. Därför behöver innovativa nya lösningar för vattenhantering i städerna som möjliggör cirkulation av resurser utforskas. Små och decentraliserade avloppsvattensystem ses som framtvämmande lösningar, delvis på grund av sin flexibilitet, eftersom de ökar potentialen för återvinning och återanvändning av resurser såsom näringsämnen. I den mindre skalan finns lokala enskilda avloppssystem på landsbygden och i utkant av städer globalt. Dock uppfyller de små avloppssystemen inte acceptabla reningsnivåer vilket bidrar till negativ miljöpåverkan och hälsorisker. Det övergripande syftet med denna doktorsavhandling är att öka kunskapen om små avloppssystem när det gäller reningseffektivitet, hållbarhet och implementering. I avhandlingen utforskas också de historiska likväld som aktuella sammanhang som har format och formar den befintliga avloppssektorn, och dess dominans av storskalig centraliserad hantering av avloppsvatten. Även hinder för implementering av alternativa, innovativa sanitetslösningar med högre potential för resursrecirkulering har studerats.

Sexton fullskaliga markbäddar och minireningsverk undersöktes i denna avhandling. Markbäddar med efterföljande alkaliska fosforfilter visade på god rening av organiskt material och hög fosforavskiljning (>92%). Sex av elva undersökta minireningsverk inklusive kemisk fällning eller alkaliska fosforfilter visade på >78% fosforreduktion. Kväveavskiljningen var generellt låg i minireningsverken, till viss del på grund av de låga medeltemperaturerna, och reningen verkade huvudsakligen ske genom sedimentation av partikulärt organiskt kväve. Viss denitrifikation observerades i anläggningar med vatten/slamrecirkulation. Högta halter av indikatorbakterier analyserades i avloppsvattnet från de flesta anläggningarna och ofta överskred EU:s badvattendirektivs gränser för utmärkt vattenkvalitet gällande intestinala enterokocker och Escherichia coli. De nivåer av läkemedel som uppmättes i avloppsvattnet från minireningsverken låg inom intervallet eller högre än tidigare rapporterade halter i avloppsvattnen från konventionella avloppsentreningsverk, vilket inkluderade antinfiammatoriska medel, β-blockerare, ACE-hämmare, antiepileptika och antidepressiva medel. Däremot var utsläppskoncentrationerna av ftalater lägre än de som tidigare rapporterats.

En scenarioanalys som jämförde nio alternativ för enskilda avloppslösningar indikerade att källsortering av gråvatten och svartvatten alternativt urinsortering var de mest hållbara alternativen när rening och recirkulering av näringsämnen prioritades högt. En konventionell markbädd eller infiltrationsbädd var de mest hållbara alternativen när rening och återföring av näringsämnen var mindre prioriterade, och i kombination med kemisk fosforrening, när CO₂-utsläpp, energianvändning och energiåtervinning var viktiga aspekter.

Resumen

Desafíos globales como el calentamiento climático, la degradación ambiental y la escasez de recursos afectan directamente a los paradigmas sociotécnicos actuales que dan forma a los sistemas de tratamientos aguas residuales. Por este motivo, es necesario explorar nuevas formas de gestionar dichos sistemas de forma innovadora y circular. Sistemas locales y descentralizados están emergiendo como posibles soluciones, por su flexibilidad y potencial para extraer y reusar recursos tales como nutrientes. Los sistemas de saneamiento de aguas domésticas en su escala más pequeña (in situ) están presentes globalmente en áreas rurales y periurbanas. Sin embargo, los efluentes suelen sobrepasar niveles aceptables de nutrientes y patógenos, afectando al medio ambiente circundante y la salud de las personas. El objetivo de esta tesis doctoral es contribuir al conocimiento sobre sistemas pequeños de saneamiento con respecto a la eficiencia de los tratamientos, sostenibilidad e implementación. La tesis intenta asimismo explorar el contexto histórico y actual que ha moldeado el sector de las aguas residuales, un sector dominado por el sistema de tratamiento centralizado con alcantarillado, así como las barreras que impiden la implementación de sistemas alternativos e innovadores que permiten una alta recuperación de recursos, como lo sistemas de separación en origen.

En esta tesis se presentan los resultados de 16 sistemas de tratamiento de aguas residuales in situ investigados en Suecia, incluyendo filtros de arena/grava y plantas de tratamiento pequeñas y compactas. Los filtros de arena/grava seguidos de filtros alcalinos para capturar fósforo, trataron los compuestos orgánicos y el fósforo de forma satisfactoria (> 92%). Seis (de las 11) plantas de tratamiento compactas que fueron investigadas, y que incluían precipitación química y filtros alcalinos, trataron más del 78% del fósforo. El tratamiento de nitrógeno en dichas plantas fue generalmente bajo, posiblemente debido a las bajas temperaturas anuales medias, y se debió principalmente a la sedimentación del nitrógeno orgánico. Los niveles de desnitrificación observados fueron generalmente bajos, y cuando ocurrieron fue primariamente en plantas con un sistema eficaz de recirculación de agua o lodos. Los efluentes de la mayoría de los sistemas presentaron densidades considerables de bacterias indicadoras fecales enterococci y Escherichia coli, sobrepasando con frecuencia los límites establecidos por la Directiva de Aguas de Baño europea. Los fármacos en los efluentes de las dos plantas compactas de tratamiento investigadas estaban en el rango de las concentraciones reportadas en estudios previos, e incluyeron antiinflamatorios, bloqueadores beta, inhibidores ACE, anticonvulsivantes y antidepresivos. Las concentraciones de ftalatos en los efluentes investigados se encontraban por debajo de los niveles reportados con anterioridad en estudios científicos sobre efluentes de aguas residuales.

Los resultados de un análisis de escenarios que comparaba nueve soluciones distintas para tratar aguas residuales in situ indicaron que la separación en origen de aguas negras y grises era la opción más sostenible cuando el tratamiento y recuperación de nutrientes eran priorizados. Cuando esto no era la prioridad, en cambio, un sistema sencillo de filtros de arena/grava o un sistema de infiltración resultaron ser las opciones más sostenibles, y en combinación con el tratamiento químico del fósforo cuando las emisiones de CO₂, el uso y la reutilización de energía se consideraron importantes.

Entrevistas con profesionales del ámbito de la gestión de aguas confirmaron que la tendencia general era optar por centralizar ciertas áreas a la red de alcantarillado público y a una planta tratamiento principal, cuando fuera posible y si las condiciones técnicas lo hacían viable. Los entrevistados mencionaron varias razones que motivaban esta solución, tales como la robustez de esta solución, la simplicidad del funcionamiento y mantenimiento y la protección de las aguas receptoras. Los entrevistados identificaron barreras a la implementación de sistemas alternativos de saneamiento y enfocados en la recuperación de recursos, tales como: legislativas (falta de requisitos o interpretación de la ley), técnicas (tecnologías inmaduras e incertidumbres asociadas), organizativas (falta de iniciativa, competencias y experiencia) y económicas (limitaciones financieras y falta de incentivos).
Desde una perspectiva histórica, el fuerte control gubernamental y el incremento de los requisitos ambientales favoreció la expansión y el dominio de los sistemas centralizados de tratamiento de aguas residuales en Suecia. Sistemas alternativos de saneamiento, que ponen énfasis en la recuperación de recursos, han recibido menos atención y apoyo institucional durante el periodo de estudio (1974-2015), a la misma vez a la que se enfatizaban los problemas que les afectaban.
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List of appended papers

This thesis is based on studies presented in the following appended papers, which are referred to in the text by the corresponding Roman numerals.


**Other publications**


Herrmann, I, Vidal, B, Hedström, A. Slutrapport av projekten "Fosforfällor för små avlopp – hur länge fungerar de?” och "Bakterieutsläpp från små avlopp" [Final report from the projects "Phosphorus filters for small sanitation systems – how long do they work?” and ”Bacteria discharge from small sanitation systems”]. Research report, Luleå university of technology, 2017. (In Swedish)
## Assessment of contribution to the appended papers

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**Responsible** – developed, consulted (where needed) and implemented a plan for completion of the task.

**Shared responsibility** – made essential contributions towards the task completion in collaboration with other members of the research team

**Contributed** – worked on some aspects of the task completion

**No contribution** – made no contribution to completion of the task for valid reason (e.g., joining the research project after its completion)

**NA** – not applicable
1. Introduction

Adequate wastewater planning and sustainable sanitation are highly important for achieving the sustainable development goals (SDGs) set by the United Nations related to the provision of adequate sanitation for all, protection of water resources from pollution (SDG 6), and the reduction of nutrient discharges into the marine environment (SDG 14) (Larsen et al., 2021). Due to the importance of efficient waste disposal, safe drinking water, and avoidance of pollution, they also indirectly, but strongly, contribute to most of the 15 other SDGs. Recent studies have discussed the likelihood of reaching those goals considering the lock-ins of the current socio-technical wastewater sector, suggesting that exploration of innovative new on-site, small grid or hybrid (combinations of centralised and decentralised) systems is required (Hoffmann et al., 2020; Öberg et al., 2020; Estévez et al., 2022). Those solutions should enable resource recovery and reuse, while coping with challenges such as climate change, eutrophication, scarcity of resources and global market disruptions (e.g., in fertilisers production) during turbulent times (Hoffmann et al., 2020; Larsen et al., 2021; Eisa et al., 2022).

Despite the high connection rate to the public sewer network typical of high-income countries, conventional individual on-site systems for domestic wastewater treatment are common in rural and peri-urban areas globally (Stanford and Weinberg, 2010; Gunady et al., 2015; Heinonen-Tanski and Matikka, 2017; Olivier et al., 2019). However, they often do not meet acceptable levels of nutrient and bacterial discharge, due to poor design, construction, maintenance and operation, thereby contributing to environmental degradation and health risks. Various treatment options can enhance desirable features of the systems, such as their nutrient or energy recovery but can also cause impacts in terms of climate change emissions, eutrophication or energy use (Lehtoranta et al., 2014; Molinos-Senante et al., 2014). To contribute to the practical assessment and implementation of on-site systems, effluent discharge parameters (BOD, nutrients, indicator bacteria and micropollutants) and the sustainability performance of the various treatment alternatives need to be well understood.

The varying performance of individual on-site treatment systems, and/or development of new housing areas, often calls for municipal intervention. In such cases, the option frequently arises to connect the areas in need of new or improved sanitation to a central treatment plant (through sewer networks). The decision-making processes usually consist of case-by-case assessments that offer windows of opportunity to implement alternatives to the centralisation option. Estimating the optimal degree of centralisation of an area considering geographical and economic factors is complex and rarely done (Eggimann et al., 2015). Alternative and innovative solutions with higher capacities for resource-recovery, like source-separating systems, are seldom considered, for various reasons, such as lack of knowledge and expertise, immaturity of technology, economic constraints and organizational difficulties (McConville et al., 2017). To counter this problem, in addition to understanding innovative systems there is a clear need to elucidate how historical developments have shaped the current wastewater system, and influences of current
discourses of water professionals, planners and decision-makers on the sector’s future trends.

1.1. Aim and objectives

This thesis focuses on small scale wastewater treatment systems, and is based on studies that addressed several key aspects using a variety of methods. The overall aim of the underlying research was to improve knowledge and understanding of small sanitation systems, including the treatment processes involved and their efficiency, sustainability and implementation in the context of a paradigm shift from “waste treatment” to “resources recovery”. The thesis also explores the historical and current contexts that have shaped, and are shaping, the existing wastewater sector and affect the implementation of alternative solutions with focus on resource-recovery.

More specifically, the three main research objectives were to:

1. evaluate the processes affecting treatment efficiency and effluent concentrations in existing small scale wastewater systems with regard to removal of organic matter, nutrients, indicator bacteria, and micropollutants, including sand filters, and package plants of different sizes and loads under varying conditions e.g., variations in temperatures (Papers I-III),
2. assess the overall sustainability of small-scale wastewater systems including conventional and source separation systems using a multi-criteria approach, sustainability indicators, different scenarios and preferences (Paper IV), and
3. investigate the historical context, socio-technical regimes and current heuristics and drivers influencing the development and implementation of centralised or decentralised sanitation systems following a transition towards more resilient, resource-efficient and sustainable wastewater systems (Papers V and VI).

1.2. Structure of the thesis

The structure and content of this thesis are visualised in Figure 1. The underlying research focused on the treatment efficiency and sustainability of decentralised wastewater treatment systems at various scales of implementation. On-site sanitation systems involving various treatment technologies were evaluated in terms of removal of nutrients, indicator bacteria, organic matter, micropollutants and sustainability aspects. An investigation of drivers of the implementation of decentralised or centralised treatment systems in rural and peri-urban areas and analysis of the historical development of sanitation systems with focus on resource recovery frames the work in a context of transition towards sustainable sanitation.
The thesis has six chapters. First, Chapter 1 presents a brief introduction to the research topic and research objectives. Chapter 2 describes the scientific state-of-the-art and knowledge gaps found in the scientific literature regarding small sanitation systems, their treatment efficiency and processes involved, scale of implementation and sustainability, from the smallest setting (on-site) to the largest (centralised systems). Chapter 3 presents the methods used in the studies the thesis is based upon: wastewater sampling, system analysis and qualitative research methods (interviews, surveys and reviews). Chapter 4 presents synthesised results from the studies reported in the six appended papers combined and results that are not reported elsewhere. Chapter 5 discusses findings of the studies in relation to the existing scientific literature. Finally, the conclusions are summarised in Chapter 6.
2. Background

2.1. Treatment in on-site sanitation systems

On-site sanitation systems (OSSs) are used to collect, store and/or treat domestic wastewater in non-sewered areas that are not connected to centralised wastewater treatment plants. They are intended to reduce contents of suspended solids, floatable grease, organic matter, nutrients and pathogens prior to discharge into the receiving environment (USEPA, 2002). Conventional on-site treatment systems consist of a septic tank, where the effluent is pre-treated (mainly by particle sedimentation), and a subsequent soil infiltration system. These are the most common on-site wastewater treatment systems used, for example, in North America (USEPA, 2002), Australia (Green and Ho, 2005), France (Olivier et al., 2019) and Nordic European countries (Heistad et al., 2006; Olshammar et al., 2015; Heinonen-Tanski and Matikka, 2017). Constructed wetlands are also implemented in areas with available space, with subsurface systems being more common in Europe and free water surface systems more common in North America and Australia (Vymazal, 2011). Package plants with biological processes such as activated sludge or attached growth have also increased in popularity in recent years (Johannessen et al., 2012; Olivier et al., 2019; Olshammar, 2021). Despite their long history and widespread use, OSSs often underperform due to inappropriate installation and operation which impairs their treatment capacity, leading to the discharge of poorly treated wastewater (Palm et al., 2002; Heinonen-Tanski and Matikka, 2017) and causing extra costs and risks for public health and water resources (USEPA, 2002).

The treatment efficiency of systems based on soil infiltration depends on the wastewater properties, filter material used, temperature, biofilm development and hydraulic loads (Rolland et al., 2009; Wilson et al., 2011). Despite their effective removal of organic matter and suspended solids (Pell and Nyberg, 1989), the P-removal efficiency of the soil and sand/gravel infiltration materials varies considerably, depending on the occurrence of adsorption and precipitation processes, especially in a long-term perspective (Eveborn et al., 2012; Batista Seguí et al., 2017). Solutions based on physico-chemical processes, including chemical dosing to promote phosphorus (P) precipitation, have been developed to increase the systems’ P removal capacity (Bunce et al., 2018), with reactive filters for P removal providing favourable results (Gustafsson et al., 2008; Cucarella and Remman, 2009; Jensen et al., 2010). Nevertheless, further research is needed on the reliability of P-removal systems using active media in full-scale applications and over prolonged periods of time, a consideration that may limit their long-term sustainability (Bunce et al., 2018).

Conventional processes similar to those used in the larger wastewater treatment plants (WWTPs) are applied in package plants, including primary, secondary and in some cases tertiary treatments, and they are automatized when processes such as chemical precipitation, pumping and/or aeration are included (Libralato, 2013). Sequencing batch reactors, activated sludge or attached growth reactors are commonly applied (Hellström and Jonsson, 2006; Vilpas and Santala, 2007). The treatment processes are generally well
known and the facilities can deliver high removal efficiencies, but they are sensitive to operational disturbances like changes in the wastewater load and failures of the dosing equipment (Hellström and Jonsson, 2006).

Investigations of the quality of effluents from on-site wastewater treatment facilities have recorded substantial variations in treatment efficiencies among both conventional systems with sand filters and package plants (Vilpas and Santala, 2007; Kauppinen et al., 2014; Heinonen-Tanski and Matikka, 2017; Martikainen et al., 2018; Olivier et al., 2019). Variations in P removal have also been observed in column experiments with mineral-based sorbents (Renman and Renman, 2010; Nilsson et al., 2013; Herrmann, et al., 2014). Moreover, effluents of sand filters and package plants often exceed the European limits for good quality bathing waters in terms of enteric microorganisms which are stipulated in the EU bathing water directive (EU, 2006), and thus contribute to the poor quality of the bathing waters (Heinonen-Tanski and Matikka, 2017). Package plants with continuous-flow and activated sludge systems reportedly remove microorganisms (faecal coliforms and viruses) in the range of 1 to 2 orders of magnitude (USEPA, 2002). A recent comparison of five package plants (with rotating biological conductors, biofilters and active sludge process) and three sand filters (with and without P precipitation treatment) reported higher concentrations of *E. coli* and enterococci in the effluents of the package plants than the sand filters (Heinonen-Tanski and Matikka, 2017). In a pilot plant study, suspended growth bioreactors with membranes immersed in an aeration tank were found to remove indicator microorganisms more efficiently (4 orders of magnitude) than other tested options, such as activated sludge with denitrification and sand filtration and up-flow anaerobic sludge blanket treatment (Ottoson et al., 2006). Furthermore, studies on conventional OSSs in cold climates suggest that the microbial purification efficiency may be reduced at low temperatures, with reported consequences for nutrient removal ranging from no significant effect (Christopherson et al., 2005; Williamson, 2010; Kauppinen et al., 2014) to substantial reduction in P removal capacity, for example of alkaline P-filters in column experiments (Herrmann et al., 2014). Bunce and Graham (2019) found that the influent temperature was an important, but not dominant, predictor of the reliability of small wastewater plants (< 250 PE), using a random forest classification model. Despite the importance of assessing the hygienic quality of OSSs’ effluents because of its potential impact on human health, pathogen removal has been less studied than nutrient removal, likely because of the existing legislation on the latter (Martikainen et al., 2018). Thus, further research is needed on the long-term removal of bacteria and viruses from OSSs, due to the risk of contamination of surface and groundwaters used for drinking and recreational purposes. There are no legal regulations regarding enteric microorganisms’ densities in effluents in Sweden at the time of writing, despite their presence in surface waters (Heinonen-Tanski and Matikka, 2017). Consequently, limits in the EU bathing water directive are often used as guidelines (EU, 2006) with maximal densities for excellent bathing quality in inland and coastal waters of 200 and 100 CFU/100 mL for intestinal enterococci, and 500 and 250 CFU/100 mL for *E. coli*, respectively.
Besides organic compounds, nutrients and pathogen organisms, domestic wastewater effluent also contains other xenobiotic pollutants. Screening studies on micropollutants emissions from OSSs have identified large numbers of compounds in the wastewater, including pharmaceuticals, personal care products, pesticides, phosphorus-containing flame retardants, artificial sweeteners and perfluoroalkyl substances (PFASs) (Blum et al., 2017; Gros et al., 2017). Significant quantities of these substances reach the aquatic environment (Gago-Ferrero et al., 2017) but no clear differences have been found in terms of concentrations and removal rates between OSSs and WWTPs (Gros et al., 2017; Schaider et al., 2017). Vegetated vertical flow constructed wetlands appeared to be more efficient at removing micropollutants than horizontal flow constructed wetlands, fixed-film reactors (referred to as non-vegetated biofilters) with P-filter units, probably because of the combined effects of vegetation and better oxygenation (Matamoros et al., 2009; Vymazal, 2011). Granular activated carbon alone or in combination with Polonite® alkaline material resulted in the highest removal efficiencies (average values above 97% for all micropollutants) in a study where the treatment efficiency of five different sorbents was evaluated with respect to 83 micropollutants (Rostvall et al., 2018).

Treatment of source-separated wastewater fractions (blackwater and urine)

Ensuring that removal rates of both nitrogen (N) and P are good appears to be technically difficult on small scales (Bunce et al., 2018). Thus, source separation systems have been proposed, and developed, in efforts to remove and recover nutrients more efficiently (Larsen et al., 2016). The different treatment options can be applied on-site or off-site, depending on the configuration and context. The faecal fraction is the main source of microorganisms in domestic wastewater, so it is reasonable to treat them separately from a hygienic perspective (Winker et al., 2009), either by separating the faeces or the blackwater (which contains faeces, urine, flushing water and toilet paper). The urine fraction is typically pathogen free but cross-contamination from faeces may occur during collection in urine diverting toilets. The intrinsic ammonia found in undiluted urine has been shown to be sufficient for the inactivation of bacteria, phages and Ascaris eggs at temperatures above 20°C in urine (Vinnerås et al., 2008; Nordin et al., 2009). For the treatment of source-separated urine, alkaline dehydration has emerged as an efficient and hygienic solution for minimizing volumes of urine and producing a marketable solid fertilizer product (Senecal et al., 2018; Simha et al., 2018). By this approach, the urine is first alkalinized with chemicals (e.g., lime) or waste products (e.g., wood ash) to prevent urea hydrolysis and minimize the nitrogen losses and is then dehydrated (Senecal et al., 2018). The so-called VUNA recycling technique stabilizes source-separated urine through partial nitrification and removes pharmaceutical residues through an activated carbon filter (VUNA, 2018). The result is a urine-based liquid fertilizer (Aurin) approved by the Swiss Federal Office of Agriculture.

Anaerobic treatment of blackwater or addition of either urea or ammonia are applied treatment processes where the degree of inactivation of pathogens present in faecal matter depends on the temperature, pH and treatment duration (Larsen and Maurer, 2011). For the treatment of blackwater mixed with kitchen waste, anaerobic digestion in up-flow
anaerobic sludge blanket (UASB) septic tanks have shown effective removal of COD at 25°C and effluent nutrients in soluble forms (ammonium and phosphate), making it an attractive option for agricultural application (Kujawa-Roeleveld et al., 2005). Higher \textit{E. coli} reduction was however needed to be able to comply with the standards for irrigation according to the same study (Kujawa-Roeleveld et al., 2005). Wet composting and urea hygienization of blackwater have also been studied and implemented (McConville et al., 2015). A study about urea and lime treatment of source-separated blackwater from low-flush toilets showed positive results in the inactivation rates of \textit{Salmonella} and \textit{E. coli} (Fidjeland et al., 2015). The treatment time could be reduced at least six-fold by adding 0.1% urea (wet weight) to meet the treatment requirements compared to the control while increasing the fertilizing value of the treated blackwater thanks to the extra nitrogen from the urea (Fidjeland et al., 2015).

2.2. Processes involved in the removal of phosphorus, nitrogen, microorganisms and micropollutants in wastewater treatment systems

\textit{Phosphorus}

Phosphorus removal from wastewater streams started to be implemented to decrease the eutrophication impact in receiving waters (Yeoman et al., 1988). Additionally, stricter discharge requirements and the increasing limitation of phosphorus availability on the global scale have boosted initiatives to enhance its removal and recovery (Cordell, 2013). Phosphorus can be removed from wastewater by physicochemical methods and/or advanced biological treatment (Bio-P) (Yeoman et al., 1988). In conventional soil-based on-site systems with gravel and sand for filtration/infiltration, biological removal of phosphorus is of minor importance as the P-uptaking biota (plants and microorganisms) is not generally harvested (Eveborn et al., 2012). Physicochemical processes for phosphorus removal are therefore more important in soil-based systems. Phosphates are adsorbed by most soil particles often forming a wide range of stable minerals after combination with metals cations such as iron, aluminium, manganese and calcium (Robertson et al., 1998). The phosphorus removal process is finite as it is limited by the soil’s capacities for adsorption processes (on Fe and Al oxide phases) and mineral precipitation reactions (of Al, Fe, Mn (acidic soils) and Ca (basic soils) phosphates), which mainly occur in the vadose (unsaturated) zone and are often difficult to distinguish from one another (Robertson et al., 1998). Studies on phosphorus removal in soil-based treatment systems have shown varying results (Christopherson et al., 2005; Vilpas and Santala, 2007; Eveborn et al., 2012; Batista Seguí et al., 2017), likely due to variations in the soil, systems age, sampling approach and assessment methodologies (e.g., mass balance calculations, sorption isotherms, simulation models). Nevertheless, it has been suggested that the long-term phosphorus removal rates in soil-based systems are lower (about 12% on average) than generally reported (Eveborn et al., 2012). The P removal can be enhanced by processes involving adsorption or direct precipitation by using reactive material with sorptive properties specifically targeting phosphorus (Arias et al., 2001). The inorganic P accumulates on the surfaces of reactive components of the filter media, e.g., calcium or iron, and thus the removal capacity depends on the mineral content of the reactive material (e.g., chemical composition, structure, particle size) and pH, with
higher sorption processes occurring with increasing pH values (Cucarella and Renman, 2009; Bunce et al., 2018).

Chemical dosing of metal salts (e.g., ferric/ferrous and aluminium salts) is the most commonly used method to promote phosphorus precipitation in the septic tank or process units/reactors of package plants (e.g., activated sludge). The process is based on the precipitation of the negatively charged dissolved phosphate (PO$_4^{3-}$) by metal ions like aluminum (Al$^{3+}$) or iron (Fe$^{2+}$/Fe$^{3+}$), forming phosphate compounds, such as FePO$_4$ and AlPO$_4$ with very low solubility (Kroiss et al., 2011). The dissolved P fraction in the treated effluent depends on the chemicals used (which create flocs with different properties), the dosage related to the phosphorus load and environmental conditions such as pH and temperature (Kroiss et al., 2011). The P precipitates and residual solids are removed by settling or filtration, with typical removal rates proportional to the mass of added precipitant, and often achieving effluent P concentrations of 1 mg L$^{-1}$ after conventional gravity settling (Tchobanoglous et al., 2014; Bunce et al., 2018). However, more comprehensive research on full-scale facilities is needed to determine, elucidate and enhance OSSs’ P-removal capacities and performance in practice to ensure that they meet new, stricter regulations for discharges.

**Nitrogen**

Biological nitrification/denitrification is the main process implemented in conventional on-site sanitation systems for nitrogen removal. The process is affected by the configuration (preanoxic, postanoxic, simultaneous nitrification-denitrification) and set-up of the treatment system (recirculation, retention time) as well as factors such as oxygen availability, temperature and pH that strongly affect the nitrification and denitrification reactions (Taylor Eighmy and Bishop, 1989; Nakhla and Farooq, 2003; Oakley et al., 2010). The nitrification process involves two steps, mediated by ammonia-oxidizing bacteria and nitrite-oxidizing bacteria, whereas denitrification is primarily mediated by heterotrophic bacteria in a series of redox transformations that reduce nitrate to N$_2$. The microbial activities and growth rates, which are influenced by temperature, will determine the facilities’ maximal treatment degrees. Certain degrees of nitrification can be expected to occur in sand filters, after a lag period that can last several months and as long as there are aerobic conditions (Laaksonen et al., 2017). Optimal conditions for denitrification, i.e., anoxic conditions and sufficient organic substrate (biological oxygen demand: BOD) for the heterotrophic organisms are difficult to establish in sand filters because of the type of treatment set-up e.g., lack of recirculation (Laaksonen et al., 2017). Oakley et al. (2010) suggested that plant uptake of nutrients and/or denitrification processes can be enhanced at the discharge site with shallow trenches and subsurface drip distribution/irrigation systems that discharge within the root zone. Anaerobic conditions may potentially be present in septic tanks, although a recent review (Shaw and Dorea, 2021) concluded that the activities of anaerobic microbial communities in biodegradation processes cannot be confirmed because of the lack of robust data. Package plants supporting submerged attached growth processes (with a primary settling tank, bioreactor with media and clarifier) and suspended growth (activated sludge) processes provide the conditions required for nitrification and denitrification if the sludge in the clarifier is
recirculated into the primary settling tank or a carbon source is available after nitrification has occurred (Oakley et al., 2010; Olivier et al., 2019). Temperature is an important factor for high nitrogen removal rates. This is because low temperatures impair bacterial activity and growth (thresholds for nitrification and nitrifies are about 8 and 4°C, respectively), so nitrogen effluent concentrations are likely to be inversely related to temperature (Sharma and Ahlert, 1977; Taylor Eighmy and Bishop, 1989). Nitrifying bacteria appear to be sensitive to phosphorus and organic substrate deficiency (Kroiss et al., 2011). Previous studies have suggested that alkaline reactive materials for phosphorus removal creates an alkaline environment that inhibits denitrification, although some nitrogen removal can be achieved, probably caused by NH₃ volatilization at high pH values (> 9.3) (Renman et al., 2009). Nitrogen concentrations in effluents from on-site package plants tend to be highly variable, and have high means, due to considerable temporal variability in flowrates and loads of the inputs, depending on the specific water uses (Oakley et al., 2010). More studies of on-site systems operating under field conditions with varying loads and temperature are needed to further quantify nitrogen removal performance capabilities.

**Microorganisms**

Bacteria and viruses can move relatively long distances through the soil under saturated conditions, increasing risks of contamination of groundwater and surface waters (Hagedorn et al., 1978; Scandura and Sobsey, 1997). Stevik et al. (2004) reviewed the mechanisms influencing the retention and removal of bacteria in infiltration systems, which are mainly straining and adsorption. Physical straining refers to the blocking of movement when the pores are smaller than the bacteria, and it is influenced by the grain size distribution of the porous media, bacterial cell size and shape, hydraulic loading and clogging of filter media. Adsorption dominates in porous media where the pores are larger than the bacteria and may be influenced by physical (e.g., presence of organic matter and biofilm), chemical (e.g., ionic strength and pH) and microbiological (e.g., hydrophobicity and bacterial density) factors (Stevik et al., 2004). Microorganisms are often attached to/associated with particles and their densities are typically reduced by 50-90% (0.5-1 log₁₀) in the pre-sedimentation phase (e.g., in the septic tank) (Stenström, 2013). Soil, loading and hydraulic conditions affect the microorganisms’ removal in soil infiltration systems, but it is generally in a similar range (about >2 logs for bacteria and viruses) as for conventional wastewater treatment plants (without disinfection) (Stenström, 2013). Adsorption of bacteria to porous media is reportedly strongly linked, positively, to temperature (Hendricks et al., 1979), and reduced with decreasing temperatures due to changes in the viscosity of the bacterial surfaces and water, reductions in chemisorption and changes in the organisms’ physiology (Fletcher, 1977). The lower the temperature, the less bacteria are expected to be adsorbed onto a solid phase (Barrios-Hernández et al., 2020). In package plants with activated sludge processes, the removal of microorganisms mainly occurs through sedimentation and adsorption or incorporation into the biological floc, encapsulation within the sludge solids and predation by other microorganisms. The removal of pathogenic microorganisms during activated sludge
treatment is highly variable, depending on the type of organism and retention times, with removal rates for pathogens ranging from 40 to 90% (Henze et al., 2008). Additionally, increases in retention times in batch reactors (e.g., through increases in reaction phases) have been associated with increased reductions of coliform bacteria (Ng et al., 1993). Previous studies have discussed the effect of alkaline P-sorption media on the removal of pathogens as the high pH (>12) contributes to the initial inactivation of bacteria (Jenssen et al., 2005; Heistad et al., 2006; Nilsson et al., 2013a). Results from column experiments suggest that pH significantly affects the removal of bacteria in Polonite® alkaline filter media loaded with high concentrations of organic material (Nilsson et al., 2013b). Additionally, the relation between efficient removal of pathogens and nutrients has been examined in both pilot and full-scale studies with sand filters, but no clear general correlation has been identified (Kauppinen et al., 2014; Martikainen et al., 2018). Further research is therefore needed to improve knowledge of the pathogen discharge and removal efficiencies of OSSs, because of the associated human health risks.

**Micropollutants**

On-site sewage treatment systems are considered non-homogenous sources of micropollutants to the environment due to the high variability of their effluent concentrations, and their emissions reportedly reach surface waters in significant quantities (Gago-Ferrero et al., 2017). Micropollutants’ fates and likelihoods to escape wastewater treatment processes are affected by several characteristics, such as their affinity to attach to surfaces (affected by their hydrophobicity) and biodegradability (affected by their bioavailability) (Luo et al., 2014). The non-volatile and polar compounds are most likely to remain untreated (Luo et al., 2014). The treatment plant’ design and operating conditions (e.g., sludge and hydraulic retention times) has been suggested to influence the removal of micropollutants because it affects the diversity of the microbial community, the time available for biodegradation and sorption processes (Fernandez-Fontaina et al., 2012; Hube and Wu, 2021). Wastewater characteristics, such as pH and temperature, may also influence the removal of micropollutants (Gao et al., 2014). The biodegradation and partition processes can be affected by temperature and micropollutant’ removal has been shown to be reduced at colder temperatures (about 12°C in the winter samples) probably because of decreased microbial activity (Vieno et al., 2005; Nie et al., 2012). However, this influence has also been reported as negligible in the temperature range 16–26°C for pharmaceuticals and personal care products (Suarez et al., 2010). Positive correlations between ambient temperature and phthalate removal have also been detected in a study of effects of treatments including activated sludge, batch reactor and upflow anaerobic sludge blanket (UASB) treatments, with lower average removals recorded at 17 °C than 30°C, especially for DEHP, DEP and DBP (Gani and Kazmi, 2016). If biodegradation is the main phthalate removal mechanism, low temperature may have a significant effect as it influences the microbial activity and
metabolism, in contrast to having adsorption as a main removal mechanism (Gani and Kazmi, 2016).

2.3. Sustainable sanitation systems – How to assess them?

Defining and measuring the sustainability of wastewater systems have been major concerns in the last two decades, as shown for example by Lundin et al. (1999), Hellström et al. (2000) and Balkema et al. (2001). Sustainability can be defined based on specific criteria or indicators that are measurable, or at least assessable (Hellström et al., 2000). Sustainable technology for domestic wastewater treatment has been defined as technology that does not threaten the resources’ quantity and quality, which will vary depending on the conditions and scenarios (Balkema et al., 2001). Similarly, sustainable sanitary systems have been defined as systems that provide the necessary services without compromising human health, use a minimal amount of resources, and do not have negative long-term environmental impacts (Lundin et al., 1999). Financial stability, use of local water resources, energy-neutrality, responsible nutrient management and access to adequate sanitation for all were identified by Daigger (2009b) as important elements of sustainable sanitation solutions.

To assess the sustainability of treatment options for domestic wastewater, systematic and comprehensive methodology is required to cover the main dimensions (economic, environmental and social-cultural) of sustainability (Bradley et al., 2002). In earlier research a wide range of methods has been applied for assessing the sustainability of sanitation systems, for example use of indicators based on sustainability principles (Hellström et al., 2000; Balkema et al., 2002; Tjandraatmadja et al., 2013), and comparison of single indicators versus multidisciplinary indicators (Molinos-Senante et al., 2014). Furthermore, life cycle approaches (Tidåker et al., 2007; Lehtoranta et al., 2014; Xue et al., 2016; Schoen et al., 2017), environmental systems analyses (Weiss et al., 2008; Spångberg et al., 2014) and sustainability assessments (Molinos–Senante et al., 2014; Diaz-Elsayed et al., 2017) have been applied with the same purpose. A review of the life cycle assessment (LCA) approach applied to wastewater treatment systems highlighted the need to link LCA methodology to economic and social evaluations to obtain an overall picture of sustainability (Corominas et al., 2013). The challenges of assessing the sustainability of small wastewater treatment systems was thoroughly discussed by Molinos-Senante et al. (2014), who compared seven technologies for secondary treatment in small WWTPs by aggregating system indicators into a composite that provided a global measure of sustainability. The sustainability of five water and sanitation systems has been compared by Xue et al. (2016) and Schoen et al. (2017), using normalized metric scores encompassing the life cycle eutrophication potential, energy consumption, global warming potential, cost and human health impact. In addition, a scenario with increasing variability of climate-related factors, such as precipitation and temperature, was discussed by Kohler et al. (2016) with regard to the fragility (the degree to which a system loses functionality) of on-site sanitation systems. The study suggested that variability of weather (prolonged precipitation patterns, high temperatures, wetter-
than-normal months, peak stream flow) affects on-site sanitation systems’ performance. The fragility of the systems (e.g., number of repairs) and the effects in terms of costs, environmental impacts and health risks should be considered in the systems’ planning, design and operation (Kohler et al., 2016).

Weiss et al. (2008) applied LCA methodology to study the environmental impacts and use of natural resources of four on-site sanitation systems (infiltration, chemical precipitation and P removal using two types of filter media). The system with chemical precipitation showed the most favourable results from an environmental (energy use, global warming and eutrophication potential) and resources conservation perspective, whereas the alternatives with reactive alkaline P-filters showed low eutrophication impacts. The potential trade-off between reduction of emissions and increase of life cycle impacts (global warming and eutrophication potential) of six alternatives were studied by Lehtoranta et al. (2014). The study concluded that the major contributors to both life cycle impacts were the direct on-site emissions, electricity consumption and sludge treatment.

Previous studies have evaluated and compared conventional on-site facilities, connection to a main WWTP and more alternative innovative options like urine diversion or blackwater separation. Lennartsson et al. (2009) found that the less conventional source separation systems provided better performance in terms of economic and environmental parameters than the conventional system (based on filtration) and centralisation option. However, they ranked lower in terms of socio-cultural criteria as the conventional systems benefited from formal institutional recognition, due to existing legislation and regulation. Tidåker et al. (2007) compared three on-site sanitation systems (urine separation, blackwater separation and chemical precipitation in a septic tank) by evaluating the energy use in a life cycle perspective, the recycling potential and the nutrient emissions. They found that the source separation options had higher recycling potential, although considerable electric energy was needed for the blackwater treatment (aeration and stirring for liquid composting) while the chemical precipitation system had a higher use of fossil fuels because of the chemicals production. Package plants, including a sequencing batch reactor and a system with attached growth using plastic carriers (referred to as a biofilter), reportedly have greater global warming and eutrophication potentials than soil-based and source separation systems (Lehtoranta et al., 2014). The potential of source separation systems in terms of resource-efficiency has been demonstrated in comparison to end-of-pipe technology (Spångberg et al., 2014; Xue et al., 2016). Urine diversion and blackwater separation have been shown to have lower global warming impact (Lundin et al., 2000; Besson et al., 2021) and higher energy efficiency than conventional centralised options, although ammonia emissions from storage and land applications of urine and blackwater must be tackled to decrease the eutrophication and acidification impacts (Spångberg et al., 2014). In a comparative analysis of sanitation systems focusing on resource recovery, Spuhler et al. (2020) discussed effects of the technological components and configurations on the overall resource recovery performance and sustainability. They concluded that short systems (in terms of length and number of technologies included) that close the nutrient loops at the
lowest possible level (fewer treatment steps, less losses) should be prioritized, and the waste streams should be separated as much as possible to maximize recovery potential. These conclusions call for new spatial concepts of decentralised and hybrid wastewater systems. The sustainability and impacts from centralised and decentralised wastewater systems have been investigated by Risch et al. (2021) following LCA methodology and including 18 categories, such as global warming, eutrophication, ecotoxicity, human carcinogenic toxicity, land use, scarcity of resources and water consumption. They found that centralised systems resulted in higher impacts related to the use of resources because of the sewer infrastructure, but performed better than decentralised systems in terms of effects on ecosystems’ quality because of the well-managed air emissions and effluent discharges (Risch et al., 2021). In addition, in an investigation based on LCA methodology, Besson et al. (2021) found that source separation systems (urine and blackwater) were better options than a centralised wastewater resources recovery plant treating mixed wastewater due to lower N₂O emissions (>60%) and the associated reduction in N fertiliser production.

The selection of the system boundaries will inevitably influence the scale of the impacts of compared systems, and source separation alternatives are often advantageous when extended system boundaries like replacement of mineral fertilizers, are included (Lundin et al., 2000). Systems analysis-thinking is required to meet the increasing demand for sustainable water management systems, and the development of frameworks that integrate sustainability indicators (Hellström et al., 2000). Environmental indicators based on LCA and material-flow analysis are often used when comparing different OSSs (e.g., Weiss et al., 2008; Thibodeau et al., 2014; Diaz-Elsayed et al., 2017; Risch et al., 2021). However, more knowledge is needed of ways to comprehensively assess the systems’ technical robustness, factors that influence emissions during specific phases (waste collection, treatment, storage and use), and integrate both qualitative and quantitative indicators in overall sustainability assessments.

In that regard, multicriteria analysis methods are widely used to systematically assess different alternatives by combining individual criteria and incorporating weights to represent the decision-maker’s priorities (Dodgson et al., 2009). The multi-criteria decision analysis literature describes numerous methods for aggregating and weighting selected criteria or indicators (Rowley et al., 2012; Kabir et al., 2014; Tscheikner-Gratl et al., 2017). The importance of choosing the right method lies in the fact that its very choice introduces subjectivity into the analysis (Rowley et al., 2012). The ELECTRE family of outranking, preference-aggregation based methods for multi-criteria decision analysis, initially proposed in the mid-sixties (Roy, 1978) have been widely applied in sustainability-related research (Velasquez and Hester, 2013). Kabir et al. (2014) found that they comprised the second most frequently applied methodology in water and wastewater research after the analytic hierarchy process, which were used in 15.1 and 28.3% of the published studies in the filed during the period 1980–2012. An important novelty in ELECTRE III was the introduction of pseudo-criteria, with use of preference thresholds to define a “buffer zone” between strict preference and indifference when
comparing two alternatives, in contrast to the use of true-criteria, which is based on strict preference for the best performance and does not account for uncertainties (Rowley et al., 2012). Adding flexibility to the comparisons reduces the method’s sensitivity to input data variations, so uncertainty and vagueness are taken into account (Bertrand–Krajewski et al., 2002). ELECTRE III is a non-compensatory aggregation method, hence a bad score for one indicator is not offset by good scores for other indicators, allowing the method to provide a strong sustainability perspective (Rowley et al., 2012).

2.4. Different scales for wastewater management

Wastewater treatment systems can be managed by a wide array of wastewater system options, with diverse scales of implementation and functions. According to definitions provided by Tchobanoglous and Leverenz (2013), centralised wastewater systems collect wastewater and sometimes stormwater from large urban and peri-urban areas through an extensive network of sewage pipes and pumps for transport to a central treatment plant, usually located near a point of convenient discharge into surface water. Decentralised systems, in contrast, are stand-alone systems that collect, treat and dispose of or reuse small wastewater flows at or near the point of wastewater generation, although they often rely on centralised facilities for management of biosolids/sludge (Tchobanoglous and Leverenz, 2013). Decentralised systems can serve individual or isolated buildings (on-site systems), residential clusters or small communities (distributed systems) and be owned by property holders, water utilities, or set of common ownership (Sharma et al., 2013; Tchobanoglous and Leverenz, 2013). Satellite or hybrid wastewater systems incorporate elements of both centralised approaches (for processing sludge and source-separated streams) and decentralised approaches (for treating and reusing the remaining wastewater).

The centralisation of wastewater streams has provided vital services to urban and peri-urban communities but they have strong path dependencies and both technological and institutional lock-in effects (Hoffmann et al., 2020). Many authors suggest that dealing with current and future challenges that affect the water sector, like rapid urbanization, eutrophication, climate change, water and resource scarcity and aging infrastructure may require exploration of new sanitation solutions, in which decentralised wastewater systems can play an important role (Sharma et al., 2010; Libralato et al., 2012; Brands, 2014; Hoffmann et al., 2020; Öberg et al., 2020). Some positive aspects of decentralised sanitation systems are related to lowering the reliance of fossil fuels by linking use of renewable energies, minimizing transport distances and volumes, and decreasing the energy demand for commercial fertilizers (as nutrients are supplied from wastewater to farmland) (Rittmann, 2013). The elimination of stormwater and other inflows into the pipe systems, as well as the possibilities for implementing source separation, makes decentralised systems attractive from a technical perspective (Tchobanoglous and Leverenz, 2013). Institutional setups and weak organizational models (Larsen et al., 2016) and users’ behaviour and acceptance (Brands, 2014) rather than undeveloped technology, have been identified as main sources of failure in the implementation and performance
of decentralised systems. The economies of scale of decentralised systems, one of the main drawbacks and sources of criticism in comparison to centralised systems, could also be seen as a key strength as decentralised systems are more flexible and have potentially lower costs per unit (Maurer, 2013), although on-site systems are highly affected by the economies of density, which depend on the management approach (e.g., routing and scheduled emptying) (Eggimann et al., 2016a). The flexibility of off-grid or decentralised systems allows them to respond better to variations in conditions that are difficult to quantify in a changing world context, such as socio-economic changes (e.g. public goals and environmental concerns), variables linked to population dynamics (e.g. water consumption trends) or technological development (Eggimann et al., 2016b). The need to equalize flow, the higher energy/resources inputs per unit of treated effluent and the physical footprint of decentralised systems remain inherent limitations (Tchobanoglous and Leverenz, 2013).

Despite the current developments in centralised systems and increasing attention to decentralised systems for wastewater management, the optimal degree of wastewater infrastructure centralisation or decentralisation for a particular region (with consideration of sustainability aspects) still remains challenging. Eggimann et al. (2015) developed a two-step technoeconomic heuristic model for optimization of the distribution of the sewer network and wastewater treatment plants based on cost and sewer-design parameters (e.g., topography). According to their findings, the optimal degree of centralisation declines with increasing complexity of terrain and settlement dispersion, and application of the model to a Swiss community suggested that the prevailing sewer system was over-centralised. Decentralised systems provide better environmental performance than centralised systems when there is a low number of households to serve with short sewer connections (Risch et al., 2021). In addition, separating the waste streams at source as much as possible and closing the nutrients loop at the lowest possible level are key to optimising resource recovery from sanitation systems (Spuhler et al., 2020). The local hydrology, available water sources, water demand, local energy and nutrient-management situations, existing infrastructure, and utility governance structure all affect the optimum scale for a particular area (Daigger, 2009).

There is a large body of literature regarding the implementation and performance of decentralised wastewater systems at the expert’ and scientific levels (Orth, 2007; Libralato et al., 2012; Brands, 2014; Fam et al., 2014; Eggimann et al., 2016b; Hoffmann et al., 2020). However, technological transfer into practice is still low, partly due to the lack of awareness and recognition of decentralised systems’ potential benefits and the ‘business as usual’ mentality commonly found at institutional and administrative levels (Capodaglio et al., 2017). In the coming years, many existing small facilities will be centralised by connection to a main wastewater plant or upgraded, and new systems will be built to provide services to new development areas or decrease the pressure on the existing wastewater treatment plants working at full capacity. The decision to centralise, upgrade or build a new plant offers an opportunity to implement solutions that follow the new paradigm for water management, where the system is adapted to local conditions and
needs. Solutions that also promote closure of the loops via a fit-to-purpose approach. Understanding the heuristics involved in current water management will help efforts to understand what is needed in the transition towards new sanitation paradigms.

2.5. The socio-technical regime in the wastewater sector and its readiness for change

Societal functions, such as provision of a potable water supply and sanitation, are fulfilled by socio-technical systems consisting of a cluster of elements, including knowledge, natural resources, organisations, regulation, user practices, markets, cultural meaning, infrastructure and maintenance (Geels, 2005). A socio-technical regime consisting of production and consumption systems often has a strong lock-in in terms of dependence between important groups, such as customers, suppliers, and the political and educational systems (Geels, 2005).

The conventional sewer system is characterized by water toilets connected to sewers conveying the wastewater to a central treatment plant from which a treated effluent is discharged. Some authors have discussed reasons why the prevailing paradigm of wastewater management must be questioned, considering that change involves great efforts, consumes resources, takes time and thus must be clearly justified (Daigger, 2009; Beck, 2013). However, population growth and an increasing standard of living are pushing the use of resources like water and nutrients beyond sustainable boundaries (Daigger, 2009; Steffen et al., 2015). Moreover, natural resources are typically used in a linear fashion, from extraction, to consumption and final disposal (“take, make, waste”), which is at the root of their current unsustainable use. Rapid urbanization is accentuating this phenomenon, leading in the water sector to enormous pressures on water resource, disruptions in geochemical flows of agriculturally crucial nutrients, their potentially harmful dispersion into aquatic ecosystems, and the financial instability of water utilities (Daigger, 2009).

Recently, increasing attention has been paid to a paradigm shift or transition towards resource recovery and circularity in the water sector in countries like the Netherlands (Ampe et al., 2020), Australia (Brown et al., 2013; Fam et al., 2014), Switzerland (Boller, 2013) and Sweden (McConville et al., 2017b; Skambraks et al., 2017). For developing nations with fast-growing urban conglomerates, under-serviced areas and informal settlements it will be very challenging to adopt the toilet-to-sewer-to-treatment model because building the infrastructure is too time-consuming and costly (Larsen et al., 2016; Öberg et al., 2020). The capacity of conventional systems to meet the challenges of the 21st century, like climate change and resources scarcity, are also questioned because of their inherent inflexibility (Brands, 2014; Larsen et al., 2016; Öberg et al., 2020). In addition, the centralised approach to wastewater collection and treatment generates sludge of lower quality due to the pollutants found in mixed wastewater, loses valuable
nutrients and constrains opportunities to explore water reuse on a fit-for-purpose basis at the local scale (Sharma et al., 2013).

The introduction of the Roadmap to a Resource Efficient Europe (2011) and commitment to The Action Plan towards the Circular Economy (2015) have shown the EU’s ambition to move towards a resource-efficient economy. This refers to systems that are competitive and inclusive while respecting planetary boundaries constraints (Steffen et al., 2015) through, for example, the reconsideration of waste as a resource (Domenech and Bahn-Walkowiak, 2019).

Alternative solutions for wastewater, such as source separation, can facilitate the shift in paradigm towards greater resource recovery. However, implementation of pilot projects in small communities and urban areas has been difficult, especially in comparison to conventional centralised solutions (McConville et al., 2017). There are also uncertainties about the effects of source separation systems, as they can be both advantageous and disadvantageous depending on the impact categories and system boundaries (Tidåker et al., 2007; Spångberg et al., 2014; Lehtoranta et al., 2022). Generally, the main challenges associated with implementation of source-separating systems have been related to the investment costs, the legal and institutional uncertainties and lack of capacity and organisation. In a study of blocking mechanisms hindering the expansion of source-separating systems, McConville et al. (2017) found that a major barrier was the weak interchange between knowledge development and entrepreneurial activity. In addition, limited knowledge (about blackwater or urine diversion options, for example) promotes perceptions among decision-makers of source-separating systems as immature and risky. The mixed sewerage regime in wastewater jurisdictions is strongly entrenched, which makes it challenging to offer a competitive alternative to the regime standard (McConville et al., 2017). The cited authors also provided several practical recommendations to challenge the entrenched views. These included technical research and development, establishment of national networks, communication platforms, standards and norms for reuse and operational parameters, clarification of policies and interpretations of regulations, and development of more refined market analysis and business models.

In addition to addressing barriers and blocking mechanisms, it is also important to recognize and promote key drivers of change, which are required for any transition of a socio-technical system and may be very context-sensitive. The main driver for change in the sanitation sector in high income countries is the ageing infrastructure and need for replacement. Scarcity of resources, such as water, energy and phosphorus, pollution and social environmental awareness are also recognized as potential drivers for innovation and regulation (Brands, 2014). In Australia, for example, identified drivers for adoption of decentralised wastewater systems include needs to overcome limitations of local wastewater service infrastructure, upgrade infrastructure, protect the environment, showcase sustainability, promote water conservation, improve landscape amenity, and promote innovative technology (Sharma et al., 2013; Fam et al., 2014). In addition,
Skambraks et al. (2017) found that local environmental goals and ambitions to gain knowledge about new technologies are important for the implementation of pilot systems for source separation in urban areas. They also found that transfer of knowledge and experience from pilot systems is crucial for planning systems for larger areas.

Applying a systemic perspective together with management and transition frameworks, for example, can help efforts to understand complex multilevel governance structures and the interactions characterizing resource governance regimes (Pahl-Wostl et al., 2010). More bridges between research, policy and practice are needed to overcome the technical and socio-economic challenges of source separation systems, such as toilet design, blockages, social acceptance, incentivisation, risks and market creation (McConville et al., 2016). Furthermore, better estimations and reliable data, e.g., of emissions from urine or blackwater treatment and management, are needed to assess their overall sustainability.

2.6. Knowledge gaps and links to the objectives

Knowledge gaps identified in the previous sections can be summarized and linked to the research objectives (Section 1.1) as needs to obtain more information on:

- Flow-proportional sampling has rarely been applied in field investigations of on-site wastewater treatment systems compared to grab sampling. The choice of sampling technique could affect the representativeness of samples, so there is a need to assess the relative merits and disadvantages of these sampling approaches (Objective 1).
- Phosphorus treatment in on-site sanitation systems like sand filters varies widely and more research is needed to understand the treatment mechanisms in full-scale systems. Nitrogen and indicator bacteria discharges from on-site systems have received less research attention than phosphorus treatment, despite their relevance for eutrophication and human health effects (Objective 1).
- Effects of temperature in full-scale on-site wastewater treatment systems (like package plants) has rarely been investigated in cold regions with long winters with snow cover and snow melt periods that can potentially affect the systems’ treatment performance (Objective 1).
- Discharges of micropollutants, such as pharmaceuticals, from on-site wastewater treatment systems and their impacts on receiving environments have received limited attention, and phthalate esters have not been previously studied at on-site scale. Thus, there are needs to quantify micropollutants because of their ubiquity in receiving waters and potential effects on human health and biota (Objective 1).
- Combinations of multiple types of data, such as qualitative and quantitative indicators of sanitation systems’ performance have not often been applied in systems analysis. However, there is a clear need to integrate indications of widely varying nature, units and scales, such as social acceptance (low-high) and treatment efficiency (%), in comprehensive sustainability assessments (Objective 2).
• Implications of the addition of weights to performance indicators in sustainability assessments of sanitation systems need more attention, as they are applied during decision-making processes and influence selections in practice (Objective 2).

• Better understanding of decision-making processes when planning sanitation systems for new-build or existing areas, and the blocking mechanisms, barriers and drivers for the implementation of alternative, innovative systems, is needed to enhance recovery of resources and ability to cope with new challenges in the water sector (Objective 3).

• More knowledge is needed of the historical development of alternative sanitation systems enabling resource recovery, like blackwater/greywater separation and urine diversion, particularly in relation to the dominance of conventional centralised wastewater treatment in Sweden. This is important for understanding and promoting the development and implementation of alternative systems (Objective 3).
3. Research methods

The research underlying this thesis involved a wide range of methods. Field sampling campaigns of various on-site wastewater treatment systems were carried out in Papers I-III. Multi-criteria analysis methodology, including life cycle analysis (LCA) and combinations of mass-balance calculations and qualitative estimations were used to investigate on-site sanitation alternatives from a sustainability perspective using performance indicators (Paper IV). Lastly, qualitative research methods, including in-depth interviews (with follow-up questionnaires) and a historical literature review were the methodological approaches applied in Papers V and VI.

3.1. Field investigation of on-site wastewater treatment facilities (Papers I-III)

3.1.1. Selection of facilities

Seven Swedish municipalities were contacted to identify potential facilities with sand and phosphorus filters (Papers I and II) and package plants (Paper III) for the studies underlying this thesis. Private and municipal operators were contacted by telephone to ask for permission to visit and inspect the potential facilities. About 45 facilities were inspected to determine their suitability for sampling. The inspections consisted of checking the inlets and outlets of the systems for accessibility and to ensure they were designed in a way that would enable appropriate sampling, e.g., flow proportional sampling of the effluents of sand and phosphorus filters. In total, 19 facilities in Sweden (and four in Finland) were investigated and included in Papers I-III. Sixteen were designed to treat both carbonaceous material (BOD) and phosphorus to acceptable levels, as required by the Swedish guidelines (SwAM, 2016a), and are presented in this thesis (Table 1). Five of these facilities (D-H) were soil-based systems and 11 facilities (I-TF) were package plants (Table 1).

Facilities D and E included sand filters constructed according to the Swedish standards, consisting of 80 cm thick filter media (sand and gravel, particle sizes 2–8 mm) with a design load of 30–60 L m$^{-2}$ d$^{-1}$ and typical surface area of 25 m$^2$, as described by Palm et al., (2012). Slotted pipes distributed the wastewater, which subsequently percolated vertically through the filter bed to the bottom of the system for collection in drainage pipes. Sand filters in facilities F, G and H (Figure 2) had a layer of drainage baskets on the top (biomodules) with a 0.5 m wide triangular cross section, where the distribution pipe was placed. All sand filters were covered with an approximately 30 cm thick layer of soil.
Table 1 Investigated on-site wastewater treatment facilities. The facilities’ names correspond to the names used in the papers that originally reported the research, to facilitate cross-checking. Facilities J and K (*) served the same 14 users in total and the wastewater was pumped from the septic tank to a distribution chamber where the flow was diverted into the two separate treatment facilities. Even flow distribution was assumed, thus each plant served seven users.

<table>
<thead>
<tr>
<th>Facility</th>
<th>Type of treatment</th>
<th>Phosphorus removal mechanism</th>
<th>Years in use</th>
<th>No. of users</th>
<th>Reported in Papers</th>
</tr>
</thead>
<tbody>
<tr>
<td>D</td>
<td>Sand filter, Swedish standard</td>
<td>Alkaline filter (Polonite®)</td>
<td>1-2</td>
<td>2</td>
<td>I and II</td>
</tr>
<tr>
<td>E</td>
<td>Sand filter, Swedish standard</td>
<td>Alkaline filter (Filtra P)</td>
<td>6-7</td>
<td>5</td>
<td>I and II</td>
</tr>
<tr>
<td>F</td>
<td>Sand filter, with biomodules</td>
<td>Alkaline filter (Polonite®)</td>
<td>1</td>
<td>2</td>
<td>I and II</td>
</tr>
<tr>
<td>G</td>
<td>Sand filter, with biomodules</td>
<td>Alkaline filter (Polonite®)</td>
<td>1</td>
<td>2</td>
<td>I and II</td>
</tr>
<tr>
<td>H</td>
<td>Sand filter, with biomodules</td>
<td>Alkaline filter (Polonite®)</td>
<td>&lt;1</td>
<td>2</td>
<td>I and II</td>
</tr>
<tr>
<td>I</td>
<td>Package plant with biological fibre material in a tank</td>
<td>Alkaline filter (Polonite®)</td>
<td>2</td>
<td>2</td>
<td>I and II</td>
</tr>
<tr>
<td>J</td>
<td>Package plant with Biop®, biofilm treatment without aeration</td>
<td>Alkaline filter (Polonite®)</td>
<td>&lt;1</td>
<td>7*</td>
<td>I and II</td>
</tr>
<tr>
<td>K</td>
<td>Package plant with Biop®, biofilm treatment without aeration</td>
<td>Alkaline filter (Polonite®)</td>
<td>&lt;1</td>
<td>7*</td>
<td>I and II</td>
</tr>
<tr>
<td>L</td>
<td>Package plant with activated sludge</td>
<td>Alkaline filter (Polonite®)</td>
<td>&lt;1</td>
<td>2</td>
<td>I and II</td>
</tr>
<tr>
<td>ASC</td>
<td>Package plant with activated sludge</td>
<td>Chemical treatment</td>
<td>5</td>
<td>32</td>
<td>III</td>
</tr>
<tr>
<td>ASF1</td>
<td>Package plant with activated sludge</td>
<td>Alkaline filters (x3) (Polonite®)</td>
<td>2</td>
<td>20-30</td>
<td>III</td>
</tr>
<tr>
<td>ASF2</td>
<td>Package plant with activated sludge</td>
<td>Alkaline filter (Polonite®)</td>
<td>2</td>
<td>2</td>
<td>III</td>
</tr>
<tr>
<td>ASF3</td>
<td>Package plant with activated sludge</td>
<td>Alkaline filter (Polonite®)</td>
<td>2</td>
<td>3</td>
<td>III</td>
</tr>
<tr>
<td>SBR1</td>
<td>Package plant (batch reactor)</td>
<td>Chemical treatment</td>
<td>4</td>
<td>4</td>
<td>III</td>
</tr>
<tr>
<td>SBR2</td>
<td>Package plant (batch reactor)</td>
<td>Chemical treatment</td>
<td>1</td>
<td>4</td>
<td>III</td>
</tr>
<tr>
<td>TF</td>
<td>Package plant with a trickling filter and final polishing filter</td>
<td>Chemical treatment</td>
<td>8</td>
<td>12-14</td>
<td>III</td>
</tr>
</tbody>
</table>

1 Years in use refer to the approximated number of years the treatment units had been in use at the time of sampling.
The alkaline P-filters consisted of bags filled with P-sorbing material (Figure 3, top). Polonite® filter material (Supplier: Ecofiltration AB, Sweden) was used in 11 of the facilities, placed in a container or in a package plant at ground level and operated in down-flow (D) or upflow (F–L, ASF1-ASF3) mode. Filtra P alkaline P-filter material (supplier: Wavin-Labko Ltd) was used in facility E, placed in two tanks in series installed at different levels, one percolating in down-flow mode and one in up-flow mode.

Figure 3 Examples of investigated package plants: two with an alkaline P-filter in the middle (facility J, top left, and module used in facilities L, ASF2 and ASF3, top right) and
two with chemical removal of phosphorus (facilities ASC, bottom left, and TF, bottom right).

Facility I was a package plant consisting of a trickling filter made of fibre material installed in a tank. Facilities J and K (Figure 3, top left) were package plants, each consisting of a unit providing multi-stage biological treatment based on attached growth with no aeration. Facilities L, ASF2 and ASF3 (Figure 3, top right) consisted of an aerated activated sludge system installed in a unit, and facility ASF1 had a similar design, but with a series of three units. Facility ASC (Figure 3, bottom left) consisted of a concrete tank with activated sludge, a chemical precipitation unit and a UV disinfection unit. Facility TF (Figure 3, bottom right) was a trickling filter with plastic discs media of about 10 cm in diameter, and a subsequent unit where chemical precipitation, sedimentation and final filtration through a filter of mineral wool occurred. SBR1 and SBR2 (Figure 4) were sequencing batch reactors (SBRs) with a collection tank and a process tank where the wastewater was treated in a series of batch phases, including aeration, addition of aluminium-based salts for phosphorus precipitation and settling, for approximately seven and a half hours.

![Figure 4](image)

**Figure 4** Examples of package plants with sequencing batch reactors like those in facilities SBR1 and SBR2, with the collection tank to the left and process tank to the right. The control panel and container for the P chemical removal are located in the middle part of the package plant.

All investigated systems included either a separate or built-in septic tank for sedimentation of coarse particles before the biological treatment.
3.1.2. Sampling methods

Effluent samples for Papers I and II were collected flow-proportionally between September and October 2015 and between May and August 2016. Each facility (D-L) was sampled on at least three occasions, for approximately 3-4 hours per occasion. Composite samples of about 2-3 L, depending on the flow, were taken at different times of the day depending on the volumes and practicalities regarding equipment and location.

Effluents from the facilities investigated in Paper III were grab sampled from an outlet pipe, sampling chamber, or the last chamber containing the treated wastewater before discharge, between August 2019 to April 2021 during 7-10 sampling occasions.

The influent wastewater from all facilities was grab sampled in the third chamber of the septic tank, the distribution/pumping box or the sedimentation tank before the biological process unit.

3.1.3. Water sample analyses

On each sampling occasion the following variables were measured on-site: the wastewater temperature; total suspended solids (TSS) contents, according to the European Committee for Standardization (2005); and pH (with WTW pH330 pH meter and WTW SenTix41 pH electrode). On some occasions the turbidity of the samples were also measured (using a HACH 2100Q instrument for ratio turbidimetric determination). In addition, the water temperature was continuously measured in the process tank of three package plants (ASF2, ASF3 and SBR1) with HOBO® Pendant®MX Temp (MX2201) loggers. Data from local weather stations (SMHI, n.d.) were used for the air temperature analyses.

Concentrations of BOD, phosphorus (total and dissolved: tot-P and dis-P, respectively), and the indicator microorganisms *Escherichia coli* and *Enterococcus* were measured in the influent and effluent water samples. Additionally, the indicator organisms total coliforms and *Clostridium perfringens* were analysed in the samples from the facilities investigated in Papers I and II. Nitrogen compounds (tot-N, NO₂-N, NO₃-N, NH₄⁺-N) were analysed in the wastewater samples from the package plants investigated in Paper III (and chloride in some of these samples).

BOD₇ (the amount of oxygen needed for microorganisms to degrade organic material in water within seven days) was analysed according to the European standard method CSN EN 1899–1 (European Committee for Standardization, 1998). Phosphorus and nitrogen compounds were analysed by the molybdate method using a QuAAtro segmented flow analyser, with persulfate oxidation digestion according to SS-EN 1189:1996 performance 6.4. The device-specific method numbers for total and dissolved P, total N, NO₃-N and NO₂-N (without reduction column) and NH₄-N contents were A-031-04, Q-003-04 and Q-001-04, respectively. NH₄-N was analysed following ISO 11732 and DIN 38 406 (Part 23, section 2) standard methods. Portions of samples used for analyses of BOD and
nutrients were stored frozen until analysis, and portions used to determine densities of
the bacteria were stored at 5°C and examined in an accredited laboratory within 24 hours.

In Papers I and II, the indicator bacteria *E. coli* and total coliforms were analysed following
Swedish standard methods SS 028167-2, whereas for intestinal enterococci and *C.
perfringens* the SS-EN ISO 7899-2 and ISO/CD 14189/6461-2 methods were used. The
results were expressed in terms of colony forming units (CFUs) per 100 ml. In Paper III,
*E. coli* and enterococci were analysed using the SS EN-ISO 9308-2:2014 and IDEXX
Enterolert® methods, respectively, and the results were expressed in terms of most
probable number (MPN) per 100 ml.

### 3.1.4. Micropollutant analyses

Samples of the influent and effluent of two facilities (ASC and TF) and blank samples of
the sampling equipment (obtained using tap water) were collected on three occasions
(March, June and August 2022) for micropollutant analyses, using a stainless-steel sampler,
then stored in glass jars until analysis. The investigated micropollutants included 19
pharmaceuticals (Table 2), an artificial sweetener (acesulfame K), caffeine and 15
phthalate compounds: bis(2-ethylhexyl) phthalate (DEHP), bis(4-methyl-2-pentyl)
phthalate (BMPP), benzyl butyl phthalate (BBP), bis(2-ethylhexyl) terephthalate
(DEHT), dibutyl phthalate (DBP), diethyl phthalate (DEP), di-n-hexyl phthalate (DHP),
diisobutyl phthalate (DIBP), diisononyl phthalate (DINP), diisopentyl phthalate (DISP),
dimethyl phthalate (DMP), diocetyl phthalate (DNOP), dipentyl phthalate (DPP),
dicyclohexyl phthalate (DCHP) and hexyl-2-ethylhexyl phthalate (HEHP).

The samples were subjected to liquid-liquid extraction (for phthalates) or solid-phase
extraction (for pharmaceuticals, the sweetener and caffeine) with no pre-filtration.
Directly after extraction the samples were concentrated and analysed with liquid
chromatography-tandem mass spectrometry (LC-MS/MS) for pharmaceuticals, the
sweetener and caffeine and with gas chromatograph-tandem mass spectrometry (GC-
MS/MS) for phthalates. Method and field blanks were used to evaluate background levels
of target phthalate compounds, and spiked control samples, with at least one procedural
blank per sample set, and internal standards were used for quality control. More detailed
information can be found in Paper III and the Supplementary Material appended to the
article.
Table 2 Analysed pharmaceuticals in the influent and effluent of two package plants (ASC and TF).

<table>
<thead>
<tr>
<th>Compound</th>
<th>Characterization/usage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diclofenac</td>
<td>Analgesic and anti-inflammatory</td>
</tr>
<tr>
<td>Ibuprofen</td>
<td>Analgesic and anti-inflammatory</td>
</tr>
<tr>
<td>Bisoprolol</td>
<td>β-blocking agents</td>
</tr>
<tr>
<td>Candesartan</td>
<td>ACE inhibitor</td>
</tr>
<tr>
<td>Clarithromycin</td>
<td>Antibiotic</td>
</tr>
<tr>
<td>Enalapril</td>
<td>ACE inhibitor</td>
</tr>
<tr>
<td>Eprosartan</td>
<td>ACE inhibitor</td>
</tr>
<tr>
<td>Fenbendazole</td>
<td>Antiparasitic drug</td>
</tr>
<tr>
<td>Fluconazole</td>
<td>Antifungal medication</td>
</tr>
<tr>
<td>Gabapentin</td>
<td>Anticonvulsant</td>
</tr>
<tr>
<td>Ketoprofen</td>
<td>Analgesic and anti-inflammatory</td>
</tr>
<tr>
<td>Levetiracetam</td>
<td>Anticonvulsant</td>
</tr>
<tr>
<td>Metoprolol</td>
<td>β-blocking agents</td>
</tr>
<tr>
<td>Primidone</td>
<td>Anticonvulsant</td>
</tr>
<tr>
<td>Ramipril</td>
<td>ACE inhibitor</td>
</tr>
<tr>
<td>Sertraline</td>
<td>Antidepressant</td>
</tr>
<tr>
<td>Venlafaxine</td>
<td>Antidepressant</td>
</tr>
<tr>
<td>Warfarin</td>
<td>Anticoagulant</td>
</tr>
<tr>
<td>Xylometazoline</td>
<td>Nasal decongestant</td>
</tr>
</tbody>
</table>

3.1.5. Data analysis

Percentage removal efficiencies were calculated using the median influent and effluent values for each facility, or influent and effluent values on each sampling occasion to relate the removal efficiency to seasonally varying parameters such as temperature.

The overall removal of each compound was based on the influent concentrations in samples taken from the last chamber of the septic tanks or influent pipe into the biological treatment units, so any removal occurring in the pre-sedimentation stage was not considered.

Parameters measured in the samples taken flow proportionally (Papers I and II) were weighted in relation to the water flow, with flow-proportional arithmetic means ($\bar{x}^*$) and standard deviations ($s$) calculated according to Eq. 1 and 2, respectively.

\[
\bar{x}^* = \frac{\sum_{i=1}^{n} x_i w_i}{\sum_{i=1}^{n} w_i} \quad (\text{Eq. 1})
\]

\[
s = \sqrt{\frac{\sum_{i=1}^{n} w_i (x_i - \bar{x}^*)^2}{\sum_{i=1}^{n} w_i - 1}} \quad (\text{Eq. 2})
\]
Here:
\[ \bar{x}^* = \text{flow-proportional arithmetic mean} \]
\[ n = \text{number of observations} \]
\[ x_i = \text{measured concentration} \]
\[ w_i = \text{volume of wastewater of the composite sample} \]
\[ s = \text{flow-proportional standard deviation} \]

Similarly, bacterial densities in flow-proportional samples were flow-weighted according to Eq. 3.

\[
\bar{x}^*_g = \left( \prod_{i=1}^{n} x_i^w \right) \frac{1}{\sum_{i=1}^{n} w_i} \quad \text{(Eq. 3)}
\]

Here:
\[ \bar{x}^*_g = \text{flow-weighted geometric mean} \]
\[ n = \text{number of observations} \]
\[ w_i = \text{volume of wastewater of the composite sample} \]

Nitrification rates were estimated by subtracting NH\text{4}^+-N\text{eualuent} from NH\text{4}^+-N\text{influent} then dividing by tot-N \times 100, and denitrification rates by subtracting NH\text{4}^+-N\text{eualuent}, NO\text{2-3}^--N\text{influent} and NO\text{2-3}^--N\text{eualuent} from NH\text{4}^+-N\text{influent} then dividing by tot-N \times 100.

Bacteriological data were log\text{10}-transformed for statistical analyses and half of the lower detection limit was used for left-censored data, and upper detection limits for right-censored data for estimations of removal rates. The facilities were grouped as soil-based systems (those with sand filters and alkaline P-filters) and package plants to compare the effluents and test for significant differences. The significance of differences between influent and effluent concentrations was assessed using the Wilcoxon signed rank test for non-parametric data and correlations between parameters using Spearman rank correlation analysis or Pearson correlation analysis for transformed data, e.g., bacteria densities (with \( \alpha = 0.05 \) threshold significance level). Mood’s median non-parametric test was used to compare the effluent concentrations between the two groups (soil-based systems and package plants). Minitab® 20 statistical software was used for most of the data analysis. Bacteriological left-censored data were analysed using the Peto-Peto nonparametric test for censored data, as implemented in the NADA package (Lopaka Lee, 2020) using R (R Core Team, 2021) and RStudio (RStudio Team, 2022). A distribution probability analysis was also conducted in R to compare the indicator bacteria densities in effluents of package plants and soil-based systems. Bacteriological data presented in Papers I–III are reported in different units, CFU and MPN, which were considered equivalent for the purpose of comparing the effluent concentrations. However, differences in the methodology and precision of the results (Gronewold and Wolpert, 2008) are acknowledged and the results were interpreted with caution.
3.2. System analysis (Paper IV)

3.2.1. Sustainability indicators

The selection of sustainability indicators was based on previous research proposing sustainability indicators to assess wastewater systems (Balkema et al., 2002; Hellström et al., 2000; Lennartsson et al., 2009; Molinos-Senante et al., 2014; Schoen et al., 2017). Sustainability criteria were organized into five main categories: environmental, economic, socio-cultural, technical and health-related. For each criterion, a number of assessable indicators (qualitative and quantitative) were defined (Table 3).

Table 3 Summary of the sustainability indicators included in Paper IV.

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nutrient removal (%)</td>
<td>Capacity of the system to remove nitrogen (N) and phosphorus (P) from the influent wastewater. Two sub-indicators (removal of N and P) were quantified based on previous studies (Palm et al., 2002; Lennartsson et al., 2009; Olshammar et al., 2015; Vidal et al., 2018) and considered equally important. Thus, each sub-indicator was given equal weight.</td>
</tr>
<tr>
<td>Potential for nutrient recycling (%)</td>
<td>Potential agricultural reuse of the waste fractions produced in the systems in relation to the nutrient content (N and P). The sludge from the septic tanks, sand from the sand filters, Polonite® filter material from the alkaline P-filters, blackwater and urine were the fractions considered potential sources of nutrients. This indicator was quantified based on the current practice in Sweden when data were available, e.g., about 34% of the generated sludge is reused as a soil conditioner (Statistics Sweden, 2018), or based on assumptions when data were not available, e.g., 100% reuse of the blackwater and urine, 5% of Polonite® filter material, and 0% of the sand from the sand filters.</td>
</tr>
<tr>
<td>Global warming potential GWP (Kg CO₂-eq. year⁻¹)</td>
<td>Greenhouse gas (GHG) emissions in kg CO₂-equivalents (eq.) released during production of the considered systems' components and materials (e.g., tanks, pipes, filter materials) and their transport (e.g., from the production site), system installation and operation (e.g., electricity consumption) and maintenance (e.g., sludge collection, chemicals and alkaline P-filter refill/replacement), as well as the post-treatment of the fractions that were not treated on-site (indirect nitrous oxide emissions from ammonia emissions during storage of sludge, blackwater and urine). The end-of-life phase was excluded from the analysis. The calculations were based on LCA methodology standards (ISO 14040, 2006) following the global warming potential impact assessment method [v1.0.1, January 2015] with use of the ELCD 3.2 database.</td>
</tr>
<tr>
<td>Cumulative energy demand CED (MJ year⁻¹)</td>
<td>Primary energy used during the production, transport and installation of the components and materials, during operation and maintenance of the sanitation alternatives and during post-treatment of the fractions that were not treated on-site (sludge, blackwater and urine). The end-of-life phase was excluded from the analysis. The calculations were based on the same LCA methodology standards as described for GWP.</td>
</tr>
<tr>
<td>Energy recovery (H-M-L)</td>
<td>The possibility to obtain energy in the form of biogas produced from the collected septic sludge. This indicator was evaluated qualitatively with a three-point ordinal scale that classified the energy recovery of the alternatives as low (L), medium (M) or high (H), and was estimated proportionally to the volume.</td>
</tr>
</tbody>
</table>
of sludge produced based on the composition of the wastewater fractions according to Jönsson et al. (2005).

**Capital cost (€ year⁻¹)**

Cost of the investment to purchase the components and manpower required for the installation of each OSS alternative multiplied by the annuity factor (amortization time and 4% interest rate).

**Operation and maintenance (O&M) cost (€ year⁻¹)**

Yearly cost for the operation and maintenance of each option, including: the collection and transport of blackwater, urine and the sludge from the septic tank; electricity use; purchase of consumables (chemicals, alkaline P-filter) and components (change of pump), and the annual maintenance service including effluent sampling and analysis.

**Social acceptance (VL-L-M-H-VH)**

User-friendliness of the alternatives regarding the convenience, effort and degree of complexity of operating the system, from the user's perspective. Assessed qualitatively on a five-point ordinal scale from very low (VL) to very high (VH) acceptance. A sand filter (A1) and drain field (A2) system was used as a reference because it is the most commonly installed type of OSS in Sweden, and thus has high social acceptance (Olshammar et al., 2015), users are familiar with them and they are considered convenient due to their simplicity (Lennartsson et al., 2009). The other alternatives were assessed in relation to A1 and A2 in terms of the increase in “inconvenience” for the users arising from the addition of specified components to the OSS.

**Robustness (L-M-H)**

Defined based on two sub-indicators, the risk of failure of the system and the adaptability to flow fluctuations, assessed qualitatively on a three-point scale from low to high. The risk of failure accounted for the possibility of a technical problem hindering the system’s treatment capacity, and the likelihood of such an incident. The adaptability to flow fluctuations accounted for the adaptability to changes in the quantity and quality of the flow, e.g., the presence or absence of users resulting in changes in average water consumption. The risk of failure had higher importance (2/3) than the adaptability to flow fluctuations (1/3) because of the more severe implications.

**Risk of pathogen discharge (VL-L-M-H-VH)**

The capacity of the OSS to remove pathogens from wastewater prior to discharge in the environment. Assessed on a five-point ordinal scale from very low to very high risk, based on the number of barriers included in the systems with potential capacity to remove pathogens and thus decrease the pathogen load (e.g., filter media, chemical precipitants). The type of receiving waters e.g., groundwater/surface waters was also considered.

For more detailed descriptions of the sustainability indicators the reader is referred to Paper IV included in this thesis and the Supplementary Material appended to the article.

### 3.2.2. On-site sanitation alternatives

Nine on-site sanitation alternatives (Table 4) were selected based on relevant literature and discussions with practitioners, including commonly used conventional systems such as sand filters and drain fields, as well as package plants and source separation systems.
### Table 4 Summary of the nine on-site sanitation alternatives investigated in Paper IV.

<table>
<thead>
<tr>
<th>No.</th>
<th>Alternative</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>A1</td>
<td>Septic tank and sand filter</td>
<td>Wastewater was collected in a three-chamber septic tank (2.2 m³) made of fibreglass-reinforced polyester (FRP) and pumped to a sand filter constructed below the ground surface. The wastewater was spread using perforated distribution pipes and treated by various physical, chemical and biochemical reactions and processes during filtration. The effluent was collected at the bottom of the filter by drainage pipes. A conventional sand filter constructed according to the Swedish standards was assumed (Swedish EPA, 2003a), with a 0.8 m deep layer of filter media (sand/gravel, 0 to 8 mm in diameter) and a surface area of 30 m². Both distribution and drainage pipes were embedded in a 0.2 m coarse gravel layer. The sludge from the septic tank was transported to the nearest WWTP for anaerobic treatment (in all alternatives with septic sludge).</td>
</tr>
<tr>
<td>A2</td>
<td>Septic tank and drain field</td>
<td>A similar design to A1 was assumed. The wastewater continued infiltrating and percolating through the underlying soil instead of being collected after passage through the filter material, a 35 cm layer of coarse gravel, 12-32 mm in diameter (Swedish EPA, 2003b). The properties of the soil were assumed to allow infiltration of the wastewater.</td>
</tr>
<tr>
<td>A3</td>
<td>Septic tank, sand filter and P-filter</td>
<td>Same design as in alternative A1 with the addition of an alkaline P-filter (Polonite®) for phosphorus removal after the sand filter (with wastewater flowing by gravity between the two sub-units).</td>
</tr>
<tr>
<td>A4</td>
<td>Septic tank with chemical P removal and sand filter</td>
<td>Wastewater was collected in a three-chamber septic tank. A chemical precipitation unit for dosing with polyaluminium chloride was installed under sink in the kitchen of the connected house. Flocculation and sedimentation occurred in the septic tank, which had a larger volume (4 m³) than in alternatives A1-3. Septic sludge was assumed to contain more P than in A1-3. The lifespan of the sand filter was assumed to be longer (25 years instead of 20 years as in A1-3), because some of the suspended solids and BOD were removed during the flocculation process, resulting in a lower load in the sand filter (Palm et al., 2002; Weiss et al., 2008).</td>
</tr>
<tr>
<td>A5</td>
<td>Septic tank with P precipitation and drain field</td>
<td>Same design as in A2 with addition of a chemical precipitation unit in the connected house, a larger septic tank (4 m³) and longer lifespan of the sand filter (25 years instead of 20 years, as in A1-3), as described for A4.</td>
</tr>
<tr>
<td>S1</td>
<td>Greywater and blackwater separation</td>
<td>Separate collection of greywater and blackwater. Greywater was collected in a septic tank then passed through a sand filter. The sand filter was assumed to have a longer lifespan than a conventional filter (25 years instead of 20), because most of the nutrients and BOD were already removed with the blackwater. Sludge from the greywater tank was transported to a WWTP. Blackwater was collected in a holding tank, which was emptied once a year. A low-flush toilet (about 1 L per flush) was used, because smaller volumes are desirable for the collection, storage and treatment. Collected blackwater is transported to a central treatment facility using treatment with 1% urea for hygienization.</td>
</tr>
<tr>
<td></td>
<td>Process Description</td>
<td></td>
</tr>
<tr>
<td>---</td>
<td>---------------------</td>
<td></td>
</tr>
<tr>
<td>S2</td>
<td>Urine diversion</td>
<td>Separate collection of greywater, urine and faeces fractions, using urine-diverting toilets, with joint collection of greywater and faeces, which were conveyed to a sand filter. The sludge from the septic tank was transported to a WWTP. The urine was collected in a container and transported to a centralised facility for hygienization (six-months storage).</td>
</tr>
<tr>
<td>P1</td>
<td>Package plant with activated sludge and alkaline P filter</td>
<td>This package plant consisted of a single unit buried underground to which the wastewater from a household was transported by gravity. The mechanical treatment occurred in the first chamber. The next chamber had aerators that provided oxygen to the water. The final treatment step consisted of an alkaline P-filter with Polonite® material placed in a bag in the centre of the treatment unit. Settled sludge was collected from the chambers for mechanical and biological treatment and transported to the nearest WWTP for anaerobic treatment.</td>
</tr>
<tr>
<td>P2</td>
<td>Package plant with activated sludge and chemical coagulation for P removal</td>
<td>This package plant consisted of a single unit buried underground, to which the wastewater from a household was transported by gravity. The plant operated in a two-phase semi-continuous regime. The main phase corresponded to a continuous activated sludge process, and the second to the discharge of excess sludge and cleaning of the filter when the level of wastewater in the equalization tank was low. The raw wastewater was first collected in an equalization tank then pumped to an aerated water-processing tank with activated sludge and chemical precipitation for P removal. A small sand filter was used as a final polishing step. All sludge was collected in a separate tank inside the unit, then transported to the nearest WWTP for anaerobic treatment.</td>
</tr>
</tbody>
</table>
3.2.3. The ELECTRE III method

The ELECTRE III method was selected to compare the alternatives for on-site sanitation with aid of the schematic workflow proposed by Rowley et al. (2012) for sustainability analyses. A flowchart summarizing the ELECTRE III methodology is shown in Figure 5.

![Flowchart of ELECTRE III methodology](image)

**Figure 5** Summary of the ELECTRE III workflow. The steps shown in grey scale (Veto and Generation of discordance indices) were not included in this study.

The alternatives were assessed using indicators in an evaluation matrix where the best outcome was represented by the maximum evaluation of each indicator. In ELECTRE III, the construction of an outranking relation between two alternatives \( a \) and \( b \) \((a S b)\) is based on two major concepts, the concordance (majority principle) and discordance (respect of minority principle), which are used to calculate a credibility index that corresponds to a value of “outranking” of \( a \) with regard to \( b \) (Figueira et al., 2005).

To calculate the concordance index \( C(a, b) \), the alternatives were evaluated against each indicator pairwise (Eq. 4), multiplying the partial concordance values \( c_i(a, b) \) obtained when comparing alternative \( a \) to \( b \) by the weights allocated by a reference group (described in section 3.2.4). The larger the \( C(a, b) \), the stronger the evidence that
alternative a is preferred over b (Vincke, 1992). Preference (p_i) and indifference (q_i) thresholds were defined, as a constant number or as a percentage of the performance value g_i (a), for each indicator and used to calculate the concordance values c_i (a, b) (Eq. 5).

\[ C_i(a,b) = \frac{\sum_{c=1}^{n} w_c C_i(a,b)}{\sum_{c=1}^{n} w_c} \]  

(Eq. 4)

where:

\[ c_i(a,b) = \begin{cases} 
\text{if } g_i(a) + q_i \geq g_i(b) \\
0 \text{ if } g_i(a) + p_i \leq g_i(b) \\
\frac{p_i + g_i(a) - g_i(b)}{p_i - q_i} \text{ otherwise}
\end{cases} \]  

(Eq. 5)

and:

\( c_i(a,b) \) is the concordance value for indicator i:

\( g_i(a) \) is an individual evaluation of the alternative a for indicator i:

\( q_i \) is the indifference threshold which determines if one alternative is “weakly preferred” over another alternative with respect to a given indicator i although its evaluation may be (slightly) lesser in value, and

\( p_i \) is the preference threshold, which determines if one alternative is “strongly preferred” over another in terms of indicator i.

Calculation of the discordance index D_i(a,b) requires the definition of veto thresholds (v_i), which express the possibility of alternative a being discredited if it is exceeded by the performance of b by an amount greater than the veto threshold, regardless of the other indicators. The discordance index D_i(a,b) was zero for all pairs of alternatives since no veto threshold (v_i) was applied in this study.

An index called the degree of credibility of the statement a S b (\( \delta(a,b) \)), which indicates the extent to which a outranks b, was calculated by aggregating the discordance and concordance indexes. Since the discordance index was zero in this study, the credibility of the outranking relation was equal to the concordance index C(a,b).

The rankings of the alternatives were determined by two preliminary rankings based on the values of \( \delta(a,b) \), namely descending and ascending preorders or distillations. In descending distillation, alternatives are ranked from the best to “less good” alternatives, whereas in ascending distillation they are ranked from “least bad” to worst. These distillation processes show the number of alternatives that are outranked by S, and the
number of actions that Si outranks, considering a threshold s (λ), as described in detail by Roy (1978). A final ranking is the result of the intersection of the two distillations.

ELECTRE III-IV software version 3.x was used for the computations (Almeida Dias et al., 2006). Further detailed descriptions of the methodology can be found in the scientific literature (Roy, 1990, 1996; Vincke, 1992; Figueira and Roy, 2002).

### 3.2.4. Reference group and allocation of weights

The panel method was used with a reference group (Rowley et al., 2012) to weight the importance of the indicators presented in section 3.1.1. A reference group was formed by six representatives from a variety of relevant stakeholders in the Swedish on-site sanitation sector. They included representatives of the environmental authority responsible for on-site sanitation systems (Swedish Agency for Marine and Water Management), Swedish Homeowners Association, Federation of Swedish Farmers, Swedish Waste Management Association, and two advisors and communicators working in the small-scale wastewater treatment sector, one from the National Platform for On-Site Sanitation (VA-guiden) and one from the Centre for Water Development in Norrtälje.

Members of the reference group were contacted by email and provided with information about the study, together with an online questionnaire asking them to (individually) allocate weights to the indicators. During a later group meeting the indicators were discussed and the members endorsed the weights already given online or modified them, individually. The indicators were ranked from the most to the least important and individual weights were allocated to each indicator by giving 100 points to the most important indicator(s) and subsequently assigning less points (from 100 to 0) to the others, depending on how important they were considered in relation to the most important one. As the group discussion did not result in a consensus, the arithmetic mean of the normalized weights was used (Eq. 3).

$$W_i = \frac{1}{n} \sum_{i=1}^{n} \frac{w_i^* \cdot 100}{W^*}$$  \hspace{1cm} \text{Eq. (3)}

where:

- $W_i = $ is the weight of indicator $i$,
- $w_i^* = $ are the points allocated by a stakeholder for each indicator, between 0 and 100,
- $W^* = $ are the total points given by a stakeholder to all the indicators, and
- $n = $ is the number of stakeholders.
3.2.5. Scenario analysis

A scenarios analysis was included in Paper IV to investigate plausible settings based on socio-economic and geographic factors, by changing the weights given to certain indicators. Scenario 0 was the baseline scenario, with the initial weights allocated by the reference group. Scenario 1 represented areas where surface waters generally do not have special protection status according to Swedish legislation (Swedish EPA, 2006) and the possibilities to recycle nutrients to farmland were limited, as they may be in some areas e.g., northern Sweden. Hence, the indicators related to nutrient removal and potential for nutrient recycling were given the lowest weights.

Scenario 2 represented areas where the removal of nutrients was important due to the presence of sensitive (e.g., eutrophic) receiving waters, and with possibilities to recycle nutrients onto farmland. The indicators related to removal and potential for recycling nutrients were given the highest weights.

Scenario 3 represented a context where a sociopolitical strategy placed higher demands on climate change mitigation, with emphasis on decreasing atmospheric emissions. Hence, the indicators related to potential for energy recovery, cumulative energy demand and global warming potential were assigned the highest weights.

The weights of the indicators mentioned in each scenario were modified using Simo´s card method (Simos, 1990; Figueira and Roy, 2002). The method involves placing the indicators on cards to improve visualization and organizing them in order of importance, with the possibility of adding blank cards between them to represent larger differences in importance. The schematic representation of the cards’ order is included in the Supplementary Material of Paper IV.

3.3. The interview-based study (Paper V)

The aim of the interview-based study presented in Paper V was to investigate the decision-making processes when planning sanitation systems for existing or soon-to-be-built areas in rural and peri-urban areas within the municipal water jurisdictions. An interview guide was designed, structured with open questions covering themes corresponding to key aspects of the specific case study, the planning process, and aspects related to sustainability within the interviewees’ organizations specifically and the wastewater sector generally. Interviews were conducted with water professionals (civil/water engineers) in relevant positions within their respective organisations e.g., heads of municipal water departments/water utilities, project developers, planners and strategists, who were familiar with the investigated cases.

3.3.1. Selection of case studies

Following a purposive sampling strategy (Flick, 2014), seven cases were selected to illustrate a range of approaches to wastewater management (Table 5). The strategic selection included typical cases of common approaches, e.g., connecting an area to a...
municipal WWTP through the construction of a sewage pipe network, and deviant cases, like an innovative wastewater management approach that included source separation of blackwater and greywater. The intention was to cover a wide range of approaches to solve common problems related to wastewater management, including varying degrees of decentralised and centralised options, system sizes and spatial variability. The authors’ pre-knowledge of the cases, the relevance and representativeness, and the willingness of the interviewees to participate in the study also influenced the selection of the cases.

3.3.2. Interview procedures

Following an expert interview approach, prospective interviewees who were expected to have relevant knowledge and high potential to contribute practical insights because of their professional position and expertise (Flick, 2014) were contacted and invited to participate in the study. They received a short description of the research study and information about data management by email upon agreement. The interviews were conducted online during May 2021 and April 2022, lasted about 60 minutes and were recorded and thereafter translated into English and summarized through paraphrasing and text reduction techniques (Flick, 2014). All respondents were guaranteed anonymity, although most of them considered it unnecessary. Their names are, nevertheless, treated anonymously in this thesis.

The interview guide was thematically semi-structured and included questions about the specific local conditions, the process of choosing the focal technical system and challenges encountered during that process. Thereafter, the questions were more general and covered sustainability considerations, common practices within their organizations and in the sector, awareness and perceptions of new technologies for local reuse of resources and barriers for implementation of those innovative systems. The guide was consistently followed in every interview, with slight adaptations.

After every interview, the interviewees received a closed-ended questionnaire listing a number of drivers and were asked to assign a score, from 0 (irrelevant) to 3 (a main reason) indicating the drivers’ relevance in the process of choosing the focal system, (see Table 5). Additional free text space was provided to allow the capture of information not covered in the questionnaire, and for explanatory comments. Some participants were contacted a posteriori to clarify ambiguities in their narratives. The interview guide and questionnaire are included in the Supplementary Material of Paper V.
Table 5 Overview of the case studies included in the investigation, extracted from Paper V. *LPS stands for low-pressure sewer.

<table>
<thead>
<tr>
<th>Case study</th>
<th>Type of new system</th>
<th>Previous wastewater system</th>
<th>Reasons to change the existing system</th>
<th>Current technical solution</th>
<th>Size</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>Decentralised</td>
<td>Conventional local on-site wastewater systems, for individual houses and clusters including holding tanks, septic tanks, sand filters and drain fields</td>
<td>Area development, following ruling that better wastewater systems were needed in the area and the municipality had to take over the responsibility</td>
<td>Local sewers and on-site facility with chemical precipitation and biological treatment in two sand filters</td>
<td>Dimensioned for 1 200 PE</td>
</tr>
<tr>
<td>B</td>
<td>Semi-centralised (source separation)</td>
<td>Unexploited land publicly owned</td>
<td>City development project in an area with limited raw water sources for drinking water</td>
<td>Separation of blackwater (treated centrally) and greywater (local treatment and reuse)</td>
<td>12 000 PE</td>
</tr>
<tr>
<td>C</td>
<td>Semi-centralised treatment</td>
<td>Four septic tanks with no further treatment</td>
<td>Field inventories showed that the systems were obsolete and needed an upgrade</td>
<td>New sewers and connection to an existing small municipal WWTP</td>
<td>About 23 households (Ca. 100 PE)</td>
</tr>
<tr>
<td>D</td>
<td>Semi-centralised treatment</td>
<td>Small on-site systems, including drain fields, sand filters, package plants and holding tanks</td>
<td>The property owners requested inclusion in the municipal responsibility area and provision with drinking water and wastewater services</td>
<td>New sewers including gravity and LPS* sewers, pumping stations and connection to a new small treatment plant built in parallel to an existing one</td>
<td>1500 PE</td>
</tr>
<tr>
<td>E</td>
<td>Centralised treatment</td>
<td>Four small local plants providing various treatments, including trickling biofiltration, activated sludge processes, and chemical precipitation</td>
<td>Ageing infrastructure, end of technical lifespan</td>
<td>New sewers and connection to the main municipal WWTP in an urban area</td>
<td>Four plants of varying sizes, 70-1500 PE</td>
</tr>
<tr>
<td></td>
<td>Treatment Type</td>
<td>Description</td>
<td>Local Receiving Waters</td>
<td>New Sewers and Connection to WWTP</td>
<td>Household Count</td>
</tr>
<tr>
<td>---</td>
<td>---------------------------------</td>
<td>-----------------------------------------------------------------------------</td>
<td>----------------------------------------------------------------------------------------</td>
<td>-----------------------------------</td>
<td>-----------------</td>
</tr>
<tr>
<td>F</td>
<td>Centralised treatment</td>
<td>Local plant with conventional treatment (mechanical and chemical precipitation)</td>
<td>Ageing infrastructure, end of technical lifespan, sensitive local receiving waters</td>
<td>New sewers and connection to the main municipal WWTP in the urban area (10 km)</td>
<td>2 500 PE</td>
</tr>
<tr>
<td>G</td>
<td>Centralised treatment</td>
<td>Individual on-site wastewater treatment systems of various types</td>
<td>Local receiving waters were affected by eutrophication; the municipal water utility took over responsibility to provide water and wastewater services</td>
<td>New sewers including LPS* systems at each property and connection to the main municipal WWTP in the urban area</td>
<td>About 400 households (Ca. 1600-2000 PE)</td>
</tr>
</tbody>
</table>

### 3.3.3. Data extraction, analysis and interpretation

The interviews provided rich empirical material, which was compiled and analysed following an inductive-deductive approach where observations were used to establish generalisations about the investigated phenomena, and a hypothesis was tested with findings from the case studies (Flick, 2014; Hyde, 2000). The initial hypothesis was that the trend in a high-income country like Sweden is to centralise the (waste)water systems by building connecting pipe networks to existing central infrastructure, wherever feasible. The empirical material was assessed by applying thematic analysis, in which the data were processed in the form of blocks of information and grouped in categories or “themes” according to the types of answers and to identify patterns within the data (Flick, 2014). The themes were selected based on the questions and topics covered during the interviews, the use of keywords and contents of the responses themselves. The themes covered were factors influencing choices of systems, common practice in the interviewee’s organisation and sector, technical decision-making, sustainability conceptualization, perceptions of and barriers hindering implementation of resource recovery sanitation systems and, lastly, reflections about the current wastewater paradigm. Results from the questionnaire were summarized based on the scoring allocated to each driver. Recurring drivers were highlighted and analysed together with the empirical material collected during the interviews. In attempts to establish and maintain plausibility, reliability and objectivity, the results were validated by iterative revisions of interpretations of the interview transcripts and constant monitoring of the research findings. Quotations included in the results and discussion section were used to provide evidence for the themes and illustrate the respondents’ perceptions about a theme.
3.4. The historical review study (Paper VI)

The aim of the study presented in Paper VI was to investigate the historical development of the sanitation sector in Sweden in an attempt to find explanations for the long-standing dominance of the centralised approach and the inertia in establishing alternative sanitation solutions. In this paper, and Paper V, alternative systems refer to sanitation options that differ from the common systems consisting of a small or large sewer network and a treatment plant with mechanical, biological and chemical treatment for mixed wastewater. Examples of alternative systems include source separating examples, like urine diversion and blackwater/greywater separation, but also wetlands or other less commonly applied solutions.

The methodology applied followed a classical historical approach using the main journal of the Swedish water sector, *VAV-nytt/Svenskt vatten*, as the source material for producing knowledge. The journal is written and read by representatives of the sanitation sector in terms of practitioners, officials and politicians at the local and national level, together with small contributions from researchers and private enterprises, so views of a wide range of stakeholders are represented in its issues, e.g., regulators, developers and implementers. The journal covers technical, economic, organizational, regulative, political and managerial aspects of sanitation systems, and has been regularly published over the 40-years studied period (1974-2015). The analysis of the empirical material was inspired by socio-technical transition literature e.g., Geels (2002), to sort, identify, evaluate and contextualize the reasons why alternative sanitation systems have not been implemented to a larger extent, and for the lengthy dominance and “lock-in” of the centralised sanitation approach, using a multi-level perspective framework. This framework helps efforts to deal with the complexity and resistance to change of large socio-technical systems, by investigating them at three levels: micro, meso and macro, corresponding to niches, regimes and landscape, respectively, on which processes interact and align during systems’ transformations (Geels, 2005).

As the sector journal is largely formulated by, and targets the sector itself (where the centralised approach is strengthened), it was considered appropriate source material to obtain insights into the main explanatory themes during the studied period. The potential inherent bias resulting from the choice of the source material was, however, acknowledged by the authors and dealt with, to some extent, by supplementing the narrative with interviews, official reports and scientific literature to obtain a more complete picture of certain processes and events.
4. Results

4.1. Treatment efficiency of on-site wastewater systems

4.1.1. BOD, phosphorus and indicator bacteria removal and discharge from package plants and sand filters with phosphorus treatment

The influent concentrations of considered variables (BOD, P and indicator bacteria) in the investigated facilities were highly variable generally, as shown for instance in Figures 6 and 7. The variability of the effluent concentrations in the facilities sampled both flow-proportionally (D-L) and by grab samples (ASC-TF) depended on the facility.

The coefficient of variation (indicating the spread of data relative to the mean) for the effluent concentrations of tot-P was generally lower in the package plants sampled flow-proportionally (20.5-44.6) than in the package plants where the effluents were taken as grab samples (31.6-124.5). The difference between the two groups indicates that the sampling technique may have significantly affected the samples' representativeness. However, fewer samples were also taken from the package plants sampled flow-proportionally (I-L, n=6) than the rest (ASC-TF, n=7-10). The coefficient of variation of the tot-P effluent concentrations in the sand filters (D-H) and package plants (I-L) sampled flow-proportionally did not differ significantly.

BOD effluent concentrations ranged from 1 to 175 mg L\(^{-1}\) (Figure 6) with some facilities (ASF1, F, H) often achieving concentrations below the detection limit of 1 mg L\(^{-1}\). Facilities with a sand filter followed by an alkaline P-filter removed BOD almost completely (99%) and had lower BOD effluent concentrations (<1–3.1 mg L\(^{-1}\)) than package plants, which had a wider range of measured effluent BOD concentrations (1.1–175 mg L\(^{-1}\)) and removal rates (42–97%) (Figure 6, Table 6). Hence, the recommended BOD removal of 90% (SwAM, 2016a) was achieved by all the facilities with a sand filter followed by an alkaline P-filter and six of the 11 package plants (Table 6).
Figure 6 BOD₇ concentrations (mg L⁻¹) measured in the investigated on-site wastewater treatment facilities. The red and blue markers indicate median influent (IN) and effluent (EFF) concentrations, respectively.

Dilution problems were suspected in sand filter E, based on field observations, e.g., high measured flows even when users were not at home and very clear effluent wastewater, and in the influent wastewater of package plant ASF1, which was very clear likely due to the poor status of the pipe network (causing groundwater infiltration and inflow) and backflush water from a local drinking water plant. Additionally, package plant ASC provided only 42% BOD removal, with insufficient removal of suspended solids (TSSᵢnf = 65.7 ± 30.4 mg L⁻¹; TSSₑff = 51.2 ± 16.9 mg L⁻¹) and high turbidity (median value: 115 Nephelometric Turbidity Units), indicating that its treatment process was not efficient.
Table 6 Removal rates of BOD, tot-P and indicator bacteria. More than half of the samples of effluents of facilities marked with an asterisk (*) had BOD and indicator bacteria levels below the detection limits of 1 mg L⁻¹ and 10 CFU or MPN, respectively.

<table>
<thead>
<tr>
<th>Facilities</th>
<th>Type of treatment</th>
<th>BOD (%)</th>
<th>Tot-P (%)</th>
<th>E. coli (Log10)</th>
<th>Enterococci (Log10)</th>
</tr>
</thead>
<tbody>
<tr>
<td>D</td>
<td>Sand filter + P-filter</td>
<td>99</td>
<td>96</td>
<td>4.2*</td>
<td>3.6*</td>
</tr>
<tr>
<td>E</td>
<td>Sand filter + P-filter</td>
<td>99</td>
<td>92</td>
<td>2.4*</td>
<td>4.0*</td>
</tr>
<tr>
<td>F</td>
<td>Sand filter + P-filter</td>
<td>99*</td>
<td>99</td>
<td>3.3*</td>
<td>2.7*</td>
</tr>
<tr>
<td>G</td>
<td>Sand filter + P-filter</td>
<td>99</td>
<td>98</td>
<td>2.7*</td>
<td>1.4</td>
</tr>
<tr>
<td>H</td>
<td>Sand filter + P-filter</td>
<td>99*</td>
<td>99</td>
<td>3.5*</td>
<td>2.3</td>
</tr>
<tr>
<td>I</td>
<td>Package plant + P filter</td>
<td>68</td>
<td>92</td>
<td>1.1</td>
<td>0.6</td>
</tr>
<tr>
<td>J</td>
<td>Package plant + P filter</td>
<td>88</td>
<td>48</td>
<td>0.4</td>
<td>1.2</td>
</tr>
<tr>
<td>K</td>
<td>Package plant + P filter</td>
<td>91</td>
<td>64</td>
<td>0.7</td>
<td>0.8</td>
</tr>
<tr>
<td>L</td>
<td>Package plant + P filter</td>
<td>72</td>
<td>92</td>
<td>2.9</td>
<td>1.4</td>
</tr>
<tr>
<td>ASC</td>
<td>Package plant + coagulant</td>
<td>42</td>
<td>13</td>
<td>0.2</td>
<td>0.5</td>
</tr>
<tr>
<td>ASF1</td>
<td>Package plant + P filter</td>
<td>70</td>
<td>3</td>
<td>1.5</td>
<td>1.6</td>
</tr>
<tr>
<td>ASF2</td>
<td>Package plant + P filter</td>
<td>93</td>
<td>83</td>
<td>2.5</td>
<td>2.3</td>
</tr>
<tr>
<td>ASF3</td>
<td>Package plant + P filter</td>
<td>91</td>
<td>46</td>
<td>1.8</td>
<td>1.1*</td>
</tr>
<tr>
<td>SBR1</td>
<td>Package plant + coagulant</td>
<td>96</td>
<td>78</td>
<td>2.1</td>
<td>1.5</td>
</tr>
<tr>
<td>SBR2</td>
<td>Package plant + coagulant</td>
<td>97</td>
<td>86</td>
<td>1.8</td>
<td>1.9</td>
</tr>
<tr>
<td>TF</td>
<td>Package plant + coagulant</td>
<td>91</td>
<td>95</td>
<td>1.3</td>
<td>2.1</td>
</tr>
</tbody>
</table>

The effluent concentrations of total phosphorus ranged from 0.02 to 6.4 mg L⁻¹ (with an outlier of 8.6 mg L⁻¹) (Figure 7). Similarly to the BOD treatment results, facilities with a sand filter followed by an alkaline P-filter had lower effluent concentrations (0.02–2.2 mg L⁻¹) and removed tot-P to a greater extent (>92%) than package plants, which showed a wider range of treatment efficiencies (3–95%) and effluent concentrations (0.2–8.6 mg L⁻¹) (Table 6).

The current Swedish guidelines suggest that total phosphorus effluent concentrations should not exceed 1 and 3 mg L⁻¹ for high and standard protection level of the receiving waters, respectively, based on a domestic consumption of 170 L day⁻¹person⁻¹ and removal rates of 90 and 70%, respectively (SwAM, 2016a). However, recent assessments suggest that a more accurate water consumption would be approximately 99 L day⁻¹ person⁻¹ (Herrmann et al., 2021), resulting in effluents limits of 2 and 6 mg L⁻¹ of total phosphorus for receiving waters with high and standard protection status, respectively. Four facilities (I, J, ASC, ASF3) had effluent tot-P concentrations slightly above 3 mg L⁻¹ (Figure 7).

The recommended tot-P removal rates (70–90%) were achieved by all the soil-based systems with an alkaline phosphorus filter (D–H) and six of the 11 package plants: three with alkaline filters (K, L, ASF2) and three including a treatment step with coagulants for phosphorus precipitation (SBR1, SBR2, TF) (Table 6). Of the package plants that did not achieve adequate tot-P removal, the hydraulic load of facilities J and K was high (J=79.4 Lm⁻² h⁻¹; K=58.5 Lm⁻² h⁻¹) because of the high number of users, and an effluent
pH below 9. The effluent pH of package plant ASF3 was also below 9 towards the end of the sampling campaign, indicating that the alkaline filter was exhausted, whereas the alkaline filters of ASF1 were likely exhausted due to high influent loads caused by infiltration and inflow of soil water (and backflush water from a drinking plant) into the influent sewers. Facilities with alkaline P filters (n= 12) showed significantly lower tot-P effluent concentrations than facilities with chemical P treatment (n=4) (\(p=0.005\), Mood’s median test). However, due to the small sample size of the latter, the comparison should be interpreted with caution.

![Total phosphorus (mg L\(^{-1}\)) concentrations measured in the investigated on-site wastewater treatment facilities.](image)

**Figure 7** Total phosphorus (mg L\(^{-1}\)) concentrations measured in the investigated on-site wastewater treatment facilities. The red and blue markers indicate median influent and effluent concentrations, respectively. Orange lines indicate maximum recommended effluent concentrations for receiving waters with standard (solid line) and high (dashed line) protection levels. Correspondingly, the green lines indicate suggested maximum effluent concentrations based on 99 L day\(^{-1}\) person\(^{-1}\) average water consumption.

Mood’s median test for non-parametric data revealed a significant difference (\(p<0.003\)) between soil-based systems and package plants (both groups including a phosphorus treatment step) in terms of BOD, TOC and total and dissolved phosphorus effluent concentrations (Figure 8). The effluent concentrations of organic compounds and phosphorus were generally higher in the package plants than in the soil-based systems with P removal.
Figure 8 BOD (A), TOC (B), total phosphorus (C) and dissolved phosphorus (D) effluent concentrations for the two categories of facilities: soil-based systems (sand filters + phosphorus filter) and package plants.

The presence of left-censored data in the effluent BOD concentrations (e.g., facilities F and H) might affect the reliability of the comparison. However, the analysis of effluent TOC, total and dissolved phosphorus suggested the same results (Figure 8) and the concentrations of these parameters were not left-censored.

Effluent densities of the two indicator bacteria were significantly higher in the package plants than in the soil-based systems ($p = 0.000$), as shown in Figure 9. Most of the bacterial effluent densities in the soil-based systems were below the detection limit, which was accounted for using the Peto-Peto statistical test that considers left-censored data. A distribution probability analysis of the effluent indicator bacteria density indicated that the package plants had a higher probability of elevated effluent concentrations of both indicator bacteria than soil-based systems.
The effluent E. coli concentrations of package plants were nearly always higher than the recommended maximum density of 100 CFU/100 ml in wastewater effluents in areas with receiving waters classified as sensitive according to the Swedish authorities (SwAM, 2016b). Furthermore, the densities for excellent bathing water quality (for inland waters) set in the EU bathing water directive (EU, 2006), 200 CFU/100 mL for enterococci and 500 CFU/100 mL for E. coli, were generally exceeded in the sampled effluent from package plants, in contrast to effluents of the sand filters, but there were exceptions. For example, E. coli levels were below 200 CFU/100 ml in four out of six samples of effluent of package plant L, and enterococci densities were below the detection limit in all but one sample of ASF3 effluent. The average E. coli removal rate was generally higher in sand filters (2.4–4.2 log10) than in package plants (0.4–2.5 log10), whereas the removal of enterococci varied between 1.4–4 log10 for sand filters and 0.5–2.3 log10 for package plants (Table 6).

4.1.2. Nitrogen removal in package plants

Nitrogen compounds in the influent and effluent of package plants were analysed in the study presented in Paper III. Most of the influent nitrogen was in the form of ammonia in all facilities (Figure 11) and tot–N removal rates were significant at five of the seven package plants: ASF2 (21%), ASF3 (23%), SBR1 (55%), SBR2 (79%) and TF (17%). Most of the nitrogen removed was organic, likely through particle sedimentation or filtration in the P filters. In facilities SBR1 and SBR2, denitrification of NO₂⁻NO₃ also contributed to tot-N removal (42% and 64% of the nitrified N was estimated to be denitrified). Only these two facilities met the N removal levels recommended by the Swedish authorities.
for receiving waters with high protection level (50%, SwAM, 2016a). Despite the good removal rates, the effluent tot-N concentrations in SBR1 and SBR2 (18.9 ± 13.3 and 7.2±7.4 mg L⁻¹, respectively) were within ranges of concentrations measured in other facilities due to the high influent concentrations (Figure 8), likely due to partial recirculation of the sludge.

**Figure 11** Concentrations of tot-N (left) and NH₄-N (right) recorded in seven package plants addressed in Paper III. The red and blue markers indicate median influent and effluent concentrations, respectively.

Nitrification occurred in most studied package plants, to varying degrees. For example, up to 51 and 78% of the NH₄⁺-N was nitrified in the batch reactors SBR1 and SBR2, respectively, while in the facilities operating in continuous mode ASF2 and ASF3 less than 44 and 27% of the NH₄⁺-N was nitrified, respectively. The water/sludge recirculation in the batch reactors likely promoted the denitrification processes by transporting the nitrified compounds from the process tank into the primary sedimentation unit with sufficient substrate and anaerobic conditions. In continuous plants the average denitrification rate was very low or negligible and the septic tank would be the only place that could potentially meet anaerobic conditions (with available substrate). The wastewater temperatures recorded in Paper III were generally too low for high nitrogen removal rates and were, at times, below the limit of 8.3°C for nitrification and the critical threshold of 4°C (no growth) for nitrifiers (Sharma and Ahlert, 1977; Taylor Eighmy and Bishop, 1989).

### 4.1.3. Micropollutants in the effluents of package plants

An investigation of micropollutants was also conducted in two package plants (ASC and TF), and included in Paper III. A list of 19 selected pharmaceutical compounds were analysed in the influents and effluents, with 10 and eight compounds detected in the effluents of the TF and ASC facilities, respectively (Table 7). The detected pharmaceuticals included anti-inflammatory drugs (diclofenac, ibuprofen and ketoprofen), β-blocker drugs (bisoprolol and metoprolol), an ACE inhibitor (enalapril), anticonvulsants (gabapentin and levetiracetam) and the antidepressant venlafaxine.
The stimulant caffeine and sweetener acesulfame K were detected in all the influent and effluent samples of the two studied facilities. Effluent caffeine concentrations were clearly lower than the influent concentrations in facility TF, but not facility ASC, corroborating the poor performance of ASC’s biological treatment.

Table 7 Summary of concentrations of the selected pharmaceuticals, caffeine and acesulfame K (µg L⁻¹) detected in effluents in the study presented in Paper III and previous studies. LOQ= Limit of quantification; ND= Not detected. Median values estimated in this study were only based on detected concentrations. Table taken from Paper III.

<table>
<thead>
<tr>
<th>Compound</th>
<th>This study</th>
<th>Other studies</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Median</td>
<td>Range</td>
</tr>
<tr>
<td>Diclofenac</td>
<td>TF: 0.9</td>
<td>TF: 0.54-14</td>
</tr>
<tr>
<td>ASC: 1</td>
<td>ASC: 0.97-1.2</td>
<td></td>
</tr>
<tr>
<td>Venlafaxine</td>
<td>TF: 1.3</td>
<td>TF: 1.3-1.4</td>
</tr>
<tr>
<td>ASC: &lt;LOQ</td>
<td>ASC: &lt;LOQ</td>
<td></td>
</tr>
<tr>
<td>Bisoprolol</td>
<td>TF: 0.06</td>
<td>&lt;LOQ-0.06</td>
</tr>
<tr>
<td>ASC: 0.3</td>
<td>ASC: 0.2-0.5</td>
<td></td>
</tr>
<tr>
<td>Ibuprofen</td>
<td>TF: 0.6</td>
<td>TF: 0.006-1.5</td>
</tr>
<tr>
<td>ASC: 56</td>
<td>ASC: 22-67</td>
<td></td>
</tr>
<tr>
<td>Ketoprofen</td>
<td>TF: 0.6</td>
<td>TF: 0.006-1.5</td>
</tr>
<tr>
<td>ASC: &lt;LOQ</td>
<td>ASC: &lt;LOQ</td>
<td></td>
</tr>
<tr>
<td>Metoprolol</td>
<td>TF: 2.8</td>
<td>TF: 1.6-2.9</td>
</tr>
<tr>
<td>ASC: 0.7</td>
<td>ASC: 0.7-1.5</td>
<td></td>
</tr>
<tr>
<td>Gabapentin</td>
<td>TF: 0.03</td>
<td>TF: &lt;LOQ-15</td>
</tr>
<tr>
<td>ASC: 52</td>
<td>ASC: 52-78</td>
<td></td>
</tr>
<tr>
<td>Levetiracetam</td>
<td>TF: 0.6</td>
<td>TF: &lt;LOQ-0.8</td>
</tr>
<tr>
<td>ASC: 0.4</td>
<td>ASC: 0.3-0.5</td>
<td></td>
</tr>
<tr>
<td>Caffeine</td>
<td>TF: 96</td>
<td>TF: 73-192</td>
</tr>
<tr>
<td>ASC: 110</td>
<td>ASC: 99-149</td>
<td></td>
</tr>
<tr>
<td>Acesulfame K</td>
<td>TF: 16</td>
<td>TF: 37-123</td>
</tr>
<tr>
<td>ASC: 17</td>
<td>ASC: 7.1-19</td>
<td></td>
</tr>
</tbody>
</table>

a) Loos et al. (2013). Effluents from 90 WWTPs.
b) Luo et al. (2014). Review of effluents from WWTPs in various countries.
c) Matamoros et al. (2009). Effluents from 13 on-site wastewater treatment systems.

The effluent concentrations of diclofenac, metoprolol and ibuprofen were above those reported in the literature (Table 7), with no significant removal of diclofenac achieved in any of the package plants, and low to moderate removals of metoprolol (<10%) and ibuprofen (<45%) achieved in facility TF. Moreover, all the effluent diclofenac
concentrations recorded in the study exceeded 0.5 µg L\(^{-1}\) (Table 7), a concentration shown to cause toxicity to aquatic organisms even with conservative estimates (Leverett et al., 2021). The ranges of detected effluent concentrations of venlafaxine, bisoprolol, ketoprofen, gabapentin and levetiracetam were similar to those previously reported in the literature including effluents from large WWTPs and on-site treatment systems (Table 7).

Seven of the 15 analysed phthalates (DEHP, BBP, DEHT, DBP, DEP, DIBP, DMP) were detected on at least one sampling occasion in effluents from the two studied plants, and effluent concentrations of DEP and DIBP were highest (380 and 270 ng L\(^{-1}\) in TF, 240 and 310 ng L\(^{-1}\) in ASC, respectively). DBP and DEHT concentrations were also high in effluents of facility ASC (408 and 380 ng L\(^{-1}\), respectively). Concentrations of phthalates in facility TF were consistently lower in the effluent than the influent, indicating some removal, but concentrations of some were higher in the effluent from the plant ASC than the influent. DEHP is the only phthalate included in the European Union’s list of priority substances (EC, 2013). The highest DEHP effluent concentration recorded in Paper III was 0.27 µg L\(^{-1}\) (Paper III, Supplementary material), which is lower than the environment quality standard (EQS, annual average value) for inland and other surface waters of 1.3 µg L\(^{-1}\) set by the European Commission (EC, 2013).

4.2. Sustainability indicators and scenarios to support systems thinking when planning sanitation required for existing/new-build areas

Ten indicators (with sub-indicators) were defined and assessed in Paper IV to evaluate selected options for on-site sanitation in terms of sustainability performance (Table 8). The indicators were developed based on existing literature (Hellström et al., 2000; Balkema et al., 2001; Kalbar et al., 2012; Ibáñez-Forés et al., 2014) and discussions with stakeholders. Furthermore, the key criteria for a sustainable wastewater system were explored in Paper V through interviews where water professionals working in different municipalities and water utilities shared their opinions and views. Outcomes of the interviews were also summarized in the form of indicators equivalent to those used in Paper IV (Table 8) for comparison.

Most of the sustainability indicators developed in Paper IV were mentioned and covered during the interviews included in Paper V, with some minor differences, supporting the choice and validating their importance for assessing the sustainability of sanitation systems. The removal of nutrients is pertinent to their impact on the receiving environment, whereas the recovery of nutrients enables their reuse and substitution of commercial fertilisers. The significant content of nutrients in wastewater makes these aspects very relevant for closing the nutrient loops in anthropogenic (food) systems. Estimating the carbon emissions associated with the sanitation systems is highly relevant in a context of climate change, due to the desirability of low carbon footprints. Thus, a global warming potential indicator was assessed in the multicriteria analysis (Paper IV). This included
emissions from the production of considered systems’ components and consumables, as well as transport and emissions from the storage of waste fractions. However, it was only mentioned indirectly in the interviews in Study V, as some interviewees suggested that emissions from transport of sludge or use of chemicals should be accounted for in a sustainability assessment. Water consumption and reuse for irrigation purposes are becoming important aspects to consider in the planning stage as the availability of water resources is being affected by extensive extraction and changes in precipitation patterns, and the social awareness is increasing. Implementing systems that promote lower water usage, for example, were suggested by some interviewees (Paper V).

Table 8 Sustainability indicators included in Paper IV and indicators developed from the interviews included in Paper V.

<table>
<thead>
<tr>
<th>Scale</th>
<th>On-site sanitation</th>
<th>On-site, small to large systems</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Environmental</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Criteria</td>
<td>Multicriteria analysis with sustainability indicators (Paper IV)</td>
<td>Criteria for a sustainable wastewater system (Paper V)</td>
</tr>
<tr>
<td></td>
<td>On-site sanitation</td>
<td></td>
</tr>
<tr>
<td>Removal of P and N</td>
<td>Nutrient removal</td>
<td></td>
</tr>
<tr>
<td>Potential for nutrient recovery</td>
<td>Nutrient reuse and recycling</td>
<td></td>
</tr>
<tr>
<td>Energy recovery</td>
<td>Energy recovery (biogas, heat)</td>
<td></td>
</tr>
<tr>
<td>Cumulative energy demand</td>
<td>Energy use</td>
<td></td>
</tr>
<tr>
<td>Global warming potential</td>
<td>Emissions from transport and use of chemicals</td>
<td></td>
</tr>
<tr>
<td>-</td>
<td>Water reuse e.g., for irrigation, rainwater harvesting</td>
<td></td>
</tr>
<tr>
<td>-</td>
<td>Water usage</td>
<td></td>
</tr>
<tr>
<td><strong>Economic</strong></td>
<td>Capital cost</td>
<td>Investment cost, return on investment</td>
</tr>
<tr>
<td></td>
<td>Operation and maintenance cost</td>
<td>Operation and maintenance cost</td>
</tr>
<tr>
<td><strong>Technical</strong></td>
<td>Robustness</td>
<td>Robustness/reliability</td>
</tr>
<tr>
<td>-</td>
<td>Status of sewer network/infiltration and inflow rates in sewers</td>
<td></td>
</tr>
<tr>
<td><strong>Social/health</strong></td>
<td>Social acceptance</td>
<td>Comfort and convenience</td>
</tr>
<tr>
<td>Risk of pathogen discharge</td>
<td>Public health protection</td>
<td></td>
</tr>
<tr>
<td><strong>Managerial</strong></td>
<td>-</td>
<td>Clarity in legislation</td>
</tr>
</tbody>
</table>
Economic aspects are often argued to be the key criteria in the decision phase of implementation of a sanitation system, especially at the lowest scale when houseowners have to cover the whole cost of a private system. Thus, both studies included investment costs (considering the return on investment), and both operation and maintenance costs. Robustness is an important indicator reflecting the technical dimension of sanitation systems and was defined (qualitatively) in terms of systems’ risks of failure and adaptability to fluctuations in flow (Paper IV), and reliability with respect to effluent quality and capacity to perform as intended (Paper V). The importance of the sewer network’s status was also highlighted during the interviews as it often causes infiltration and inflow of extraneous water into systems and affects their treatment performance, especially for large-scale systems covering long distances. In addition, the type of sewer network, e.g., gravity sewers vs low-pressure sewers (LPS), was suggested (Paper V) as a relevant aspect to consider when assessing the sustainability of sanitation systems, as low-pressure sewers use pumps (and additional energy) and were considered to have higher technical requirements in terms of installation, operation and maintenance than gravity sewers.

The social sustainability dimension of sanitation systems encompasses social and health-related aspects, including their general social acceptance (Paper IV) and their comfort and convenience for the end-user (Paper V). In addition, the effluents’ hygienic quality is an important element of the public health protection responsibilities of public operators (municipalities or water utilities), and must be taken into consideration in sustainability assessments, as risks of pathogen discharges from sanitation systems should be minimized as much as possible.

The interviewees suggested that clear guidance by the authorities and better specifications in the legislation, e.g., in terms of climate emissions or resource recovery requirements, would facilitate the planning of wastewater systems (Paper V) and favour systems with low carbon footprints or those designed to recover and recirculate nutrients. Similarly, the scenario analysis presented in Paper IV indicated that treatment alternatives allowing nutrients recovery not only offered better nutrient recovery but also lower global warming potential than other alternatives like package plants.

**Sustainability indicators analysis of on-site sanitation systems**

The implementation of indicators to assess the performance and sustainability of on-site sanitation systems (Paper IV) allowed transparent and overall comparison of the considered alternatives. The removal of nutrients was found to be highest in the studied source separation alternatives, blackwater and greywater (S1) and urine diversion (S2), for both N and P because most of the nutrients (90% of the N and 80% of the P) are contained in the faeces and urine (Jönsson et al., 2005). Treatment alternatives with a P removal step, using either alkaline P-filter material (A3 and P1) or the addition of chemicals (A4, A5, P2) were also found to provide high P removal (90%). The alternatives with a sand filter (A1) or drain field (A2) were found to have the lowest P removal (40%) and they are not designed to be a stand-alone treatment for long-term P removal (Eveborn et al., 2012; Sinclair et al., 2014) but are instead intended to degrade carbonaceous material. Similarly, the highest potential for nutrients recycling (both P and
N) was offered by the source separation systems S1 and S2. The LCA-based indicators, global warming potential and cumulative energy demand, were lowest for the simpler systems, A2 followed by A1 (the standard drain field and sand filter), due to the lower use of components and consumables (extra tanks, chemicals, and filter material). For most of the alternatives, the largest contributors to the LCA-based indicators were the production of the components and treatment of the sludge (anaerobic digestion and dewatering). For the package plant alternatives, the additional use of electricity also contributed to their energy use and emissions. The potential to recover energy (assessed qualitatively) was considered higher for the alternatives with chemical precipitation of phosphorus (A4, A5, P2), as larger volumes of sludge (rich in organic matter) could be collected from the septic tanks and digested to produce biogas. The investment costs were lowest for the stand-alone sand filter (A1) and drain field (A2) alternatives because of their simplicity and smaller number of components. The source separation alternatives required investments in double tanks and adapted toilets, an ultra-low flushing toilet for the blackwater separation (S1) and urine-diverting toilet for the urine diversion alternative (S2). However, the investment costs were highest attained for the alternatives with package plants and the soil-based system with alkaline P-filter material (A3) because of the purchase costs of the plants, filter bag and tank. The operation and management costs consisted of the yearly emptying of the tank(s), purchase of alkaline filter material for the phosphorus treatment, electricity consumption, and management contracts with providers for routine maintenance such as cleaning, replacement of worn parts and sampling of sludge and effluent water. In terms of social acceptance, the conventional systems (A1 and A2) had very high acceptance because of their convenience and low complexity (Lennartsson et al., 2009). For the package plants (P1 and P2), acceptance was high, despite their complexity as their design made them convenient for the operators, who did not have to monitor them regularly as management and maintenance were included in the management contracts. The alternatives with chemical dosing equipment installed in the households (A4 and A5) had medium acceptance, because of the inconvenience of the space usage and more frequent monitoring (e.g., refilling the dosing tank), which may require greater effort from the homeowners than (for example) changing a P-filter every 2–3 years as for A3. The blackwater separation system (S1) had higher acceptance than the system with urine diversion (S2) which may cause problems with odours and inconveniences (e.g. extra cleaning) and users require pre-knowledge of the system (Hellström and Jonsson, 2003; Larsen et al., 2009; McConville et al., 2017a). Simpler systems like sand filters (A1) and drain fields (A2) were assessed as having high robustness, as these systems generally work well if they are correctly designed and loaded (Palm et al., 2002), with clogging of the filter material being the main risk (USEPA, 2002; Rolland et al., 2009). Adding an extra polishing step to those systems, as in alternatives A3–A5, reduced assessed robustness to medium because of the increased risks associated with additional components and consumables that require replacement (Hellström and Jonsson, 2003). Similarly, the system with blackwater separation (S1) was assessed as having medium robustness because the holding tank for blackwater does not adapt to flow fluctuations like a septic tank with an outlet, and the need to monitor an extra tank and use of a vacuum toilet could increase risks of technical complications. The
urine diversion system (S2) and package plants (P1 and P2) had the lowest robustness because of their higher complexity. The urine-diverting toilets may present problems with ventilation or blockages of the urine-conducting pipe (Udert et al., 2003). Package plants generally have risks of failure due to the presence of moving parts, sensors or electrical control systems (USEPA, 2002) and they are also often sensitive to operational disturbances (Hübinette, 2009). Lastly, the risk of pathogen discharges was determined to be lowest for the blackwater separation system, because the faeces fraction (which has the largest pathogen load in wastewater), was stored in a holding tank then collected and treated in a separate facility. The results for each alternative varied depending on the number of technical treatment barriers included in the sanitation systems (Stenström, 2013). The alternatives with package plants (P1 and P2) and the systems with a specific P removal step (A3, A4) were considered as having two barriers and thus a lower risk of pathogen discharge than alternative A1 with only sand filter (one barrier) and S2 (one barrier for the faecal fraction), or the alternatives A2 and A5 with drain fields (one barrier).

Scenario analysis

As the importance of the indicators may vary, depending on the context, a scenario analysis was applied in Paper IV, with changes in the weights allocated to each sustainability indicator to evaluate settings with different socio-demographic and environmental conditions. These included: areas with standard receiving waters and limited farmland (Scenario 1); areas with sensitive receiving waters and presence of farmland (Scenario 2); areas subject to a political strategy assigning high importance to energy recovery and climate change mitigation (Scenario 3). Results of the scenario analysis showed that changes in the priorities of decision-makers, by increasing or reducing the importance assigned to certain indicators, can affect the sustainability ranking of the alternatives considered (Table 9).

Table 9 Ranking of alternatives in indicated scenarios. Table taken from Paper IV.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Description</th>
<th>Ranking of alternatives</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scenario 0</td>
<td>With original weights from reference group</td>
<td>S1 &gt; A1, A4 &gt; A3 &gt; A2 &gt; A5 &gt; S2 &gt; P2 &gt; P1</td>
</tr>
<tr>
<td>Scenario 1</td>
<td>Lowest importance for nutrient-related indicators (e.g., in sparsely populated areas with non-eutrophic receiving waters)</td>
<td>A1, A4 &gt; A2, S1 &gt; A5 &gt; A3 &gt; S2 &gt; P2 &gt; P1</td>
</tr>
<tr>
<td>Scenario 2</td>
<td>Highest importance for nutrient-related indicators (e.g., in densely populated countryside with farmland and eutrophic receiving waters)</td>
<td>S1 &gt; S2 &gt; A4 &gt; A5 &gt; A3, P2 &gt; P1 (A1, A2)*</td>
</tr>
<tr>
<td>Scenario 3</td>
<td>Highest importance to energy recovery, CED and GWP (e.g., change in political strategy)</td>
<td>A4, A5 &gt; A1 &gt; A3, P2 &gt; A2 &gt; S1 &gt; S2 &gt; P1</td>
</tr>
</tbody>
</table>

*Alternatives A1 and A2 were excluded from the ranking in scenario 2 because they generally do not meet the Swedish guidelines in terms of nutrients removal.
The position of the blackwater separation alternative S1, which scored highest in the baseline scenario, was lower in the rankings of most sustainable alternatives in Scenario 1, as the simpler systems with a sand filter (with and without a P-removal step) obtained the highest rankings. In contrast, the source separation systems S1 and S2 clearly outranked the other alternatives in Scenario 2 because of their good performance in terms of prioritised indicators (nutrient removal and recycling). Scenario 3 represented a context where climate-emissions were prioritized, which favoured the soil-based alternatives that included chemical removal of P, due to their higher potential to recovery energy and moderate emissions and energy use. In this scenario, the source separation alternatives S1 and S2 fell to the lowest rankings because of their relatively high climate emissions, mainly due to the use of extra tanks, transport of larger volumes and the N₂O emissions during treatment and storage of the blackwater and urine. The system boundaries of the multi-criteria analysis did not include replacement of mineral fertilizers with sanitized wastewater fractions as the focus was on the systems’ treatment function.

4.3. Historical context and barriers to implementation of alternative and resource-oriented wastewater systems

The barriers and obstacles (perceived reasons) to implementation of sustainable (as described in Section 2.3) and resource-oriented sanitation systems were discussed during the interviews presented in Paper V. Furthermore, historical factors affecting the slow implementation of alternative systems that focus on recourse recovery, like blackwater/greywater separation and/or urine diversion (and the dominance of the centralised approach) were addressed in Paper VI. This analysis focused on trends and events that occurred in the last 40 years in the wastewater sector, using the main journal of the Swedish sanitation sector (VAV nytt/Svensk Vatten) as a major source of information.

Sweden is an interesting case in the context of resource-oriented sanitation because the country was an early pioneer of urine diversion and there has been relatively strong political awareness of alternative sanitation solutions (Paper VI). However, the implementation rates have been slow. The centralisation of the Swedish sanitation system (based on sewers for mixed wastewater connected to a main WWTP) is continuously expanding, as discussed in the interviewee-based study (Paper V), which clearly showed that the centralisation of areas with local or no municipal sanitation was generally the preferred option. A large-scale technological system, such as the centralised sanitation system, is the overall result of many, long-standing incremental changes, decisions, and efforts. According to the historical review (Paper VI), the extensive “lock-in” of the centralised sanitation solution was due to strong national policy intervention for a major, long-term upgrading and development of the sanitation system. A major overall motivation for expansion of the centralised sanitation system in the 1970s was to relieve small local receiving waters of inputs from inadequate and malfunctioning private on-site sanitation systems, and to adapt to increasing treatment requirements. The municipal responsibility to provide water services increased, at the same time that a decision was
taken that water and sanitation services were not suitable for privatization and commercialization. In the 1960s and 1970s, the Swedish centralised sanitation system underwent large-scale development in terms of improvements in biological and chemical treatment. This involved a wide array of policy measures, including municipal and environmental reforms, large-scale infrastructural visions, government control, and governmental subsidies. Technical and environmental challenges arose in parallel with the expansion, such as structural and operational problems with the long sewer networks, pumping stations and treatment facilities that caused interruptions. Additionally, the operating personnel often lacked sufficient competence.

Small plants and on-site systems also experienced challenges during the study period, due to tightened environmental requirements and neglect. Due to greater difficulties in meeting the stricter regulations with smaller plants, partly associated with the higher flow and pollution variations, connecting sewers were regarded as better options, even if they were more expensive. Inappropriate maintenance, the lack of relevant research and lack of clarity in guidelines and responsibilities affected their implementation and function, and local inventories of on-site systems during the early 2000s showed that management was often inadequate.

The centralised approach started to be questioned during the great environmental mobilization of the late 20th century, and alternative, smaller scale “environmentally sound” technologies started to be considered as perceptions about wastewater and recycling changed. However, the established wastewater treatment sector did not appear to consider alternatives such as dry toilets and urine diversion systems as realistic options, because of bad reported experiences, the infrastructural requirements and challenges associated with the collection, treatment, storage and spreading of the waste fractions. Additionally, finding farmers to use the nutrients remained difficult according to the scrutinized documentation.

Today, barriers like those encountered historically (Paper VI) are still present at municipal levels (Paper V) and the interview-based study highlighted similar obstacles to the implementation of resource-oriented sanitation, such as the financial constraints, lack of incentives, initiative and competences, and uncertainties associated with technology’s immaturity. The barriers suggested by the interviewees were of economic, legislative, socio-political, technical and organizational nature (Paper V).

Due to the financial limitations and lack of competition linked to the sector’s structure, extra incentives were needed to make implementation of alternative sanitation solutions economically feasible and sustainable. This was expressed by one interviewee as follows:

'We’re always a monopoly driven by tariffs, so there’s no competition like in a free market. If there was a free market, there would be another type of pressure on innovation.'

The development of marketable products (fertilizers) and creation of a market for them, are essential, as well as coupling the new system generating the different fractions (e.g.,
greywater or urine separation) with conventional wastewater infrastructure. Policy intervention is thus appropriate in both the market and technology development fields, and a “niche” for the new solutions would need to be created.

Existing Swedish legislation, the Swedish Environmental Act (Swedish government 1999) states that reuse and recycling, as well as other management of materials, raw materials and energy are encouraged with a view to establishing and maintaining natural cycles. However, there appear to be disparities in its interpretation and implementation. The enforcement of more specific and explicit requirements for nutrient recovery and reuse (and not only removal) would, as suggested by several interviewees (Paper V), decrease disparities in the interpretation of the law and favour alternative solutions with focus on resource-recovery. That could prompt the municipalities to change their planning of wastewater systems and increase the prioritization of maximizing the recycling of resources.

The lack of common visions and willingness was also considered an important socio-political barrier to the implementation of alternative systems that enable high rates of resource recovery. Several interviewees reflected on who should be responsible for promoting alternative sanitation systems, and the optimal source of the initiative. The immaturity and shortcomings (in terms of users’ acceptance, and increases in transport and odours) of alternative systems are still emphasized today (Paper V) – as they were in the past (Paper VI) – even when good examples of functioning systems, and solutions to problems that commonly arise, are available. The lack of knowledge, experience and competence – which was also an issue during expansion of the centralised system (but was solved with training and research development) – also poses a barrier at the municipal level, expressed by one interviewee as follows:

‘Perhaps it’s us who are the barriers, we who do not think about it (implementing alternative solutions). It’s difficult when we haven’t done it before and don’t have experience. It’s difficult to think “new”, and there are also risks.’

Lastly, uncertainties associated with the planning, construction, performance and management of alternative systems pose technical challenges to the implementation of alternative systems, while risk aversion and limitations of available time reduce municipalities’ willingness to tackle the challenges.
5. Discussion

5.1. On-site wastewater systems: What hinders their performance?

Results from the field studies (Papers I-III) showed that 26 of 45 inspected on-site wastewater treatment facilities (58%) were not suitable for sampling. The main reasons, in the case of sand and phosphorus filters, were related to the absence of flow in the effluent pipe, lack of an inspection/sampling chamber, inaccessibility due to the design or presence of ponding water in the outlet. The problems encountered with the package plants were more related to the technique and logistics, and it was not always feasible to program the batch reactor facilities (SBRs) to enable their sampling at a reasonable time, as the timing of the batch-treatments depended on available volumes and lasted about 7.5 hours. This prevented checking of these facilities in practice. Lack of flow into some of the alkaline P-filters was a clear sign of bad design and system malfunction, indicating that the effluent water from the precedent sand filters was infiltrating into the soil rather than reaching the subsequent polishing site. Lack of access to outlet pipes to sample effluent does not necessarily indicate malfunction, but inaccessibility to the system makes it difficult or practically impossible to evaluate the treatment performance and efficiency. Similarly, in a monitoring campaign of 40 on-site wastewater systems including sand filters, drain fields and package plants in Sweden, Larsson et al. (2017) found that about half of the facilities did not meet requirements for an appropriate full inspection. Common cited reasons included missing or incorrectly placed pipes, presence of sludge in the distribution chambers or ventilation pipes, incorrect installation or technical problems with the phosphorus precipitation.

Three of the eight sand filters addressed in Papers I and II appeared to have dilution problems, indicating that water infiltrated into the pipes and possibly the sand filter itself. Thus, their technical function could not be expected to match that of a facility with a standard design. The package plant ASF1 also appeared to have dilution problems, likely originating in the sewer network because of its bad condition and the presence of backflush water from a local drinking water plant. Consequently, for these facilities with infiltration and inflow of extraneous water, lower effluent concentrations were observed that were not necessarily due to a good treatment performance. Dilution of influent wastewater is more likely to appear in sand filters, because of their construction and contact with the surrounding environment, than package plants that are contained in units, as discussed by Heinonen-Tanski and Matikka (2017). Moreover, the effluent samples from a package plant are often taken directly from the process tank or sampling chamber, which likely makes the samples more representative than effluents from soil-based systems taken from an outlet pipe (at the far end) where there may be less certainty that they solely represent treated wastewater.

Common reasons for the underperformance of some of the alkaline P-filters addressed in Papers I–III were high hydraulic loads and poorly treated effluent from the preceding biological treatment step (in package plants J–K), and clogging due to age or exhaustion after its expected lifespan (2–3 years) as most reactive calcium ions had precipitated or
been flushed out (in facilities E and ASF3). Adequate design and maintenance, for example changing the alkaline P-filter when the pH is below 9 (Renman and Renman, 2010), would improve the performance of the system as discussed in Papers I-III.

Facilities with package plants have more diverse and complex designs than soil-based systems, so the problems that affect them widely vary. Under-dimensional design (which affects the hydraulic retention and sedimentation times), insufficient aeration or oxygenation, high sludge volumes because of long emptying intervals likely affected the performance of the investigated package plants. Additionally, suboptimal pH, low chemical dosage, insufficient mixing of coagulant, flocculation and/or sedimentation, equipment malfunction (e.g., clogging) or lack of coagulant are commonly reported reasons for low P removal in package plants (Kroiss et al., 2011; Johannessen et al., 2012; Heinonen-Tanski and Matikka, 2017) and may explain the low treatment efficiency observed in some of the investigated plants (e.g., ASC).

A review of on-site sanitation systems’ performance in Western Australia concluded that most failures of the systems were usually associated with inappropriate design and construction, or operation and maintenance, rather than inherent technological flaws (Gunady et al., 2015). Another evaluation of the performance of 21 models of package plants used in Sweden showed they often performed deficiently and few of them treated the wastewater to the same extent as described by the suppliers (Hübinette, 2009). Technical failures like missing or incorrect dosing of chemicals were attributed to the lack of supervision and maintenance.

Apart from technical reasons related to facilities’ design and construction (or maintenance), ambient factors like temperature also affect the treatment process. Varying results on temperature’s effect on the removal of BOD (Niku and Schroeder, 1981) and nutrients have been reported in the literature, from no significant effect in the removal of nutrients in sand filter systems (Christopherson et al., 2005; Kauppinen et al., 2014) to clear effects on, for example, the retention of P in alkaline P-filters (Herrmann et al., 2014), and soil (Barrow and Shaw, 1975). Bacterial growth and activities can be retarded or inhibited by low temperatures (Rodriguez-Caballero et al., 2012), so for example nitrification rates may be higher during warmer periods and the removal of pathogenic microorganisms could be generally reduced during colder periods (Axler et al., 2005; Kauppinen et al., 2014). Paper III specifically discusses the effect of temperature on the treatment processes in package plants. The data collected did not show strong correlations between the effluent parameters and wastewater temperature. However, inverse trends were detected for BOD, nitrogen (in two package plants where nitrification and denitrification occurred), phosphorus (in one alkaline P-filter) and indicator bacteria in effluents of facilities that worked well and had been sampled more than seven times. The low average temperatures (the maximum recorded temperature in SBR1 was <17°C) likely prevented high nitrification rates, which could be enhanced by improving the plants’ insulation to keep the water temperature above the limit for nitrification during all seasons (about 8°C, Sharma and Ahlert, 1977). The outside temperature generally influenced the temperature inside the treatment facilities (Paper III), but the water
temperature remained positive during the monitoring period and the very cold winter
temperatures that typically occur during shorter periods in northern Swedish winters
(reaching temperatures < -20°C) had no visible effect. However, the lowest wastewater
temperatures coincided with the snowmelt period, indicating a stronger cooling effect
(due to water from melting snow infiltrating into the systems) on the wastewater
temperatures than air temperatures. Mattsson et al. (2017) also found that the wastewater
temperatures in the sewer pipes were lowest during the snow melt period (April-June)
in subarctic areas due to large volumes of cold infiltrating waters mixing with the sewage.

5.2. Discharges of nutrients, indicator bacteria and micropollutants from
on-site systems

Facilities with alkaline P filters appeared to have lower tot-P effluent concentrations than
facilities with chemical P treatment (Figures 6, 7 and 8). The low phosphorus removal
observed in some plants with chemical P treatment like ASC, could have been due to
insufficient sedimentation and subsequent release of particles (Johannessen et al., 2012),
insufficient sludge collection and lack of thorough mixing (high turbulence) at the dosing
points, which is required for good precipitation of dissolved P (Kroiss et al., 2011).
However adequate operation and maintenance would likely be sufficient to ensure high
P removal rates, and the results from Paper III suggest that the effect of cold climatic
conditions in P removal can be considered negligible.

Nitrogen removal was only studied in package plants (Paper III) and sufficient degrees
were mainly observed in the two batch reactors (SBR1 and SBR2; 55 and 79%,
respectively) (Figure 11). Nitrogen removal by denitrification can be fostered by
recirculation of the water or sludge from the aerated process tank into a tank with
sufficient carbon substrate such as a primary sedimentation or septic tank with anaerobic
conditions (Shaw and Dorea, 2021), which was present in the batch reactors SBR1 and
SBR2. The alkaline effluent environment in facilities with reactive filter material (e.g.,
ASF2 and ASF3) could, however, inhibit denitrification due to high pH (Renman et al.,
2008). The total nitrogen removal rates in ASF2 and ASF3 were 21 and 23%,
respectively, but the denitrification rates were negligible. The water temperatures in the
studied facilities were lower than the range considered optimal for growth of nitrifying
bacteria (28–36°C) and often below the limit for nitrification and growth of nitrifiers (4–
8°C)(Taylor Eighmy and Bishop, 1989), which likely explains the low nitrification rates
and effluent nitrogen levels in most facilities. Nitrogen removal in sand filters were not
investigated in this thesis, but can be expected to be low, due to inefficient denitrification
as appropriate conditions (an anoxic environment with sufficient organic substrate) are
difficult to establish (Laaksonen et al., 2017).

The indicator bacteria E. coli and enterococci were often found in the effluent of most
facilities (Figure 9). Their densities also generally exceeded limits set for excellent bathing
water quality (for inland waters) in the EU bathing water directive in effluents of package
plants (unlike those of sand filters), with some exceptions. Effluent concentrations
exceeding those established for excellent bathing water quality are not uncommon in
package plants (Vilpas and Santala, 2007; Hübinette, 2009), suggesting that a subsequent polishing step may be required to ensure that some plants provide hygienically acceptable outputs. The removal mechanisms in infiltration systems, such as physical straining and adsorption of bacterial cells to the media (Stevik et al., 2004), may be more effective for removing bacteria than those of package plants with activated sludge, where adsorption onto the biological flocs and settling, predation and decay due to environmental stresses are the main removal mechanisms (Ng et al., 1993; Barrios-Hernández et al., 2020). Porous media such as alkaline filters for phosphorus removal could also potentially contribute to the bacterial retention and removal, as suggested in previous studies (Kauppinen et al., 2014; Herrmann et al., 2017), in addition to high pH (Jenssen et al., 2005; Heistad et al., 2006; Nilsson et al., 2013a). Likely, the retention times in the studied batch reactors and continuous plants were too short for high degrees of removal of the bacteria.

In the study of micropollutant removal (Paper III), caffeine and acesulfame K were detected in all the influent and effluent samples of the two investigated facilities (TF and ASC). Acesulfame K has known persistence and insensitivity to treatment (Luo et al., 2014; Doummar and Aoun, 2018), which were confirmed in this study as the influent and effluent concentrations did not vary greatly. In contrast to acesulfame K, caffeine is readily degradable in conventional WWTPs and on-site sewage systems, with high (>70%) reported removals (Matamoros et al., 2009; Deblonde et al., 2011; Luo et al., 2014). However, the influent and effluent caffeine concentrations recorded in Study III indicated moderate removal (<49%) of the compound. The pharmaceuticals diclofenac and metoprolol have been reported to be poorly removed (<40%), ketoprofen moderately removed (40–70%), and ibuprofen highly removed (>70%) in conventional WWTPs (Luo et al., 2014), and generally similar removal rates were observed in the study, particularly for diclofenac and metoprolol. However, ibuprofen was removed to a smaller extent (<45%) and only in facility TF. High removals of the anticonvulsant gabapentin by activated sludge treatment were not confirmed, in contrast to results reported by Kasprzyk-Hordern et al. (2009). Similar influent and effluent concentrations of the antidepressant venlafaxine in facility TF confirmed its persistence in aerobic processes and ability to resist wastewater treatment, as reported in Falás et al. (2016). The predicted non-effect concentration (PNEC) of ibuprofen is 1 µg L⁻¹ according to the Swedish pharmaceutical database (FASS, n.d.), which was exceeded in all the effluent samples analysed in Paper III. The lowest PNEC for diclofenac is 0.05 µg/l according to the Norman Ecotoxicology Database (Norman, n.d.), but an environmental quality standard (EQS) of 0.04 µg L⁻¹ for diclofenac is under consideration for inclusion in the EU water legislation (watchlist) (Leverett et al., 2021). All effluent diclofenac concentrations recorded in the study exceeded 0.5 µg L⁻¹ (Table 5), implying environmental hazards because of its toxicity to aquatic organisms (Loos et al., 2013).

Phthalates, which are widely used in personal care products and as plasticizers in plastics, have been shown to pose toxicity risks for aquatic and human health due to their endocrine-disrupting activity (Deblonde et al., 2011). The phthalates DEHT, DBP, DEP
and DIBP were found in the influents and effluents of the studied facilities at concentrations below those reported in effluents of conventional WWTPs (Deblonde et al., 2011; Luo et al., 2014; Gao and Wen, 2016), likely due to their smaller scale and domestic wastewater inputs. The removal rate is reportedly greater than 90% for most studied phthalates (Deblonde et al., 2011), but the limited data collected in Paper III prevent confirmation of the high removal rates.

Gros et al. (2017) found no clear differences between on-site package plants and medium and large-scale WWTPs in terms of concentrations and removal rates of pharmaceuticals and personal care products, and also concluded that the removals of micropollutants in both types of facilities seemed to be rather low. In contrast, on-site sand filters showed higher removal of pharmaceuticals (generally >75%) than package plants, likely due to the presence of biofilms, higher biodegradation rates, and soil sorption mechanisms (Gros et al., 2017). Regarding effects of the type of treatment technology, trickling filters reportedly provide lower pharmaceuticals removal (<70%) than activated sludge systems (85%) in full scale WWTPs (Kasprzyk-Hordern et al., 2009). In Paper III, the facility with a trickling filter (TF) seemed to have higher removal than the plant with activated sludge (ASC), which was underperforming, so the results should not be extrapolated. However, the results also showed the reality of many on-site sewage plants that function poorly and discharge significant amounts of micropollutants into the surrounding environment and receiving waters, in contrast to the general results of studies limited to relatively well-functioning sewage systems as highlighted by Blum et al. (2017).

5.3. On-site wastewater systems: What system is better where?

The results presented in this thesis (compiled results of Papers I-III) indicated that soil-based systems with an alkaline P-filter were more likely to achieve lower BOD, phosphorus and indicator bacteria effluent concentrations than package plants (Figure 8). The investigated package plants showing unacceptable BOD removal rates had specific designs, construction and maintenance flaws, as discussed in Section 5.1, which could be amended in practice. In addition, soil-based systems were also more sustainable than package plants according to the multi-criteria analysis (Paper IV). For example, sand filters with and without a phosphorus polishing step scored more highly in terms of single indicators, like robustness, cost and social acceptance. They also had lower LCA impacts. Generally, it was assumed that soil-based systems were more robust than the more advanced package plants (as explained in section 4.2), which have higher complexity and maintenance requirements, based on reports of performance of various commercial package plants (Hellström and Jonsson, 2003; Heinonen-Tanski and Matikka, 2017) and monitoring reports (Hübinnette, 2009). This assumption and the lower performance in terms of other indicators contributed to the low ranking of the alternatives with package plants (P1 and P2, Paper IV). However, the recent development of on-site technology for package plants, which often include sensors and alarm systems to assure treatment efficiency, can make these options more robust and reliable when managed properly (Schneider, 2020), resulting in high nutrient reduction, as shown in Paper III.
Additionally, investments in package plants (but not the soil-based systems) usually include contracted services with the company responsible for their installation for yearly supervision and maintenance visits. Similar studies on the environmental impacts of on-site wastewater systems have concluded that package plants had higher LCA impacts (in terms of carbon footprints and eutrophication potential) than infiltration facilities, mainly because of the associated electricity use, equipment manufacture, sludge transport and treatment, and local emissions (Lehtoranta et al., 2014). The land area requirements were not included as a criterion in Paper IV (Table 3), but their consideration would favour package plants as they are more compact and typically require smaller areas than, for instance, sand filters or drain fields (Balkema et al., 2002). Moreover, package plants can be installed in areas where bedrock, soils or fluctuating groundwater tables limit the implementation of soil-based systems (USEPA, 2002). The two models of package plants included in Paper IV (P1 with an alkaline P-filter and P2 with chemical precipitation for P removal) performed similarly in terms of most of the considered indicators, and only the potential for P recycling and energy recovery was higher for P2 compared to P1.

In terms of phosphorus removal, soil-based systems do not provide sufficiently high removal rates as stand-alone solutions (Paper I) and an extra phosphorus removal step is always required if the receiving waters are sensitive and/or eutrophic. Both soil-based systems with an alkaline P-filter and package plants can treat phosphorus satisfactorily and reach high removal rates (>70-90%) as recommended by the authorities, when installed, operated and maintained adequately. In terms of nitrogen removal, package plants with some kind of water/sludge recirculation would be suitable because significant nitrification and denitrification rates could be reached, based on the results from Paper III and others (e.g., Johannessen et al., 2012). However, the low temperatures may slow the nitrification process and lower rates of nitrogen removal could be expected in winter (as observed, for example, in facilities SBR1-SBR2, Paper III). Improving the plants’ insulation and avoiding any water intrusion during the snow melting season could help to maintain higher temperatures. A more sustainable option for areas where local nutrient discharge must be limited (Scenario 2 in Paper IV) is to install systems that include blackwater or urine separation, where the nutrient-rich fractions are removed from the main wastewater stream and treated centrally elsewhere. The considered treatment and management options for the separated fractions in Paper IV were urea hygienization of the stored blackwater and urine. However, other options such as anaerobic digestion of blackwater (Wendland et al., 2007; Kjerstadius et al., 2015) and partial urine nitrification and distillation (Etter and Uder, 2016; Faust et al., 2022) or urine alkaline dehydration (Simha et al., 2018) have shown promising results. Similarly, areas with agricultural activities that depend on fertilizers inputs would also benefit from implementation of blackwater/greywater or urine separation systems as they provided the highest potential to recycle nutrients of the alternatives compared in the research underlying this thesis (Paper IV). According to Lehtoranta et al. (2022), the implementation of source separating systems (blackwater and urine) in peri-urban and rural areas (where agriculture is actually practiced) could contribute to replacement of up to 60 and 65% of mineral nitrogen and phosphorus fertilisers, respectively, while reducing the generation of CO2-
eq. by 24–26% and the eutrophication impact by 44–53%, depending on the context, compared to conventional centralisation of the wastewater treatment in those areas.

In the third scenario presented in Paper IV the highest importance was assigned to the indicators related to energy (energy recovery and cumulative energy demand) and climate change (global warming potential) so the systems with chemical precipitation were favoured because of their larger volumes of generated sludge (rich in organic matter) that could be digested to produce biogas (Table 9). In this scenario, the ranking of the greywater-blackwater alternative, which topped the sustainability ranking in the baseline scenario, fell because of the slightly larger climate emissions and supposedly lower potential to recycle energy (greywater produced less sludge). These results highlight the potential to optimize greywater-blackwater systems, by including (for example) anaerobic digestion to recover biogas (Kjerstadius et al., 2015; Wendland et al., 2007), which would improve their energy recovery potential. The system boundaries of the multi-criteria analysis excluded replacement of mineral fertilizers, which would have likely enhanced the outcome for the blackwater source-separating system in Scenario 3 as investigated in Spångberg et al., (2014).

The sustainability of specific systems depends to a large extent on the local settings and priorities set by the planners and decision-makers, and a suitable treatment facility in one context may not be sustainable in another. Soil-based systems, on-site facilities with conventional treatment options like those implemented in package plants (e.g., activated sludge), and source separation systems are also feasible options in larger scales and contexts. At larger scales, factors not addressed in this thesis, such as house density, sewer network length and topography will affect the relative suitability and sustainability of decentralised and centralised wastewater systems (Eggimann et al., 2015; Risch et al., 2021), so the scale of centralisation is also important to consider. The framework presented in Paper IV using indicators (Table 8) and scenarios (Table 9) could support the planning phase in cases such as those described in Paper V, including several larger-scale alternatives. New indicators describing the complexity of larger contexts, like the house density, elevation and sewer distance would need to be developed. Similarly, new scenarios would also help prioritization. For example, the greywater/blackwater separation case presented in Paper V (case study B) was favoured because of a very specific priority (mitigating water scarcity in the region), which led to the need to reduce water consumption (by using vacuum toilets for blackwater) and increase water reuse (by treating and reusing the greywater locally). In case study A, in contrast, a conventional on-site system designed for 1200 PE with chemical P precipitation and two sand filters was implemented (Table 5). The area’s requirement for nutrients for agriculture was not significant (as in Scenario 1, Paper IV, for example) according to the interviewee and implementing a simple, robust economic system was the main priority. Different contexts led to different choices and the outcomes were directly affected by the priorities set and the alternatives considered in the initial evaluation or optioneering phase.
5.4. What affects the implementation of alternative, resource-oriented sanitation systems? Lessons learned from past and present contexts

Since the 1990s, alternative sanitation solutions with more diversity in terms of handling specific fractions, like source separation (e.g., urine diversion), decentralised and hybrid systems (e.g., combined local grey water treatment and off-site black water treatment), have been identified as important paths in addressing pressing global challenges like rapid urbanization, ageing infrastructure, eutrophication, and climate change (Sharma et al., 2010; Brands, 2014; Capodaglio et al., 2017; Hoffmann et al., 2020; Larsen et al., 2021). Conventional centralised sanitation systems are inflexible due to high capital costs, as well as both technological and institutional lock-ins. Consequently, their “lack of adaptation” raises concerns about the resilience of the sector (Öberg et al., 2020). The inherent inflexibility of conventional centralised sanitation systems also complicates the recovery of resources, as they were originally designed for different purposes, i.e., to improve urban hygiene by removing pathogen-rich effluents from the streets, and to tackle water pollution (Hoffmann et al., 2020; McConville et al., 2017).

In Sweden, a possible “intervention point” for policy in the direction of alternative sanitation solutions appeared in the late 1980s and 1990s, because of the strong sustainability and recycling discourse that arose during the decade, and a parallel public debate which started to question the centralised sanitation approach (Paper VI). The conflicting interests between the recovery of nutrients and the containment, treatment, and risk management of the wastewater fractions, crystallized in the challenging issue of sludge management, which is still under discussion today (Stockholm Environment Institute, 2020). Even if the alternative sanitation solutions e.g., source separation systems would have worked well hygienically and technically in the early 1990s (during the possible intervention point), there appeared to be difficulties in finding farmers interested in the recovered resources, which they feared would affect their ability to sell their products. The alternative sanitation solutions were thought to lack obvious advantages over the centralised solution, even in terms of resource-recovery.

The dominance of the centralised sanitation solution in terms of adequate treatment and risk management was continuously reinforced during the period covered by the historical review (years 1974–2015, Paper VI). This was due to the long-term overcoming of many health and environmental challenges, which was continuously emphasized in the sector journal used as documentation material. At the same time, limitations of the alternative (and smaller) solutions were constantly highlighted.

Small, decentralised wastewater systems are often associated with individual on-site facilities (like those covered in Papers I-III) with ineffective treatment or old wastewater treatment plants that are approaching the end of their technical lifespan and need an upgrade. Bad experiences from the past (and present) of decentralised systems have probably influenced these perceptions, as discussed in the interview-based study (Paper V). So decentralised systems are still often regarded as examples of a bad idea (Capodaglio et al., 2017) regardless of the number of successfully implemented pilot examples (Boller,
Reasons for negative experiences include inappropriate design, ineffective operation and maintenance, as well as the fact that some systems were born technologically obsolete (Capodaglio et al., 2017). However, smaller and decentralised wastewater treatment systems are not necessarily individual on-site systems for single households. They also include systems with various degrees of semi-centralisation and satellite small-grid or hybrid systems, in contrast to grid-dominated systems (Libralato et al., 2012; Tchobanoglous and Leverenz, 2013; Hoffmann et al., 2020). An advantage of implementing decentralised technologies is that it gives planners opportunities to consider implementation of source separation system, e.g., urine separation, blackwater, greywater, or resource-saving systems with (for example) low flush/vacuum toilets to enhance resource and energy recovery. Such systems have considerable potential in rural and peri-urban areas, and could replace more than 60% of the mineral phosphorus and nitrogen fertilisers used (Lehtoranta et al., 2022). Source separation systems can also be implemented in urban settings and follow decentralised or (semi)centralised approaches (Kjerstadius et al., 2016; Skambraks et al., 2017).

A wide array of obstacles to the implementation of alternative resource-oriented sanitation has been encountered historically (Paper VI) and are still present at municipal levels (Paper V). As also discussed in previous studies, they include: financial constraints; lack of clear legislation, visions and goals, incentives, responsibility for initiating projects, and competences (experience and skills); and uncertainties associated with the immaturity of the technology (Fam et al., 2010; Brands, 2014; McConville et al., 2017a; Lennartsson et al., 2019). Steering the existing sanitation system toward change requires a radical socio-technical transition, including introduction of new technology and infrastructure, together with the development of new products and markets, user practices, regulations, institutions, and socio-cultural adaptations across diverse sectors, including the agricultural, energy, water and sanitation sectors (Fam et al., 2010). Thus, sufficiently strong drivers for such transformative change are required.

Planning including consideration of the optimal degree of centralisation of wastewater infrastructures is challenging and seldom applied in practice, which may result in more areas being connected to a central system than the economically optimal number, i.e., with costs of sewer lines and pumping exceeding the economies of scale of the WWTPs (Eggimann et al., 2015). Paper V presents two illustrative examples of planners and decision-makers choosing a different (non-fully centralised) planning approach and implementing a (semi-)local solution. In case study B, source separation of the wastewater (greywater/blackwater) was the selected option, based on concerns about local water shortages. This shows that constraints on resources (e.g., drinking water) or environmental pressure (e.g., drought and water scarcity) can prompt wastewater planners to think more broadly and include other options during their planning processes, increasing their receptivity to change. In this case, having clear goals, visions (a strong sustainability profile) and strategies initiated by politicians and driven by water professionals following a previous showcase, the H+ project in Helsingborg (Lennartsson et al., 2019), enabled materialisation of the planning of the alternative system in practice.
In contrast, difficulties in establishing a common vision and the lack of a strategic unit may significantly affect the implementation of alternative systems, as the Swedish experience shows (Lennartsson et al., 2019). In addition, having steering documents with clear guiding policies and goals included in the budgets can increase the chances of alternative systems being taken into account and prioritised, and consequently provide some leverage for the local administrations to implement them (Lennartsson et al., 2019). Case study A also illustrated the importance of engaging key persons in the evaluation phase (in which different alternatives are considered), as well as the planning and decision-making phases. Options considered in Case A included a local decentralised option, a traditional treatment option with a sand filter suggested by the consultant in charge of the project that was evaluated together with other alternatives. Key people leading change is another factor associated with receptivity to change (Cettner et al., 2014), as human factors such as commitment, personal knowledge and experience can be highly influential in the decision-making process. Removing barriers so that all solutions are considered equally seriously in order to identify and implement the most effective ones could be a good starting point to break the sector’s inertia towards adopting decentralised alternatives.

Traditionally, the wastewater sector has focused on avoiding symptoms of unsatisfactory treatment, such as eutrophication, rather than exploiting the opportunities to provide benefits such as resource recovery and the promotion of circularity. The problems we are facing in urban water management stems from mixing and dilution (Larsen et al., 2015) apart from the entrenched and locked-in linearity of the current paradigm in the sector. A change of this paradigm seems necessary to address the new challenges the sector is facing, in terms of water (re)use, recovery of nutrients and energy, and removal of micropollutants. Using a systems-approach (systemic perspective) rather than the traditional end-of-pipe approach could support the transition towards more resilient water systems while accounting for their complexity.

5.5. Practical implications of the findings

Planning sanitation systems requires understanding the water sector’s role in addressing current global challenges like urbanisation, climate change, environmental degradation, depletion of resources and geopolitical instability. Interdisciplinary approaches are needed to tackle these challenges and comprehend the interconnections between the diverse sectors that contribute to urban/rural metabolism (Larsen et al., 2016; Wielemaker et al., 2018; Hoffmann et al., 2020).

Existing Swedish Environmental Code (Swedish government, 1999) clearly states that its purpose is to promote sustainable development, and that “reuse and recycling, as well as other management of materials, raw materials and energy are encouraged with a view to establishing and maintaining natural cycles”. The wastewater sector plays an important role in maintaining natural cycles as it deals with key resources like water, nutrients and energy. The responsibility of the wastewater utilities is therefore to provide adequate sanitation
services while managing the resources as sustainably as possible, by closing the loops as far as reasonably possible. However, the characteristic linearity of the current system is not compatible with a circular approach.

Local authorities issuing permits for installation of on-site sanitation systems should implement existing legislation and favour systems that promote closure of the nutrient loops and have lower climate footprints. Source-separating solutions like blackwater and greywater separation and urine diversion can contribute in this respect as they can effectively separate major nutrients (to be treated elsewhere) and have lower climatic impacts than (for example) package plants (Paper IV). Soil infiltration systems (with additional P-treatment steps) can be robust options, but do not allow high resources recovery and do not tackle nitrogen discharges into the receiving environment. For larger settings, including development areas, small communities or clustered houses, assessment of the optimal level of centralisation of wastewater treatment (including consideration of economic, geographic and environmental factors) should be encouraged during the evaluation or planning phase. Sometimes, solutions other than centralisation can be built locally, including robust systems that can be operated and supervised remotely without much operational input or maintenance (Libralato et al., 2012). Further work on source-separating systems is also needed to clarify the scales and contexts in which these systems can be considered economically and environmentally viable alternatives (McConville et al., 2017a). Wastewater planners need to think more broadly and include other options during their evaluation and planning processes, generating experience and knowledge, and increasing the receptivity to change in the water sector. This could be encouraged by attending educational courses and conferences, cooperating with universities and research centres and involvement in pilot studies. Current source-separating systems may not be the solutions of the future, but by investing resources and stimulating new ideas, new collaborations and piloting we will succeed in transitioning to a more sustainable and circular system (McConville et al., 2017a).
6. Conclusions

More than half of the on-site sanitation facilities inspected in the research underlying this thesis were not suitable for sampling (especially flow-proportionally), for construction or design reasons and indicating that it is not possible to investigate the effluent quality and treatment efficiency of many existing systems.

The investigated sand filters generally had lower effluent BOD values, P concentrations, and densities of indicator bacteria than the package plants. Effluents of package plants were more variable than those of sand filters with alkaline P filters, which had lower total-P concentrations. Insufficient sedimentation, lack of thorough mixing and ineffective dosing in facilities with chemical P precipitation, together with clogging and exhaustion of the alkaline P-filters, were common issues affecting the systems’ treatment efficiency. Adequate operation and maintenance would likely be sufficient to ensure high P-removal rates. Densities of indicator bacteria in package plants’ effluents nearly always exceeded the thresholds set in the EU bathing water directive for excellent bathing water quality (of inland waters), in contrast to effluents of sand filters with an alkaline P-filter.

Nitrogen removal in the investigated package plants was generally low and mainly mediated by sedimentation of particulate organic nitrogen. Only two facilities achieved high nitrification rates (>51%), especially during the warmest periods. The low annual average temperatures (8±4°C) likely prevented high nitrification rates. Denitrification was observed, to a limited extent, in two facilities that included water/sludge recirculation of the nitrified N to the septic tank.

An expected significantly negative effect of cold climatic conditions on the treatment efficiency of package plants was not confirmed, although reductions in BOD, nitrogen and indicator bacteria levels appeared to be lower in some facilities (e.g., SBR1, SBR2) when the wastewater temperature was lowest. The lowest wastewater temperatures coincided with the snow melting period, rather than when the air temperatures were coldest (about -30°C).

Package plants may be sources of micropollutants to the receiving environment. Concentrations of the pharmaceuticals measured in the effluents were within or higher than ranges previously reported in effluents of conventional WWTPs, but those of phthalates were below previously reported effluent concentrations.

In terms of sustainability performance, sand filters and soil infiltration systems were found to be more sustainable than package plants according to the multi-criteria analysis, because of their higher robustness and social acceptance, and lower costs and LCA impacts. When removal of nutrients (both P and N) and recycling were prioritized (Scenario 2), source-separating systems like blackwater/greywater and urine diversion were the most sustainable options as they enable closure of nutrient loops to a greater extent than the conventional systems.

Centralising the water systems by building long pipe networks and investing in upgrades of existing main treatment plants (end-of-pipe technology) was found to be common
practice and a trend across Sweden. Limitations of resource (e.g., drinking water), environmental pressure (climate change), key leadership and experience, clear goals, common visions and strategies can all promote shifts in planning sanitation systems, favouring (for example) source-separating alternatives or decentralised local systems. However, the drivers need to be strong enough to break the inertia and create change in a sector characterised by strong locks-in.

Historic barriers to the implementation of alternative systems with higher resource recovery capacities are still present at municipal levels, including the lack of incentives, initiative and competences, and the uncertainties associated with immaturity of the technology and interpretation of the regulation. Sweden has played a pioneering role in the development of resource-recovery sanitation solutions. However, even in Sweden the limitations of source-separating and decentralised systems have been highlighted in recent decades, while the dominance of the wastewater management centralisation approach, offering advantages in terms of treatment and risk management, has increased.

Overall, the results presented in this dissertation contribute to better understanding of small sanitation systems’ treatment processes, efficiency, sustainability and implementation. At a larger scale, the thesis also provides information about the Swedish wastewater sector’s historical development and current heuristics that characterise the planning and decision-making phase, with views to move towards more circular models of resource management.
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Phosphorus reduction in filters for on-site wastewater treatment

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Phosphorus reduction in filters for on-site wastewater treatment

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A B S T R A C T

Discharges of phosphorus (P) from on-site wastewater treatment systems generally contribute to eutrophication problems in Swedish freshwaters and, ultimately, in the Baltic Sea. Such concerns have led to a growing interest in improving P removal in treatment facilities. This study investigated the reduction of P in 12 full-scale on-site treatment systems featuring sand filters and alkaline P-filters by sampling and analysing filter influents and effluents. The flow-proportional samples collected were analysed for total and dissolved P, BOD₇, total and dissolved organic carbon (TOC and DOC), and pH. Seven of the eight investigated sand filters did not provide satisfactory total P reduction; the likely explanations are filter clogging and wastewater dilution by extraneous water. In addition, effluents from four of the eight sand filters had total P concentrations higher than 3 mg L⁻¹, which is the Swedish effluent limit recommended for common receiving waters, indicating that a subsequent polishing step would be needed. Six of the nine investigated P-filters reduced P adequately, with total P concentrations in the effluent ranging between 0.1 and 1.9 mg L⁻¹. The three underperforming P-filters had effluent pH values below 9; filter age, clogging, and hydraulic overload were identified as probable reasons for their poor performance. A statistically significant correlation was found between total-P reduction and filtrate pH, but no significant correlation was detected between organic load in the influent and P reduction by the P-filters. The P-filter media replacement frequency could not be established, but filtrate pH appeared to be a good estimator.

1. Introduction

Eutrophication, caused by excess nutrients such as nitrogen (N) and phosphorus (P) in water bodies, is acknowledged as the single largest environmental problem in the Baltic Sea [1]. All Swedish waters—including lakes, rivers and coastal waters—have been identified as sensitive to P discharge [2]. Moreover, the gross discharge of P from private small-scale wastewater systems (Septic systems) in Sweden is as large as the discharge from all municipal wastewater treatment plants together, accounting for 15% of the total anthropogenic release of P [3]. The Swedish regulation for small-scale wastewater facilities (< 25 person-equivalents) consists of a set of guidelines recommending certain reductions for P, N, and organic carbon measured as BOD₇ [4]. The suggested criteria are 70 and 90% removal of total P (tot-P) for common and highly sensitive receiving waters, corresponding to tot-P effluent concentrations of less than 3 and 1 mg L⁻¹, respectively. However, it is estimated that less than 50% of the approximately 700,000 existing facilities provide adequate treatment and comply with these guidelines [5,6]. Thus, better understanding of how much P these facilities remove and how long the filters maintain their efficiency after installation is needed.

On-site wastewater treatment facilities in Sweden usually consist of primary treatment in a septic tank followed by secondary treatment; usually either soil-based systems such as sand filter beds or drain fields, or package treatment plants. Both biological and physicochemical processes can occur in secondary treatment. Additionally, where required, a tertiary treatment with an alkaline filter (P-filter) can be added as a final polishing step to remove P. The effluent water is discharged into the subsoil or into receiving water bodies (e.g., ditches, streams, lakes or the sea).

Extensive reviews of P removal and recovery technologies [7] and specifically of reactive filter materials for P sorption [8] have highlighted the potential benefits and development of P removal and recycling. Removal of P in P-filters and sand filters is influenced by the hydraulic and organic load of the incoming wastewater [9,10] and the physical and chemical properties of the filter material, e.g. mineralogy, grain size and pH [10–12]. Biofilm growth and clogging may occur and diminish the efficiency of the filter material. The efficiency of sand filters with respect to hydraulic and biological behaviour such as biofilm development [10], clogging mechanisms [13] and performance under cold-weather conditions [14,15] has been previously studied in laboratory and pilot-scale tests and field conditions [16]. Materials used in P-filters for on-site treatment have been also studied in laboratory-

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2. Materials and methods

2.1. Investigated facilities

Twelve on-site wastewater treatment facilities (A–L) were investigated (Table 1): A–C (sand filters), D–H (sand filters with P-filters) and I–L (P-filters). Four of the P-filters were installed in small package plants and five were installed downstream from sand filters. The remaining three sand filters did not have any downstream treatment.

The sand filters in facilities A, D and E were built according to Swedish standards and consisted of a 80 cm layer of filter media (sand and gravel, particle sizes 2–8 mm) with a design load of 30–60 L m⁻² d⁻¹ and a typical surface area of 25 m² [22]. The lifespan of sand filters has been estimated to 15 years [23] however, it is often used for 20–30 years [6]. The wastewater was distributed through slotted pipes and filtered vertically through the filter bed to the bottom of the system, where it was collected in drainage pipes. The sand filters in facilities B and C were smaller in size than the Swedish standard and contained plastic crates (biomodules) to increase their hydraulic capacity. Sand filters F, G and H used a slightly different construction method than the standard, and had a layer of drainage baskets on the top (biomodules) with a triangular cross section, about 0.5 m wide, where the distribution pipe was located. All sand filters were covered with an approximately 30 cm thick layer of soil.

The nine P-filters consisted of bags filled with P-sorbing material. In eight of them, the filter media Polonite was used (supplier: Ecofiltration AB, Sweden), placed in a plastic container installed at ground level, and operated in down-flow (D) or up-flow (F–L) mode. In facility E, the filter media Filtra P (supplier: Wavin-Laboko Ltd) was used and placed in two chambers installed at different levels so the wastewater percolated downwards through the filter media in the first chamber and upwards in the second chamber. Facilities I, J, K and L had P-filters installed inside package plants that functioned as a pre-treatment for the P-filters. Alkaline P-filters should be replaced after 2–3 years of use, according to the producers [24].

Facility I was a package plant (4evergreen by Biorock®) consisting of a trickling filter made of fibre material installed inside a HDPE tank where the wastewater percolated through. Facilities J and K were Biop® plants that consisted of a PE unit with multi-stage biological treatment based on attached growth with no aeration and facility L was an aerated activated sludge system installed inside of a HDPE unit (Ecobox Small by Ecotech AB).

2.2. Water sampling and analyses

Sampling was carried out between September and October 2015 and between May and August 2016 (Table 2). For each facility, grab samples were taken in the third chamber of the septic tank or from the distribution/pumping box where the wastewater was transferred to the sand filter. Flow-proportional samples were taken from the outlets of the sand filters, package plants and the P-filters. The samples were taken at different times of the day over a period of several hours, depending on the specific flow (it varied greatly between facilities) and operated in down-flow (D) or up-flow (F–L) mode.
practicalities regarding equipment and location. Flow was measured manually (volumetric flow measurements), using effluent flow for the sand filters and either influent or effluent flow for the P-filters. At each sampling event, two composite samples of approximately 2 to 3 L (depending on flow) were taken from the effluent water collected from the outlets of the sand filters, package plants and P-filters over a period of 1.5–2 h approximately.

Total suspended solids (TSS, not measured on all occasions) and pH were measured on-site. The pH was measured using a WTW pH330 pH meter with a WTW SenTix41 pH electrode. TSS was determined following the European standard EN 872:2005 [26]. The samples for determining total and dissolved P (dis-P), total and dissolved organic carbon (TOC, DOC) and biological oxygen demand (BOD7) were frozen and stored for later analysis in the laboratory. Samples for DOC and dis-P analyses were filtered through 0.45 μm filters before freezing.

BOD7 was analysed according to the European standard method CSN EN 1899-1 (modified) [27]. Phosphorus was analysed with a Quattro spectrometer and the device-specific method number A-031-04, according to European standards (ammonium molybdate method) with digestion (persulfate oxidation, SS-EN 1189 performance 6.4) [28]. IR detection (based on CSN EN 1484, CSN EN 16192, and SM 5310) was used to analyse TOC and DOC [29,30].

The overall reduction of tot-P in the 12 treatment facilities was calculated based on P concentrations measured in the last chamber of the septic tank and the effluent from the final treatment step (sand filter or P-filter); hence, the estimated P reduction of the system did not include treatment in the septic tank.

2.3. Tracer tests

In three P-filters (H, J, K) the hydraulic residence time was estimated with tracer tests by using an optical monitoring sonde (YSI 600OMS V2) with a rhodamine probe (YSI 6130). A rhodamine water tracing (WT) solution with a 70 μg L⁻¹ concentration made from 20% rhodamine WT concentrate was used. Estimations of the required volume of tracer solution were based on the volume (0.7 m³) and density (730 kg/m³) of the filter media in the P-filter bag. The 0.5 L of the rhodamine WT solution was added to the inlet pipe of the P-filters, and the rhodamine WT concentration was then measured in the filter effluent. The sonde was left in unattended mode collecting data (concentration of rhodamine WT measured by the optical sensor) over one (J), two (K) and six (H) days in December 2016. The rhodamine WT concentration was measured every two minutes (J, K) and five minutes (H). The data was processed based on time of the first arrival.

2.4. Statistical analysis

The concentrations of tot-P, dis-P, TOC and DOC as well as the pH, were weighted for the water flow. The flow-proportional arithmetic means (x̄ w) and standard deviations (s) were calculated according to Eq. (1) and (2), respectively.

\[ x_w = \frac{\sum_{i=1}^{n} w_i x_i}{\sum_{i=1}^{n} w_i} \]  

\[ s = \sqrt{\frac{\sum_{i=1}^{n} (x_i - x_w)^2}{\sum_{i=1}^{n} w_i - 1}} \]

Where:
- \( x_w \) = flow-proportional arithmetic mean
- \( n \) = number of observations
- \( x_i \) = measured concentration
- \( w_i \) = volume of wastewater making up the composite sample
- \( s \) = flow-proportional standard deviation

It was assumed that the influent flow rate to the filter was the same as the measured effluent flow rate and vice versa when the influent flow rate was the one measured instead.

The software Minitab® [31] was used for the statistical analyses. Spearman’s rank-order correlation was used to analyse the nonparametric data (normality test Anderson-Darling applied).

3. Results and discussion

Fig. 1 shows the average concentrations of tot-P measured in the effluent from the septic tanks, sand filters, package plants and P-filters in the studied facilities (A–L). The concentrations of tot-P in the septic tank effluent varied greatly among the study facilities, ranging between 6 and 29 mg L⁻¹ (Fig. 1), possibly as a result of the different water consumption patterns and contribution by different sources of P in each household such as human waste and the extent of synthetic detergent use. Concentrations ranging between 1.2 and 21.8 mg L⁻¹ are normally found in septic tank effluents as reported in a U.S. study [32] modelling the fate and transport of nutrients from on-site wastewater systems.

3.1. Phosphorus reduction in the sand filters

Four (B, F, G and H) of the eight investigated sand filters (A–H) had effluent concentrations for tot-P higher than 7.5 mg L⁻¹ (Fig. 1),
Thus, good total P reduction in a sand filter—in facility F—was almost 15 mg L$^{-1}$ (Fig. 1).

The highest total P reduction was observed in sand filters D and H, showing a 62 and 70% reduction, respectively (Fig. 2), which is acceptable according to current Swedish guidelines (although both sand filters were quite different with respect to the influent P load). The total-P concentration in the effluent from sand filter D was 1.8 mg L$^{-1}$, which is below the recommended value of 3 mg L$^{-1}$, whereas in sand filter H the effluent concentration was still high (10.6 mg L$^{-1}$) despite the good P reduction capacity of the sand filter. This was caused by the very high initial concentration of total-P in the septic tank effluent (29.3 mg L$^{-1}$). Thus, good total-P reduction in a sand filter does not necessarily translate into an acceptable total-P concentration in the effluent.

Low P reduction was observed for three sand filters (B, F and G). In sand filter B (which had been in operation for three years) the measured water flow through the filter was low (3.2 L h$^{-1}$), with a high suspended solids content at the outlet (TSS = 17 mg L$^{-1}$), high bacterial concentrations [25], dark colour and strong odour, indicating that this filter was clogged. The main reasons causing clogging are high concentrations of extracellular bacterial slimes in the sand pores, accumulation of suspended solids or precipitation and deposition of compounds such as calcium carbonate [13,33]. Moreover, clogging can persist and increase when ponding of wastewater occurs between the batches [34]. Neither the clogging characteristics nor any obvious malfunction were observed in sand filters F and G (five and three years in operation, respectively). Their inefficiency could be explained by the chemical composition of the sand used. Gill et al. [35] discussed the clear relationship between P removal in subsols and soil mineralogy after studying on-site wastewater effluent discharge in six sites in Ireland. At pH > 6, as observed in the influent and effluent of the sand filters F and G in this study, the P-removal capacity of the sand is dominated by a combination of physical adsorption to iron (Fe) and aluminium (Al) oxides and precipitation of P as sparingly soluble calcium phosphates [11,35]. According to Eveborn et al. [36], mechanisms for P removal in sand filters were explained by the strong relation between oxalate extractable P and Al. Hence a lack of Al (or Fe) compounds in the sand filters F and G could have hampered the adsorption and mineral precipitation of P. The distribution of wastewater in the filter bed could also affect the P removal, as uneven distribution has been shown to promote preferential flows and decrease the potential to remove P in sand filters [37,38].

At least three (A, C, E) of the eight investigated sand filters had likely dilution problems, i.e. extraneous soil/ground/rainwater infiltrating into the filter bed and diluting the wastewater. Dilution was suspected to occur based on field observations, such as high measured water flow through the sand filters, even when the users were not at home and when the pump transporting wastewater from the septic tank was not operating during collection, as well as very clear effluent water. In these facilities, the low P concentration could therefore be explained by the dilution of the phosphorus concentration and not the filters’ efficiency. For this reason, correct installation and supervision of sand filters are crucial and highly recommended to ensure that the filter material is properly sealed and there is no infiltration/exfiltration between the treatment facility and the surrounding environment.

P reduction occurring in the septic tank was not investigated in this study, but this reduction has been estimated to be 15 ± 10% of the incoming quantity [5]. Assuming this percentage of reduction in the septic tank, facilities B, F and G, whose sand filters showed low P reduction, would be estimated to reach only 33–39% total-P reduction, which is still below the recommended reduction of 70% suggested by the Swedish guidelines. The low P reduction capacity of sand filters had already been shown in previous studies; e.g. Eveborn et al. [36] reported 8–16% P removal in four sand filters by using a mass balance approach, while Vilpas and Santala [20] reported low performance from two conventional sand filters, with total-P concentrations in the effluent exceeding the Finnish regulation of 3 mg L$^{-1}$. A similar trend was shown by Wilson [39] who investigated the long-term effects of filter length and wastewater loading on performance of eight sand filters in Canada. In the mentioned study, the P removal decreased with time as adsorption sites in the sand filters became exhausted. The estimated removal of total-P in sand filters in Sweden, including reduction in septic tank, is 50 ± 30% [5], which is a wide range that reflects the expected large variations in removal efficiency. Sizeable variation in sand filter performance and their general low P reduction was corroborated in this study.

The four package plants (I–L) reduced P to a lesser extent than the sand filters. The influent total-P concentrations ranged between 7.5 and 11.6 mg L$^{-1}$ of total-P (Fig. 1), and the highest reduction rate in the package plants was only about 11% (facility K) (Fig. 2). In contrast to the sand filters, these units have not been designed as stand-alone treatment systems but as a pre-treatment step before the P-filter units to reduce organic matter. Hence residence time is short and P treatment capacity is low.

3.2. Phosphorus reduction in the P-filters

Six out of the nine investigated P-filters generally reduced P satisfactorily and covered for a considerable amount of total-P removed in
average concentrations). B. Vidal et al.

much higher in...functioned well ranged between 0.15 and 1.9 mg L\(^{-1}\) (Table 3). This suggests that the...t- and dis-P measured in the influent and effluent of the P-filters. The two other P-filters with low reduction (J, K) were highly loaded (\(J = 79.4\) L m\(^{-2}\) h\(^{-1}\); \(K = 58.5\) L m\(^{-2}\) h\(^{-1}\)) because of the...measure to P. P- filters that worked well and had low P effluent concentrations below 3 mg L\(^{-1}\)–namely D, F, G, H, I and L–had varied...D, F, G, H, I and L–had varied influent tot-P concentrations. For example, P-filter D had already low concentrations of tot-P in the influent (1.8 mg L\(^{-1}\)), while the rest had much higher influent concentrations, ranging from 7.5 to 14.8 mg L\(^{-1}\) (Table 3). This suggests that the filter is able to function adequately and remove most of the tot-P regardless of influent concentration within this range (14.8–1.8 mg L\(^{-1}\)).

Three P-filters (E, J and K) did not remove P satisfactorily; J and K had the highest average effluent concentration of tot-P, 3.04 and 4.4 mg L\(^{-1}\), respectively. P-filter E was possibly saturated with respect to P and, because of its construction, some of the influent water bypassed it without passing through the filter material. The pH in the effluent was very low (around 6), and almost no P was reduced. P-filter E was the oldest P-filter in the study (built in 2009) and was more than six years old by the time the samples were taken. The filter medium used was Filtra P, in contrast to all the other investigated P-filters, which contained the filter medium Polonite. Both filter media have been reported to have significant potential for P removal, although Filtra P has been shown (in column experiments) to be prone to clogging due to structural degradation of the material, which in practice translates into a shorter service life [40].

The pH in the effluents of both filters was below 9, although the filters had been installed only around two weeks before sampling started. The low pH could be explained by the washing out of calcium from the filter caused by the high hydraulic load, as previously studied by Herrmann et al. [9]. Furthermore, the high hydraulic load corresponds to short residence time, so that there was probably not sufficient time for the calcium phosphates to precipitate. Another possible explanation would be differences in the pore distribution and the formation of preference channels, which can also affect the sorption efficiency of P-filters, as discussed in [41]. The organic matter concentrations in the influent to the P-filters in facilities J and K were in the same range as for most of the other P-filters with high P reduction rates (Table 3), ruling out high organic matter influent concentrations as a reason for the low P reduction in facilities J and K.

### 3.2.1. pH dependency of phosphorus reduction in the P-filters

The effluent pH of the P-filters ranged between 6.0 and 12.1 (Table 3). The P-filter with the highest tot-P reduction (F) was also the one with the highest effluent pH (12.1) (Fig. 3). Significant correlation was found between the P reduction in the P-filters and the pH measured in their effluents (Spearman rank correlation coefficient = 0.833, \(p = .005\)). Higher pH values favour the precipitation of calcium phosphate [42], which is the filters’ main retention mechanism [40].

In previous column studies, [12] found a positive correlation and a strong relationship between the P removal in Polonite filters and pH. Nilsson et al. [43] reported the same positive correlation, but only when the P-filters were loaded with high BOD concentrations (120 ± 11 mg L\(^{-1}\)). In the same study, no significant correlation was found between the P removal and pH when the Polonite filters were

![Fig. 3](image_url) Relationship between the reduction of tot-P in the P-filters and pH measured in the effluent of the nine P-filters studied (D–L).
loaded with low BOD concentrations (20 ± 5 mg L\(^{-1}\)), which is in the range of the corresponding data in the present study (Table 3). However, the positive correlation between pH and P removal in P-filters with influent BOD concentrations ranging from 1.9 to 31.4 mg L\(^{-1}\) was confirmed in this study.

The P-filters with an influent pH of 9.4 or higher had 75% or more tot-P reduction (Fig. 3). However, pH is not the only factor affecting the P reduction in the P-filter, and several facilities showed different rates of P reduction despite similar pHs. For example, in P-filters J and K, with similar pH values of 8.6 and 8.8, respectively, residence times were estimated to be considerably different (Fig. 4), and their P-reductions differed by approx. 16 percentage points. This indicates that the residence time of the wastewater in the P-filter is an important factor governing P removal. An earlier comparative study of various filter materials by Cucarella and Renman [44] concluded that low effluent pH values indicated low P sorption, but higher pH values did not necessarily imply greater P sorption. These findings are in line with the results of the present study when comparing the performance of the P-filter in facility G with the other facilities with influent pH > 9: e.g. facility G had higher effluent pH than facilities (D, L, I and H), but lower P reduction (Fig. 3).

### 3.2.2. Effect of organic load on phosphorus reduction in the P-filters

The TOC concentrations in the influent to the P-filters were rather low, ranging between 8 and 40 mg L\(^{-1}\) TOC (Table 3). No significant correlation was found between P reduction in the P-filters and the TOC in the influent to the filters (Spearman rank correlation coefficient = 0.150, \(p = .700\)). Previous studies on the effect of organic matter content in wastewater on P removal in P-filters have shown that higher P removal is achieved when extensive pre-treatment of the wastewater occurs [12,43]. A study with batch experiments [42] found that humic substances negatively affected phosphate removal efficiency when the pH of the solution was 8 but had almost no effect when the solution had a pH of ≥9. The inhibitory effect of the humic substances on phosphate precipitation was explained by the combination of calcium and humic substances and by the blocking of active growth sites on newly nucleated precipitates of calcium phosphate. Similarly, the lack of correlation between P reduction and influent TOC in the present study could be explained by the fact that effluent pH from only one facility (E) was low (6.0) (Table 3), while effluent pH for the other eight facilities ranged between 8.6 and 12.1. Hence, for the statistical analysis, the inhibitory effect of organic matter on P precipitation might have been small because most of the facilities had a high pH.

In column experiments, [43] reported a significant positive correlation between TOC removal in Polonite filters and pH concentration in the effluent, for low concentrations of BOD (20 ± 5 mg L\(^{-1}\)). However, this was not confirmed in the present study, which found no significant correlation (Spearman rank correlation coefficient = 0.133; \(p = .732\)) when comparing effluent pH and TOC reduction in the P-filter.

### 3.2.3. Estimating hydraulic residence time in the P-filters

Tracer tests were carried out to estimate the arrival times of the influent water in three P-filters (H, J and K). In P-filter H, the rhodamine WT was detected in the effluent after approximately 5.5 days (Fig. 4). In P-filters J and K, the rhodamine WT was detected in the effluent after 1.5 h and 4.5 h, respectively (Fig. 4).

The water flow was not measured during the tests, but average flows from previous sampling campaigns were 5.3, 39.9 and 29.4 L h\(^{-1}\) in P-filter H, J and K, respectively. In P-filters H and K, the end transit pulse of the rhodamine was not reached, hence no estimations of the flow rate were made. However, the end transit pulse of the rhodamine was reached in P-filter J. The peak concentration of rhodamine was achieved after approximately 80 min, decreasing asymptotically afterwards. The calculated average flow for this facility, based on the measured rhodamine concentrations and assuming complete recovery of the rhodamine WT, was 0.23 L h\(^{-1}\). The twofold difference between
the measured flow during the sampling campaign (39.9 L h⁻¹) and the calculated average flow (0.231 L h⁻¹) may be due to several reasons. Explanations include the lack of pumping cycles during night hours, the uncompleted mixing of rhodamine with the influent water which would, consequently, affect the detection and/or the presence of holes or cracks in the bag that would allow the water outflow without being detected by the sensor. The oscillations observed in the graph for P-filter K may show the system’s pumping cycles.

The observed time of arrival was inversely related to the previously measured wastewater flow, although the flow measurements were carried out during limited time periods (Table 2). Higher P reduction occurred in the filter with the longest arrival time (H), and lower reduction in the filter with the shortest arrival time (K), possibly because high loading rates have washed out more P precipitates or reactive calcium ions, and the short residence time was insufficient to form calcium-P precipitates.

Previous column experiments with Filtralite P⁺ have shown that a short residence time (15 min when wastewater was used) was sufficient for P removal in P-filters [9]. However, more time might be needed for P that has already reacted to be retained in the filter material. The short residence time in P-filters J and K and the higher flows may possibly have had a negative effect on the binding capacity of the P-filters.

3.2.4. When and how often should P-filter media be changed?

According to the manufacturers of Polonite (Ecofiltration Nordic AB), P-filters are built to maintain high P reduction during the first 2–4 years, depending on load, with decreasing reduction during the remainder of the filters life cycle. Based on data obtained and the field observations in this study, it is difficult to determine how often the P-filters should be changed. Both old (E, 6 years) and new P-filters (J, K, 2 weeks) showed low P reduction, but for different reasons; age and clogging were assumed to account for the poor performance of P-filter E, while high hydraulic load hindered the performance of P-filters J and K. The pH in the effluent to all these P-filters (E, J and K) was below 9, suggesting that pH is a good indicator for estimating filter performance.

P-filters showed high reduction rates, both when the influent P concentrations were high (e.g. facility F, 14.78 mg/L) and when they were low (e.g. facility D, 1.80 mg/L), indicating that the influent P concentration is not decisive for their functioning, at least for the concentration range covered in this study. A review of sorption filter materials [44] found a clear tendency for higher P sorption capacity of the materials investigated with increasing initial P concentration.

However, the studies selected in the review [44] included a wide variety of materials, such as blast furnace slag, fly ash, Filtralite P⁺ and Shell sand, and the initial P concentrations ranged from normal values of 0–30 mg P L⁻¹ to very elevated concentrations of 10,000 mg P L⁻¹.

Pratt et al. [45] argued that the complex relation between the P-filters lifespan and the hydraulic retention time (HRT) makes it difficult to estimate the filters lifespan, as flow dynamics, weathering reactions and removal mechanisms must be taken into account. Continuous-flow conditions are needed to predict the long-term performance and lifespan of the P-filters, conditions that are often not met in full-scale systems.

4. Conclusions

This study evaluated the reduction of P in 12 full-scale on-site wastewater facilities with two general types of filters used individually and in-combination—sand filters and P-filters—and compared the effluent concentrations of tot-P to current Swedish guidelines.

Only one sand filter (D) of four was confirmed to remove P satisfactorily (effluent concentration below 3 mg P L⁻¹). The inefficiency of the four sand filters that did not function adequately (discharged more than 3 mg P L⁻¹) indicates that a downstream treatment step is needed. The remaining three sand filters were suspected to have dilution problems. This means that correct installation and proper sealing of the filter must be ensured to prevent mixing wastewater with other water sources in order to achieve adequate P reduction within the treatment unit.

Six out of nine of the investigated P-filters generally removed P well (75–99% P reduction) and accounted for a considerable amount of tot-P reduced in the treatment facilities, with the exception of one filter that was old and clogged, and two other filters that had to cope with high hydraulic loads.

The analysis found a significant positive correlation between P reduction in the P-filters and pH measured in the effluent, indicating that pH could be used as an indicator of P-filter efficiency. In the three P-filters that worked insufficiently, with effluent concentrations higher than 3 mg P L⁻¹ (J, K) or 0.7% tot-P reduction (E), when pH was 8.8 or less.

No significant correlation was found between the reduction of P in the P-filters and the TOC measured in the influent to the filters. It may be that the low incoming TOC concentrations and the high pH found in the majority of the P-filters prevented this effect from reaching quantifiable levels. With the low incoming TOC, the inhibitory effect of organic matter on P precipitation was probably small.

It was difficult to determine how often the filter media in P-filters should be changed, possibly due to the variation in the data in this study and because the filters included in it were rather new (except for P-filter E). With proper installation, the P-filters in the study that handled a moderate load, had a pH above 9.4 and not too many years of use (2–4 years as suggested by manufacturers) functioned well and reduced the incoming concentration of tot-P to below 1.9 mg P L⁻¹.

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References


Discharge of indicator bacteria from on-site wastewater treatment systems

Desalination and Water Treatment

Discharge of indicator bacteria from on-site wastewater treatment systems

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ABSTRACT

Small-scale on-site wastewater treatment facilities present the risk of microbial pollution of groundwater used for drinking water and surface water used for recreational purposes. This study assessed the discharge of indicator bacteria, total coliform, Escherichia coli, intestinal enterococci and Clostridium perfringens, by flow-proportional sampling from 12 full-scale on-site treatment systems featuring biological treatment units (mainly sand filters) and alkaline filter beds for phosphorus treatment (P-filters). Correlations of effluent bacterial concentrations with pH, total and dissolved organic carbon, filter age and hydraulic load were evaluated. The bacterial concentrations in the effluents of the sand filters were considerable. The concentrations for excellent bathing water quality set in the EU bathing water directive, 200 and 500 colony forming units (cfu)/100 mL for intestinal enterococci and E. coli, respectively, were exceeded in three (intestinal enterococci) and one (E. coli) of the eight investigated sand filters. In one of the sand filters, effluent E. coli concentrations were high although no obvious malfunction of the filter was observed. In the effluent from the other investigated biological treatment units (a trickling fibre filter, two units with attached growth treatment and one aerated activated sludge technique), bacterial concentrations were very high (75,000 cfu/100 mL of Clostridium perfringens and 85,000 cfu per 100 mL of total coliform), possibly because of a shorter retention time of the wastewater in these facilities, missing aeration and little time between start-up and measurements. Three and four of the nine investigated P-filters exceeded excellent bathing water quality in coastal waters as stipulated by the EU bathing water directive in respect of E. coli and intestinal enterococci, respectively.

Keywords: Sand filter; Alkaline filter; Polonite; Total organic carbon; pH

1. Introduction

In Sweden, there are large areas of land that are sparsely populated, where about 11% of households are not connected to municipal sewerage [1]. On-site wastewater treatment is, therefore, widely used, potentially causing hygiene problems. Previous studies in different countries have shown that microbiological contaminants from on-site wastewater treatment impact the groundwater [2,3] and are a possible reason for elevated bacterial levels in streams [4–7]. Therefore, the discharge of pathogens from Swedish on-site wastewater treatment facilities is potentially problematic, because of the risk of contaminating the groundwater used for drinking water in rural areas and the surface water used for recreational purposes, especially in areas with summer houses.

On-site wastewater treatment facilities in Sweden typically consist of a primary treatment in a septic tank as well as a secondary treatment, achieved with soil-based systems such as sand filters or in mini-package treatment plants. To prevent eutrophication of the receiving water, the reduction requirements for phosphorus (P) are between 70% and 90%,
depending on the protection status of the area in question [8]. To comply with these regulations, many rural on-site facilities have recently been upgraded with a tertiary treatment unit, in the form of a filter bed that traps P (P-filter). The effluent from these facilities is either discharged to soil (infiltration) or, in many cases, directly discharged to receiving water bodies such as ditches, lakes, the sea or groundwater.

In previous pilot studies, the removal of bacteria from wastewater using sand filters was shown to be high, for example, in gravel-filled unplanted constructed wetlands [9] and sand filters treating primary effluent [10]. However, the treatment efficiency has been shown to depend on the filters’ age [11] and effluent characteristics [12]. Bacterial discharge from full-scale sand filters has not yet been investigated. However, many on-site wastewater treatment systems, such as drain fields, distribute the wastewater to the ground for secondary treatment (instead of using constructed sand filters) and a number of studies have, therefore, dealt with bacterial reduction in soil [13–15]. Bacterial reduction in mini-package plants has rarely been studied. Results from grab sampling presented in a Swedish study indicated differing bacterial reduction in three mini-package plants (not including the ones investigated in this study) [16].

P-filter media and P-filters have been extensively studied in laboratory-scale tests and pilot tests [17,18], but there have been only a few studies dealing with the reduction of microbiological contaminants. Many filter materials used in P-filters are alkaline, thus increasing the wastewater’s pH so may therefore also help to decrease the bacterial content of the effluent. A pilot experiment by Nilsson et al. [19] showed the pH in the filter media Polonite to range from 9 to 12.3 and indicated a reduction of enterococci ranging from 52% to 91%. In a study with a pilot-scale P-filter using the filter media Filtra P, high log removal rates of Escherichia coli, enterococci and clostridia were measured [10]. Grab samples from full-scale P-filters using the filter media FiltraLite P® indicated very low effluent bacterial concentrations [20]. Another FiltraLite P® system completely removed bacteria during the first 3 years of operation, but removal efficiency decreased after the system’s design capacity was exceeded [21].

In summary, the discharge of pathogens from on-site wastewater treatment facilities has not been comprehensively studied. In particular, facilities that discharge directly into receiving water bodies (instead of infiltration into soil), such as sand filters and mini-package plants, have been overlooked. Microbiological contaminant removal in sand filters has only been investigated at pilot-scale using grab sampling. The full-scale studies available focus on bacterial removal in soils. Bacterial removal by the P-filter media Polonite, which is widely used in Sweden, has not been investigated at full-scale. Therefore, the aim of this study was to investigate the discharge of indicator bacteria from full-scale on-site biological treatment units (mainly sand filter beds), and alkaline P-filters (mainly Polonite filters) using flow-proportional sampling. Knowledge about the discharge of indicator bacteria might help assessment of the potential risk of microbial pollution of natural water bodies from on-site wastewater treatment.

2. Materials and methods

2.1. Identification of on-site treatment facilities

In co-operation with seven Swedish municipalities, municipal databases were searched for on-site wastewater treatment facilities that used sand filters and/or P-filters. The operators of these facilities were contacted by telephone to obtain their approval. Thirty-four facilities were inspected for their suitability to the sampling intended to be carried out in this study. During inspection, the sand filter outlets were checked to ensure they were designed in a way that would enable flow-proportional sampling, that is, manual flow measurements during sampling. For P-filters, inlets and outlets were both checked for their accessibility for sampling and whether at least one of them was suitable for measuring the flow.

2.2. Investigated on-site treatment facilities

A description of the on-site wastewater treatment facilities investigated is given in Table 1. Eight sand filters, four other biological treatment units, namely a trickling filter with biological fibre material in a tank, two biofilm units without aeration and one aerated activated sludge unit, and nine P-filters were studied.

The sand filter beds that were constructed in accordance with the Swedish standard design (sand filters A and E) had an 80 cm thick layer of filter media (sand/gravel with particle sizes from 0 to 8 mm). A design load of 30–60 L m⁻² d⁻¹ is common in Sweden [22] and a sand filter for one household usually has a surface area of 25 m². The wastewater was spread over the filter through slotted pipes and collected in drainage pipes at the bottom of the filter. Both distribution and drainage pipes were embedded in a 30 cm thick gravel layer. Sand filters F–H deviated from the standard by having a layer of drainage baskets (biomodules) with triangular cross sections (width ca. 0.5 m) on the top (hosing the distribution pipe); the sand layer was 60 cm and the top gravel layer 10 cm. Sand filters B and C were smaller and had a thinner sand layer but were improved with plastic crates (biomodules) increasing the hydraulic capacity, allowing the filter bed to be smaller compared with a bed built in accordance with the standard. All sand filters were covered with a 30 cm thick layer of soil.

In facilities I–L, mini-package plants were used instead of sand filters, followed by a P-filter. In facility I, the wastewater was biologically treated in a trickling filter made of fibre material (4evergreen by Biorock®). In facilities J–K, a Biop® unit with attached growth treatment was installed, however, there was no aeration. In facility L, an aerated activated sludge technique was used.

Among the nine investigated P-filters (Table 1), eight were bags filled with the filter material Polonite (supplier: Ecofiltration AB, Sweden) which were placed in a pit and operated in downflow mode (D) or up-flow mode (F – L). The filter material Polonite provided by the company Ecofiltration Nordic AB is frequently used in P-filters installed in Swedish on-site wastewater treatment facilities. The material has grain sizes between 0.5 and 8 mm [23] and is produced by heating opoka rock [24]. In a pilot-scale test, pHs up to 12.3 have been measured [19]. P-filter E was a tank with two chambers, filled with the filter material Filtra P (supplier: Wavin-Labko Ltd., Tampere, Finland), where the water percolated downwards through the filter media in the
In Filtra P, pH values up to 12.7 have been measured [25].

2.3. Indicator bacteria

Pathogenic organisms were not expected to be present in the investigated on-site wastewater treatment facilities due to the small number of users (often only two, Table 1). Therefore, indicator organisms were used as a surrogate. Four groups of indicator bacteria were investigated in this study: total coliforms, Escherichia coli, intestinal enterococci and Clostridium perfringens. Each of these indicator organisms indicates faecal contamination and is used in establishing performance criteria for drinking water, freshwater and saltwater recreation [26].

2.4. Sampling and analyses

Twelve on-site wastewater treatment facilities (A–L, Table 1) were sampled, each on at least three occasions during ca. 3–4 h at different times of the day. The number of sampling events and the times when samples were taken differed between the investigated facilities (Table 2), because after sampling at the first sites chosen had started, the inspection of new facilities continued. At each sampling event, two samples were taken from the third chamber of the septic tank or from the distribution/pumping pit from where the wastewater was transferred to the biological treatment, from the outlet of the biological treatment unit and from the outlet of P-filter. During sampling, the flow was measured manually either at the outlet of the sand filters, or the outlet or inlet of the P-filters by capturing the influent/effluent in a measuring container and recording the time taken to fill it. Samples from the outlet of the biological treatment units and P-filters were taken, proportional to the measured flow, generating two composite samples at each sampling event. Septic tank samples were taken, using grab samples, at the beginning and end of each sampling event. Total suspended solids (TSS, not measured at all events), temperature and pH were measured in the samples in situ. TSS was determined using the European standard EN 872:2005 [27]. The pH was measured using a WTW pH330 pH meter with a WTW SenTix41 pH electrode.

Bacterial samples were stored in cooling bags and transferred to the laboratory for analysis directly after the sampling was completed. In some cases, when samples were taken in the late evening, they were transferred to the laboratory the following day. Bacterial analyses were carried out at two laboratories using the Swedish standard methods SS 028167-2 (modified) for E. coli and total coliforms, SS-EN ISO 7899-2 for intestinal enterococci and ISO/CD 14189/6461-2 for C. perfringens.

Samples for analysis of total and dissolved organic carbon (TOC and DOC) were frozen and stored for later analysis. Samples for analysis of DOC were filtered through 0.45 µm filters in situ (before freezing). TOC and DOC were analysed using IR detection (based on CSN EN 1484, CSN EN 16192, SM 5310).

2.5. Statistical analyses

The measured bacterial concentrations were weighted for the flow, that is, flow-weighted geometric means $\bar{x}_w'$ were calculated:

$$\bar{x}_w' = \left( \prod_{i=1}^{n} x_i^{w_i} \right)^{1\sum_{i=1}^{n} w_i}$$

(1)

Table 1: Properties of the investigated on-site wastewater treatment facilities

<table>
<thead>
<tr>
<th>Treatment facility</th>
<th>Biological treatment (time of start-up)</th>
<th>P-filter (time of start-up)</th>
<th>No. of users</th>
<th>Frequency of use</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>Sand filter, Swedish standard design (2009)</td>
<td>–</td>
<td>3</td>
<td>Year-round</td>
</tr>
<tr>
<td>B</td>
<td>Sand filter with biomodules (2013)</td>
<td>–</td>
<td>2</td>
<td>Year-round</td>
</tr>
<tr>
<td>C</td>
<td>Sand filter with biomodules (2015)</td>
<td>–</td>
<td>5</td>
<td>Year-round</td>
</tr>
<tr>
<td>D</td>
<td>Sand filter, Swedish standard design (2014)</td>
<td>Bag with Polonite, down-flow (2014)</td>
<td>2</td>
<td>Year-round</td>
</tr>
<tr>
<td>F</td>
<td>Swedish standard design, modified (Oct 2010)</td>
<td>Bag with Polonite, up-flow (May 2015)</td>
<td>2</td>
<td>Summer only</td>
</tr>
<tr>
<td>G</td>
<td>Swedish standard design, modified (2012)</td>
<td>Bag with Polonite, up-flow (2015)</td>
<td>2</td>
<td>Summer only</td>
</tr>
<tr>
<td>H</td>
<td>Swedish standard design, modified (Nov 2015)</td>
<td>Bag with Polonite, up-flow (spring 2016)</td>
<td>2</td>
<td>Year-round</td>
</tr>
<tr>
<td>I</td>
<td>Biological fibre material in a tank (2012)</td>
<td>Bag with Polonite, up-flow (2014)</td>
<td>2</td>
<td>Ca. 6 months/year</td>
</tr>
<tr>
<td>J</td>
<td>Biop®, biofilm treatment without aeration (2008)</td>
<td>Bag with Polonite, up-flow (June 2016)</td>
<td>14</td>
<td>Year-round</td>
</tr>
<tr>
<td>K</td>
<td>Biop®, biofilm treatment without aeration (2008)</td>
<td>Bag with Polonite, up-flow (June 2016)</td>
<td>14</td>
<td>Year-round</td>
</tr>
<tr>
<td>L</td>
<td>Activated sludge with aeration (June 2016)</td>
<td>Bag with Polonite, up-flow (June 2016)</td>
<td>2</td>
<td>Year-round</td>
</tr>
</tbody>
</table>
Table 2
Number of sampling events, bacterial samples taken from the biological treatment units and P-filters and the time when sampling was carried out

<table>
<thead>
<tr>
<th>Facility</th>
<th>No. of sampling events</th>
<th>Time of sampling</th>
<th>Total duration of sampling (h)</th>
<th>Mean flow (L h⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>7</td>
<td>Sept 2015–June 2016</td>
<td>25</td>
<td>47.8</td>
</tr>
<tr>
<td>B</td>
<td>3</td>
<td>May–June 2016</td>
<td>13</td>
<td>3.2</td>
</tr>
<tr>
<td>C</td>
<td>3</td>
<td>May–June 2016</td>
<td>9</td>
<td>10.5</td>
</tr>
<tr>
<td>D</td>
<td>6</td>
<td>Sept 2015–May 2016</td>
<td>17</td>
<td>9.5</td>
</tr>
<tr>
<td>E</td>
<td>6</td>
<td>Sept 2015–May 2016</td>
<td>17</td>
<td>15.7</td>
</tr>
<tr>
<td>F</td>
<td>3</td>
<td>Aug 2016</td>
<td>7</td>
<td>6.6</td>
</tr>
<tr>
<td>G</td>
<td>3</td>
<td>Aug 2016</td>
<td>8</td>
<td>17</td>
</tr>
<tr>
<td>H</td>
<td>4</td>
<td>Aug 2016</td>
<td>9</td>
<td>5.3</td>
</tr>
<tr>
<td>I</td>
<td>3</td>
<td>Aug 2016</td>
<td>8</td>
<td>68.1</td>
</tr>
<tr>
<td>J</td>
<td>3</td>
<td>Aug–Sept 2016</td>
<td>10</td>
<td>39.9</td>
</tr>
<tr>
<td>K</td>
<td>3</td>
<td>Aug–Sept 2016</td>
<td>11</td>
<td>29.4</td>
</tr>
<tr>
<td>L</td>
<td>3</td>
<td>Aug–Sept 2016</td>
<td>8</td>
<td>47.6</td>
</tr>
</tbody>
</table>

where \( \bar{x}_g \) = flow-weighted geometric mean; \( n \) = number of observations; \( w_i \) = volume of wastewater making up the composite sample.

Flow-weighted arithmetic means \( \bar{x}_a \) were calculated for TOC, DOC and TSS as well as the \( \text{H}^+ \) ion activity (pH):

\[
\bar{x}_a = \frac{\sum_{i=1}^{n} x_i w_i}{\sum_{i=1}^{n} w_i},
\]

where \( \bar{x}_a \) = flow-weighted arithmetic mean; \( n \) = number of observations; \( w_i \) = volume of wastewater making up the composite sample.

To determine whether there was a difference between influent and effluent bacterial concentration, a t-test was carried out. Pearson correlations of the measured parameters were calculated using the statistical software Minitab [28].

3. Results and discussions

3.1. Identification of on-site wastewater treatment facilities

The Swedish Agency for Marine and Water Management recently suggested the initiation of regular checks (including sampling) of the function and regulatory compliance of on-site wastewater treatment facilities by competent authorities [29]. In this study, however, only 12 of 34 inspected facilities could be sampled, thus meaning about 65% of facilities were not suitable for sampling. The main reasons were that there was no flow in the outlet pipe or the outlet pipe was not accessible. Furthermore, contacting the operators (private property owners) was time consuming and the long distances between the facilities made visiting impractical. These experiences show that monitoring of installed on-site treatment systems would be challenging.

3.2. Discharge of bacteria from sand filters

The bacterial concentrations in the effluents of the sand filters (facilities A–H) were, in some cases, considerable (Fig. 1). For example, in the effluent of sand filter B, concentrations of intestinal enterococci were as high as >100,000 cfu/100 mL and in the effluent of sand filter H, average \( E. coli \) concentrations were 1,192 cfu/100 mL (Fig. 1). These concentrations were many times higher than those in a study by Kauppinen et al. [10] who investigated a pilot-scale sand filter similar to the Swedish standard design at temperatures between 0°C and 15°C, and found effluent concentrations of \( E. coli \) and intestinal enterococci to be 180 and 4 cfu/100 mL, respectively. The concentrations set for excellent bathing water quality are 200 and 100 cfu/100 mL for intestinal enterococci and 500 and 250 cfu/100 mL for \( E. coli \) in inland and coastal waters, respectively [30]. The geometric mean concentration of intestinal enterococci of three of the investigated sand filters exceeded the concentration set for inland waters (Fig. 1). For \( E. coli \), this was the case in one sand filter (H). For irrigation of food crops, coliform indicators should be below \( 10^5 \) cfu/100 mL [31]. Two of the studied sand filters (B and H) exceeded this value in respect of total coliform and one (H) in respect of \( E. coli \).

The mean bacterial concentrations in the outlets of the sand filters differed considerably between facilities (Fig. 1). There are several possible reasons for this. Rolland et al. [32] found that the level of compaction of the sand affected the treatment efficiency of sand filters. The particle size distribution of the coarse sand that the authors investigated was in the same range as recommended for sand filters in Swedish guidelines. It is possible that the sand filters with elevated bacterial outlet concentrations were not well compacted during construction. As bacteria adsorption is an important removal mechanism in sand filters which depends on the adsorption capacity of the sand [33], the observed differences in removal efficiency could also be due to different properties of the sand used in the investigated filters. Filter age has been shown to affect the number of bacteria present in the filter [34] and can, therefore, potentially affect pathogen treatment efficiency. Seeger et al. [11] observed more efficient bacterial reduction in a sand filter after 1.5 years, probably due to the development of a microbial community in the top layer of the filter (schmutzdecke) that needs time to develop. In this study, however, no correlation between filter age and removal of indicator bacteria was found, possibly because
Most investigated sand filters were at an age (Table 1) where the schmutzdecke was already fully developed and other factors were predominant. For example, sand filter B was only 3 years old but performed poorly both in respect of bacteria (Fig. 1) and biological parameters (Table 3), probably due to clogging as the effluent was dark in colour with a strong smell and also because the outflow from this filter was very small (Table 2). Clogging can be caused by, among other factors, filamentous particles originating from toilet paper [35] and quickly increases when ponding of water does not disappear between feed batches [36].

While the effluent from sand filter B was dark in colour with a strong smell, no such observations were made at sand filter H where the effluent was clear and effluent TOC concentrations were at an acceptable level (Table 3), despite high effluent bacterial concentrations (Fig. 1). This shows that bacterial concentrations can be high even though there is no obvious malfunction of the sand filters observed. Recently

![Fig. 1. Measured concentrations (hollow markers) and flow-weighted geometric mean concentrations (filled markers) of total coliform (upper left), E. coli (upper right), intestinal enterococci (lower left) and C. perfringens (lower right) in the outlets of the septic tanks, biological treatment units and P-filters of the investigated facilities.](image-url)
Table 3
Flow-weighted arithmetic means of pH, TOC and DOC measured in the effluents of the biological treatment units and P-filters

<table>
<thead>
<tr>
<th>Facility</th>
<th>Weighted mean pH</th>
<th>Weighted mean TOC</th>
<th>Weighted mean DOC</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Biological</td>
<td>P-filters</td>
<td>Biological</td>
</tr>
<tr>
<td></td>
<td>treatment units</td>
<td></td>
<td>treatment units</td>
</tr>
<tr>
<td></td>
<td>[ ]</td>
<td>[ ]</td>
<td>[ ]</td>
</tr>
<tr>
<td>A</td>
<td>4.5</td>
<td>9.7</td>
<td></td>
</tr>
<tr>
<td>B</td>
<td>6.9</td>
<td>160</td>
<td></td>
</tr>
<tr>
<td>C</td>
<td>4.8</td>
<td>9.3</td>
<td></td>
</tr>
<tr>
<td>D</td>
<td>6.2</td>
<td>9.4</td>
<td>8.2</td>
</tr>
<tr>
<td>E</td>
<td>4.2</td>
<td>6.0</td>
<td>13.2</td>
</tr>
<tr>
<td>F</td>
<td>7.4</td>
<td>12.1</td>
<td>19.8</td>
</tr>
<tr>
<td>G</td>
<td>7.2</td>
<td>10.5</td>
<td>12.0</td>
</tr>
<tr>
<td>H</td>
<td>6.8</td>
<td>9.9</td>
<td>20.1</td>
</tr>
<tr>
<td>I</td>
<td>7.2</td>
<td>9.6</td>
<td>39.8</td>
</tr>
<tr>
<td>J</td>
<td>7.3</td>
<td>8.6</td>
<td>25.3</td>
</tr>
<tr>
<td>K</td>
<td>7.5</td>
<td>8.8</td>
<td>22.0</td>
</tr>
<tr>
<td>L</td>
<td>8.0</td>
<td>9.7</td>
<td>29.3</td>
</tr>
</tbody>
</table>

The concentrations of total coliform, C. perfringens and intestinal enterococci (but not E. coli) measured in the effluents of the sand filters were strongly positively correlated with the concentrations of TOC and DOC in the effluents, which was indicated by large Pearson correlation values of 0.98 (for E. coli, p = 0.007) for total coliform, 0.77 [p = 0.02] for E. coli and 0.68 [p = 0.045] for intestinal enterococci. For example, the flow through the P-filters I–L was higher compared with most of the other P-filters (Table 2) and the effluent bacterial concentrations had the same trend (Fig. 1). This shows the importance of the flow conditions and residence time of the wastewater in the P-filters for bacterial reduction, as shown by Sélas et al. [12].
As with the sand filters, P-filter effluent concentrations of total coliform and \textit{E. coli} correlated positively with effluent TOC concentrations, which were higher in the effluents of P-filters I and L than in the other P-filter effluents (Pearson correlations of 0.77 ($p = 0.01$) for total coliform and 0.73 ($p = 0.03$) for \textit{E. coli}). Furthermore, with increasing influent bacterial concentrations, effluent bacterial concentrations increased (Pearson correlations between inlet and outlet concentrations were 0.78 ($p = 0.014$) for \textit{C. perfringens}, 0.87 ($p = 0.002$) for total coliform, 0.93 ($p = 0.000$) for \textit{E. coli} and 0.85 ($p = 0.004$) for intestinal enterococci) while log removal rates (Fig. 2) were not affected.

Log removal of bacteria varied between bacteria type and P-filters, with negative removal observed in several cases (Fig. 2). Due to the high data variability, paired t-tests showed no significant ($\alpha = 0.05$) difference between the average influent and effluent concentrations of either bacteria type, measured at facilities D–L. This means that the collected data did not generally confirm a further reduction of the bacterial content of the wastewater in the P-filters, despite their high pH (Table 3) and despite a strong correlation between the log removal of total coliform and pH (Pearson correlation were 0.75, $p = 0.02$). Possibly, the pH in many of the P-filters was not high enough to support removal of bacteria other than coliforms. The overall ineffective reduction of bacteria in the P-filters could be due to the fact that the P-filters were flow-saturated, thus decreasing the attachment of bacteria to the filter particles, as suggested by Cooper et al. [38], who observed that increased moisture content likely reduced bacterial attachment to soil. However, P-filters D, F, H and J removed all four bacteria types (positive log removal, Fig. 2) indicating a potential of the P-filters to serve as a cleaning step not only for P but also for bacteria. Possibly, the number of P-filters investigated in this study was not large enough to prove their efficiency.

P-filter media need to be changed after a certain time in use because they become saturated with P. Although the necessary change intervals are uncertain, it is usually recommended to change the material after 2 years of use. In terms of bacterial reduction, the age of the filter was not observed to be a decisive factor as the outlet bacterial concentrations did not correlate with filter age, and only the log removal rates of \textit{E. coli} correlated with filter age (Pearson correlation of –0.73, $p = 0.03$).

3.5. Seasonal differences in bacterial reduction

Facilities A, D and E were sampled during the cold season (autumn 2015) as well as in warmer weather (spring to autumn 2016). Below-zero air temperature during sampling was measured at facilities A and D. At facility A, the temperature was –1°C when sampling on 15th October 2015. At facility D, the temperature was –0.6°C during the evening sampling event on 13th October 2015. In the sand filter effluents of facilities A and D, the geometric mean concentration of \textit{E. coli} was lower during the cold sampling events (9 and 8 cfu/100 mL at A and D, respectively) compared with the other sampling events (30 and 37 cfu/100 mL at A and D, respectively). This contradicts findings of previous studies where the adsorption of bacteria had been reported as reducing with decreasing temperature while the survival of \textit{E. coli} increases down to 5°C [37]. However, as air temperatures during sampling were low, the temperatures of the water when samples were taken were, on average, 4.8°C and 4.9°C at sand filters A and D, respectively. Further cooling during outside storage in the cooling bag is possible. Thus, possibly, \textit{E. coli} did not survive the handling of the samples indicating that this indicator bacteria type is unsuitable for assessing on-site wastewater systems at cold temperatures.

Differences in concentration of intestinal enterococci were not as distinct. The effluent concentration of intestinal enterococci of sand filter A was much lower at cold (<10 cfu/100 mL) compared with warmer temperatures (140 cfu/100 mL). For sand filter D, however, no such pronounced difference was observed (10 cfu/100 mL at the cold temperature sampling event compared with 8 cfu/100 mL during all other events).
3.6. Risk of pathogen discharge from on-site wastewater systems

In this study, only facilities with an above-ground outlet were investigated. However, at many facilities, the effluent is infiltrated into the ground (below-ground outlet) with the potential risk of contaminating the groundwater. In Sweden, about 1,200,000 people use their own private wells for drinking water supply [22], underlining the importance of clean groundwater in areas with on-site wastewater treatment. Although soil has been found to reduce microbiological contaminants effectively [14,15], also in cold climates [13], contamination of drinking water wells [39] and the groundwater [2,40,41] has been reported as occurring. It has also been stressed that it is important that an effective reduction of bacteria is achieved in the unsaturated zone because, in saturated soil, bacteria are spread faster and over longer distances [42]. Therefore, the relatively high effluent concentrations of indicator bacteria observed in this study (Fig. 1) confirm the risk to the groundwater created by Swedish on-site wastewater treatment systems, especially if the vadose zone below the infiltration is not deep enough.

As many Swedish on-site facilities are located by lakes and watercourses used for recreational purposes, pathogen discharge (Fig. 1) is an important issue because it represents a risk to good bathing water quality as stipulated in the EU bathing water directive [30] or the US Recreational Water Quality Criteria, thus posing a risk to human health [43]. This problem has also been reported in other countries, such as Australia [44]. Health risks are especially great if the facilities directly discharge into small watercourses used for human activities where the effluent is hardly diluted and microbiological contamination can reach high levels. On the other hand, at times, cool temperatures in Sweden possibly contribute to a concentration decrease of certain bacteria after discharge.

4. Conclusions

The bacterial concentrations in the effluents of the sand filters were considerable. Three and one of the eight investigated sand filters exceeded the criteria set for excellent water quality by the EU bathing water directive with regard to intestinal enterococci and E. coli, respectively. In one sand filter, effluent E. coli concentrations were high although no obvious malfunction was observed. The effluent concentrations of total coliform, C. perfringens and intestinal enterococci (but not E. coli) measured in the effluents of the sand filters were strongly positively correlated with the effluent concentrations of TOC and DOC. However, effluent bacterial concentrations did not significantly correlate with pH and filter age. Unexpectedly, E. coli concentrations in the effluents of the sand filters decreased during a sampling event carried out at sub-zero temperatures, possibly due to sample handling. This indicates that E. coli is unsuitable for assessing on-site wastewater systems at cold temperatures.

In the effluents of the other investigated biological treatment units (a trickling fibre filter, two units with attached growth treatment and one aerated activated sludge technique), bacterial concentrations were very high, possibly due to reasons such as a shorter retention time of the wastewater in these facilities, missing aeration and little time between start-up and measurements.

The theory that P-filters used as a cleaning step after the biological treatment would further reduce the bacterial content of the treated water was not generally confirmed. Results from P-filters investigated in this study showed that, on average, the filters did not further reduce the bacterial content. However, data variability was high, making this result somewhat uncertain. Reduction in bacterial concentration was higher in P-filters with low hydraulic load as well as low effluent TOC.

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References

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Package plants for wastewater treatment in cold climates – Treatment efficiency and occurrence of micropollutants

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Package plants for wastewater treatment in cold climates – Treatment efficiency and occurrence of micropollutants

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Package plants for wastewater treatment in cold climates – Treatment efficiency and occurrence of micropollutants

Abstract

Package plants for on-site sanitation are implemented in many areas of the world not connected to a sewage network, and their numbers have more than doubled in Sweden in the last six years. Biochemical treatments are commonly applied in them, such as activated sludge and coagulation/sedimentation. Previous studies on their performance have detected highly variable and elevated effluents concentrations of nutrients and bacteria, indicating that they negatively influence the receiving waters. Small package plants are exposed to environmental factors such as low temperature, especially in cold regions with low winter temperatures and deep frost penetration, that can adversely influence the biochemical processes. Thus, the aim of this study was to investigate the treatment efficiency and effects of cold temperatures on existing package plants. Analysis of water quality parameters of 11 package plants’ influents and effluents generally detected no strong correlation between wastewater temperature and biological oxygen demand (BOD) or levels of nutrients and indicator bacteria in the effluents, due to high variability of the data and effects of other process parameters. However, weak negative relations between effluent concentrations and wastewater temperatures were detected in specific plants, indicating that temperature does have effects. Most plants strongly removed BOD (>91%). Six plants provided good phosphorus removal (>71%). Small degrees of nitrification were observed in almost all the facilities, despite the low temperatures, while denitrification was only observed in two plants which had recirculation and high nitrification rates (>51%). When managed adequately, package plants can provide high BOD and phosphorus removal, but nitrogen and bacteria removal remains challenging, especially at low temperatures. Pharmaceutical compounds were detected in the effluents at concentrations within or above ranges reported for large treatment plants, suggesting that package plants may be diffuse sources of micropollutants in the environment. Phthalate ester concentrations were below commonly reported effluent concentrations.

Keywords On-site, nitrogen, phosphorus, bacteria, phthalates, pharmaceuticals
Package plants for wastewater treatment in cold climates – Treatment efficiency and occurrence of micropollutants

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1. Introduction

On-site wastewater treatment systems are used for treatment and disposal of domestic wastewater in areas where households are not connected to a municipal sewage network. Soil-based systems such as infiltration systems or sand filters are traditionally used, despite their contributions to discharges of nutrients and bacteria into surface waters [1–4]. Alternatively, package plants have become attractive options in areas where space is restricted, or implementation of soil-based systems is limited by the bedrock, soil composition or fluctuating groundwater tables. About 5% of the on-site sewage systems in Sweden are package plants according to the latest official estimates [5], compared to 2% six years earlier [6].

Although well-established processes are applied in package plants, their treatment performance has not been comprehensively studied at full scale, and the limited number of available studies and monitoring reports have shown that their performance varies greatly [4,7–9]. Estimated pollutant loads from on-site wastewater treatment systems are usually based on person equivalent (PE) discharges of organic matter and nutrients, often with assumptions that they work as designed. However, the treatment efficiency is often poor due to construction errors, operational deficiencies or inadequate maintenance, so discharges are likely to be much higher than estimated, with adverse effects on the receiving waters and environment [4,6,10–12].

Phosphorus removal by on-site systems has been investigated, partly due to its implications for eutrophication of the Baltic Sea e.g., [2,11], but their nitrogen removal has not been thoroughly studied. Effects of cold temperatures on the biochemical processes in package plants also require more attention as water temperatures strongly influence microbial growth rates, metabolism and substrate affinities [13]. In cold regions such as Canada and northern parts of Europe, the USA, China and Russia, average annual temperatures are low (<15 °C), so biological processes for wastewater treatment in on-site systems may be perturbed, as shown by Kauppinen et al. (2014) [14] in a study of on-site sand filters for wastewater treatment. On-site sewage systems may also reportedly discharge significant quantities of micropollutants into the aquatic environment [15] although very few studies have investigated the presence, removal and discharge of micropollutants including pharmaceuticals in on-site wastewater systems. High removal rates of per- and polyfluoroalkyl substances (PFAs) and phosphorus-containing flame retardants (PFRs) have been observed in package plants, whereas soil-based systems have been shown to remove pharmaceuticals more thoroughly [16]. The presence of phthalates in on-site wastewater systems has not been previously studied. The present study addresses the above-mentioned gaps in our knowledge.

As package plants are becoming more widely implemented and national regulations stricter, better understanding of effects of local conditions on the treatment processes is needed because of possible impacts on the receiving environment. Thus, objectives of this study were to investigate the treatment
efficiency and effects of temperature on the pollutant removal mechanisms in different types of package plants treating domestic wastewater already in operation in rural areas. Specifically, we evaluated the treatment efficiency of 11 on-site package plants in terms of removal of organics, solids, nutrients and indicator bacteria. We also assessed levels of selected micropollutants, including pharmaceuticals and phthalate esters, in influents and effluents of two selected facilities.

2. Methods

The study design was selected to be able to evaluate the variability of influent and effluent wastewater quality over time and effects of seasonal and other operational variations on treatment in full-scale package plants. Because influent wastewater quality data is rarely reported in scientific studies, it was important to include sampling of influent wastewater in the study design.

2.1 Selected package plants

The 11 evaluated on-site facilities included seven types of batch and continuous flow plants (Table 1).

Six plants operated in continuous mode: one with a trickling filter (TF), one with a rotating biological contactor (RBC) and four with activated sludge with phosphorus removal by coagulation (ASC) or alkaline filter (ASF1, ASF2, ASF3) systems. Five plants supplied by three different manufacturers operated in batch mode with activated sludge and coagulation for phosphorus removal. The facilities were located at ~65 °N latitude in a subarctic climate (Dfc classification, Köppen), with about 600 mm precipitation (half falling as snow on average).

Table 1 Specifications of the 11 evaluated package plants, with codes reflecting the main treatment process.

<table>
<thead>
<tr>
<th>Code</th>
<th>Treatment steps</th>
<th>Type</th>
<th>People connected</th>
<th>Age (years)</th>
<th>Number of sampling occasions</th>
<th>Location</th>
</tr>
</thead>
<tbody>
<tr>
<td>ASC</td>
<td>S, AS, C, D</td>
<td>Continuous</td>
<td>32</td>
<td>5</td>
<td>Influent (10)</td>
<td>Sweden</td>
</tr>
<tr>
<td>ASF1</td>
<td>S, AS, F</td>
<td>Continuous</td>
<td>20-30</td>
<td>2</td>
<td>Influent (9)</td>
<td>Sweden</td>
</tr>
<tr>
<td>ASF2</td>
<td>S, AS, F</td>
<td>Continuous</td>
<td>2</td>
<td>2</td>
<td>Influent (10)</td>
<td>Sweden</td>
</tr>
<tr>
<td>ASF3</td>
<td>S, AS, F</td>
<td>Continuous</td>
<td>3</td>
<td>2</td>
<td>Influent (10)</td>
<td>Sweden</td>
</tr>
<tr>
<td>RBC</td>
<td>S, C, R</td>
<td>Continuous</td>
<td>10-30</td>
<td>30</td>
<td>Influent (4)</td>
<td>Finland</td>
</tr>
<tr>
<td>SBR1</td>
<td>S, AS, C</td>
<td>Batch reactor</td>
<td>4</td>
<td>4</td>
<td>Influent (13)</td>
<td>Sweden</td>
</tr>
<tr>
<td>SBR2</td>
<td>S, AS, C</td>
<td>Batch reactor</td>
<td>4</td>
<td>1</td>
<td>Influent (9)</td>
<td>Sweden</td>
</tr>
<tr>
<td>SBR3</td>
<td>S, AS, C</td>
<td>Batch reactor</td>
<td>2</td>
<td>1</td>
<td>Influent (2)</td>
<td>Finland</td>
</tr>
<tr>
<td>SBR4</td>
<td>S, AS, C</td>
<td>Batch reactor</td>
<td>1-4</td>
<td>2</td>
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<td>Finland</td>
</tr>
<tr>
<td>SBR5</td>
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<td>Batch reactor</td>
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<td>2</td>
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<td>Finland</td>
</tr>
<tr>
<td>TF</td>
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<td>Continuous</td>
<td>12-14</td>
<td>8</td>
<td>Influent (8)</td>
<td>Sweden</td>
</tr>
</tbody>
</table>

1 Years in operation when the first sample was taken in 2019.
All the systems included either a separate or built-in septic tank for sedimentation of coarse particles before the biological treatment. In plants using coagulation, aluminum-based salts were added directly into the process tank before the sedimentation phase (SBR1-S, or a different chamber after the bioreactor at the inlet of the final clarifier (ASC, RBC, TF). For better visualization, photos of some of the facilities can be found in the Supplementary Material.

2.2 Wastewater sampling and analyses

During the period August 2019-April 2021, covering different seasons and temperature ranges we collected influent samples from each plan’s sedimentation tank before the biological process facility and effluent samples from an outlet pipe, sampling chamber, or the last chamber containing the treated wastewater before discharge. On each sampling occasion we measured the wastewater temperature, total suspended solids (TSS) contents and pH of the samples, and their turbidity on some occasions. In addition, the water temperature was continuously measured in the process tank of three package plants with HOBO® Pendant®MX Temp (MX2201) loggers. Data from local weather stations [17] were used for the air temperature analyses. Levels of BOD, phosphorus (total and dissolved: tot-P and dis-P, respectively), nitrogen (tot-N, NO2-N, NO3-N, NH4+-N), and the indicator microorganisms Escherichia coli and enterococci were measured in approximately 3 L grab samples of influent and effluent water (and chloride in some of these samples). Portions of samples used for analyses of BOD and nutrients were stored frozen until analysis, and portions used to determine densities of the bacteria were stored at 5 °C and examined in an accredited laboratory within 24 hours. Detailed information about the physicochemical analyses is presented in the Supplementary Material.

2.4 Micropollutant analyses

Samples of the influent and effluent of two facilities (ASC and TF) and blank samples of the sampling equipment (using tapwater) were collected on three occasions (March, June and August 2022) for micropollutant analyses, using a stainless-steel sampler, then stored in glass jars before analysis. The investigated micropollutants included 19 pharmaceuticals (Table 2), an artificial sweetener (acesulfame K), caffeine and 15 phthalate: Bis(2-ethylhexyl) phthalate (DEHP), Bis(4-methyl-2-pentyl) phthalate (BMPP), Benzyl butyl phthalate (BBP), Bis(2-ethylhexyl) terephthalate (DEHT), Dibutyl phthalate (DBP), Diethyl phthalate (DEP), Di-n-hexyl phthalate (DHP), Diisobutyl phthalate (DIBP), Diisononyl phthalate (DINP), Diisopentyl phthalate (DISP), Dimethyl phthalate (DMP), Dioctyl phthalate (DNOP), Dipentyl phthalate (DPP), Dicycleyhexyl phthalate (DCHP) and Hexyl-2-ethylhexyl phthalate (HEHP).

The samples were subjected to liquid-liquid extraction (for phthalates) or solid-phase extraction (for pharmaceuticals, the sweetener and caffeine) with no pre-filtration. Directly after extraction the samples were concentrated and analyzed with LC-MS/MS for pharmaceuticals, the sweetener and caffeine and with GC-MS/MS for phthalates. Method and field blanks were used to evaluate background levels of target phthalate compounds, and spiked control samples, at least one procedural blank per sample set, and internal standards were used for quality control. More detailed information is presented in the Supplementary Material.

<table>
<thead>
<tr>
<th>Compound</th>
<th>Characterization/uses</th>
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</thead>
<tbody>
<tr>
<td>Diclofenac</td>
<td>Analgesic and anti-inflammatory</td>
</tr>
<tr>
<td>Ibuprofen</td>
<td>Analgesic and anti-inflammatory</td>
</tr>
<tr>
<td>Bisoprolol</td>
<td>β-blocking agents</td>
</tr>
<tr>
<td>Candesartan</td>
<td>ACE inhibitor</td>
</tr>
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</table>
2.5 Removal efficiency calculations and data analyses

The percentage removal efficiency was calculated for each facility using median influent and effluent values (Table 3) on each sampling occasion to relate the removal efficiency to parameters such as temperature. The overall removal of each compound was based on the influent concentrations in samples taken from the last chamber of the septic tanks or influent pipe before the process tank, so the estimated removal excluded possible removal in the pre-sedimentation stage. When influent samples were not taken within 24 hours of the effluent samples, the average of all measured influent concentrations was used to calculate the pair-wise removals. Nitrification rates were estimated by subtracting NH$_4^+$-N$_{effluent}$ from NH$_4^+$-N$_{influent}$ then dividing by tot-N * 100, and denitrification rates by subtracting NH$_4^+$-N$_{effluent}$, NO$_2$-$N_{influent}$ and NO$_2$-$N_{effluent}$ from NH$_4^+$-N$_{influent}$ then dividing by tot-N * 100.

Bacteriological data were log$_{10}$-transformed for statistical analyses, half of the lower detection limit was used for left-censored data, and upper detection limits for right-censored data. The significance of differences between influent and effluent concentrations were assessed using the Wilcoxon signed rank test for non-parametric data and correlations between parameters using Spearman rank correlation analysis (with $\alpha = 0.05$ significance level).

Table 3 Means and standard deviations, median (in bold) and minimum and maximum concentrations (in parentheses) of selected parameters measured in the influents (Infl) and effluents (Effl) of the studied package plants. No value was obtained for asterisked (*) standard deviations as all the input values were the same.
<table>
<thead>
<tr>
<th></th>
<th>ASC</th>
<th>ASF1</th>
<th>ASF2</th>
<th>ASF3</th>
<th>RBC</th>
<th>SBR1</th>
<th>SBR2</th>
<th>SBR3</th>
<th>SBR4</th>
<th>SBR5</th>
<th>TF</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>BOD₇</strong> (mg/L)</td>
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<tr>
<td><strong>Infl</strong></td>
<td>89.8±70.2</td>
<td>8.5±6.8</td>
<td>39.2±25.6</td>
<td>48.2±11.2</td>
<td>173±209</td>
<td>323.5±17.8</td>
<td>352.3±247.2</td>
<td>220±84.9</td>
<td>116.5±132.2</td>
<td>440±84.9</td>
<td>206.4±157.5</td>
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<tr>
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<td>1.96±1.2</td>
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<td>4.5±4.7</td>
<td>9.4±7.8</td>
<td>33.8±42.5</td>
<td>11.5±12.5</td>
<td>9.4±5.5</td>
<td>9.4±10.5</td>
<td>35±13.5</td>
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<tr>
<td><strong>P₁₅₀</strong> (mg/L)</td>
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<tr>
<td><strong>Infl</strong></td>
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<td>7.8±2.6</td>
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<td>17.7±10</td>
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<td>3.5±2.2</td>
<td>24.3±1.1</td>
<td>7.0±4.5</td>
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<td><strong>Effl</strong></td>
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<td><strong>P₅₀</strong> (mg/L)</td>
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<tr>
<td><strong>Infl</strong></td>
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<td>0.5±0.2</td>
<td>6.9±2.3</td>
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<td>14.4±7.1</td>
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<td>21±11</td>
<td>5.7±3.8</td>
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<tr>
<td><strong>N₅₆</strong> (mg/L)</td>
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<td></td>
</tr>
<tr>
<td><strong>Infl</strong></td>
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<td>8.2±2.3</td>
<td>45.8±10.7</td>
<td>43.1±19.1</td>
<td>145.2±29</td>
<td>115.6±122.9</td>
<td>136.1±44.4</td>
<td>598.5</td>
<td>60.5±30.6</td>
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<td>62±10.5</td>
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<td>7.3±2.5</td>
<td>36±10.3</td>
<td>32.5±13.9</td>
<td>31.3±23.3</td>
<td>45.7±25.2</td>
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<td>37.8±21.5</td>
<td>48.3±27</td>
<td>171.8±58.2</td>
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<td><strong>NH₃</strong> (mg/L)</td>
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<tr>
<td><strong>Infl</strong></td>
<td>35.9±13.3</td>
<td>4.1±2.7</td>
<td>25.7±4.7</td>
<td>29.5±4.4</td>
<td>28.5±202</td>
<td>85.1±29.9</td>
<td>112.3±33.6</td>
<td>49±2.2</td>
<td>45.2±27.2</td>
<td>190±173</td>
<td>51.2±5.6</td>
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<td><strong>Effl</strong></td>
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<td>19.4±2.8</td>
<td>9.1±17.3</td>
<td>18.9±13.3</td>
<td>7.2±7.4</td>
<td>15.2±7.5</td>
<td>32.7±16.7</td>
<td>145.3±62.5</td>
<td>46.8±8.2</td>
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<td><strong>E. coli</strong> (log10)</td>
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<td></td>
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<tr>
<td><strong>Infl</strong></td>
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<td>6.5±0.89</td>
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<td>1.7±*</td>
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<td>4.8±0.7</td>
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<tr>
<td><strong>Enterococci</strong> (log10)</td>
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</tr>
<tr>
<td><strong>Infl</strong></td>
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<td>2.1±0.7</td>
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<td>4.5±0.5</td>
<td>4.4±0.4</td>
<td>5.5±*</td>
<td>3.3±1.1</td>
<td>4.4±0.2</td>
<td>4.5±0.6</td>
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<td><strong>Effl</strong></td>
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<td>3.2±0.9</td>
<td>1±0.6</td>
<td>3±1.1</td>
<td>3.0±0.8</td>
<td>2.5±0.5</td>
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<td>1.4±0.9</td>
<td>3.8±0.6</td>
<td>2.5±0.7</td>
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<tr>
<td><strong>4.97 (3.0-5.7)</strong></td>
<td><strong>2.1 (1-2.8)</strong></td>
<td><strong>2.9 (2.4-5.2)</strong></td>
<td><strong>1 (1-2.7)</strong></td>
<td><strong>3.3 (2.3-8)</strong></td>
<td><strong>2.7 (1.7-4.3)</strong></td>
<td><strong>2.7 (1.8-3.1)</strong></td>
<td><strong>1.9 (1-5.1)</strong></td>
<td><strong>2.1 (1-3)</strong></td>
<td><strong>3.6 (3.3-4.4)</strong></td>
<td><strong>2.2 (1.7-4.6)</strong></td>
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</tbody>
</table>
3 Results and discussion

3.1 Relationship between air and process temperatures

Effects of the air temperature on the wastewater temperature in the process tanks (measured with sensors) were evaluated in three of the package plants (Fig. 1C). Outside temperature generally influenced the temperature inside the treatment facilities as indicated by linear regression coefficients obtained for facilities ASF3 and SBR1 ($R = 0.71$ and 0.65, respectively; Fig. 1A-B) although other factors, such as snow melting also had a considerable impact. The horizontal distribution of the data in Fig. 1A-B indicates that the water temperature in the process tanks ($2$-$20.6^\circ$C in ASF3 and $1.4$-$17.5^\circ$C in SBR1) can remain stable across a wide range of ambient temperatures, e.g., from $-35$ to $30^\circ$C, indicating strong buffering.

Bunce and Graham (2019) [12] observed similar buffering of influent wastewater temperature of 12 small treatment plants in rural UK (range: 4-$19.1^\circ$C) in air temperatures ranging from $-1.4$ to $24.3^\circ$C. They concluded that seasonal changes were not strong predictors of the reliability and performance of such plants.

![Fig. 1. A and B: Relationships between air and wastewater temperatures in process tanks of two selected facilities, ASF3 and SBR1, respectively. C: Time series of the continuous water temperature measurements in the process tanks of three facilities (ASF2, ASF3 and SBR1) and corresponding air temperatures.](image)

Fig. 1. A and B: Relationships between air and wastewater temperatures in process tanks of two selected facilities, ASF3 and SBR1, respectively. C: Time series of the continuous water temperature measurements in the process tanks of three facilities (ASF2, ASF3 and SBR1) and corresponding air temperatures.

The water temperature in the facilities remained positive during the monitoring period, with the lowest recorded temperatures being $1.4$, $1.7$ and $2^\circ$C in SBR1, ASF2 and ASF3, respectively, during April-May in both years (Fig. 1C). The coincidence of the lowest temperatures with the snowmelt period indicates that cold water from melting snow may have infiltrated the surrounding soil, cooling the process tanks and/or water infiltrated into the sewage pipes before the plants. The influent BOD concentrations in the two samples taken during April-May from SBR1 were lower ($140$ and $121$ mg L$^{-1}$) than the median ($334$ mg L$^{-1}$, Table 3), indicating infiltration of snowmelt. The chloride analysis corroborated the dilution as the influent chloride levels measured in this period were $20.8$-$26$ mg L$^{-1}$, more than 2-fold lower than those recorded in the summer months ($June/August: 55.7$-$67.7$ mg L$^{-1}$). Measures to mitigate snowmelt infiltration into sewers feeding the plants could reduce the cooling effect. Moreover, heat provided by the influent domestic wastewater may significantly help to keep treatment plants at operable temperatures [18]. Biological activity and insulation provided by the snow...
cover and the systems’ components (e.g., an insulating lid) may also contribute to the stability of the wastewater temperature.

3.2 Treatment processes and temperature effects

3.2.1 Organic matter removal

Median effluent BOD concentrations were generally low, varying between 1.5 and 13.8 mg L$^{-1}$ at all facilities except two (ASC and SBR5), where concentrations were 38.9 and 36 mg L$^{-1}$, respectively (Table 3). BOD concentrations were significantly lower in effluents than in influents at seven of the 11 facilities, with removal rates of 70% at ASF1 and 91–99% at ASF2, ASF3, RBC, SBR1, SBR2 and TF. At the facilities SBR3, SBR4 and SBR5, BOD concentrations were also substantially lower in the effluents than in the influents, but the low number of data points ($n = 4-5$) limited statistical analysis of the BOD treatment. Hence, the BOD removal levels set by the Swedish authorities (70-90%) (SwAM, 2016) and Finnish authorities (80-90%) [20] were met by at least seven and six of the 11 facilities, respectively, and perhaps a further three (with the statistical caveat just mentioned). However, facility ASC provided only 42% BOD removal, with insufficient removal of suspended solids (TSS$_{inf} = 65.7 \pm 30.4$ mg L$^{-1}$; TSS$_{eff} = 51.2 \pm 16.9$ mg L$^{-1}$), and ASC samples were highly turbid (median value: 115 Nephelometric Turbidity Units), indicating that its treatment process was not efficient. In addition, very low median influent concentrations of BOD, phosphorus, nitrogen and indicator bacteria presumably contributed to low removal rates at facility ASF1 (Table 3), where chloride measurements on two occasions ($13.1 \pm 2.5$ mg L$^{-1}$) indicated that dilution of influent wastewater had occurred.

An inverse trend between effluent BOD concentrations and effluent wastewater temperature was observed at ASF2, ASF3, SBR1, SBR2, SBR4, SBR5 and TF facilities (Fig. 2), although no significant correlations were found between the two parameters ($p=0.823$). Overall, no correlation was found in the pairwise comparison between BOD removal (calculated from influent and effluent concentrations recorded on each sampling occasion) and effluent temperature ($p=0.092$).

Previous studies on BOD removal in activated sludge processes have reported both strong and weak correlations between water temperature and BOD effluent concentrations, depending on the plants studied [21]. The removal of BOD and TSS in these processes is known to improve with increasing temperatures due to enhancements of microbial activity and floc sedimentation [21,22]. For example, at SBR1 the observed BOD removal was lowest (< 30%) during the snowmelt period and coldest months (74% removal at $T_{inf}= 7.5^\circ$C and $T_{eff}= 6.5^\circ$C). As the variables that affect treatment performance vary across plants, other factors such as influent BOD concentration, retention time or sludge settleability may also account for some fluctuation in effluent BOD [21], masking the temperature effects.
Fig. 2. Effluent BOD and temperature at four selected facilities with sample sizes \( n \geq 9 \). The lines show data trends and do not indicate significance.

3.2.2 Nitrogen treatment

Most of the influent nitrogen was in the form of ammonia in all facilities (Table 3). Tot N removal (the difference between influent and effluent tot N concentrations) was only significant at five of the plants: ASF2 (21%), ASF3 (23%), SBR1 (55%), SBR2 (79%) and TF (17%). Most of the nitrogen removed was organic, likely through particle sedimentation or filtration (in the P filters). In facilities SBR1 and SBR2, denitrification of \( \text{NO}_2^-\text{NO}_3^-\) also contributed to tot N removal (42% and 64% of the nitrified N was denitrified). Nine of the 11 plants did not meet N removal levels set by the Swedish authorities (50%, [19]) and Finnish authorities (30-40%, [20]). In SBR1 and SBR2, effluent tot-N concentrations (18.9 ± 13.3 and 7.2±7.4 mg L\(^{-1}\), respectively) were within ranges of concentrations measured in effluents of other facilities due to the high influent concentrations (Table 3), likely due to partial recirculation of the sludge, despite good removal rates.

Package plants’ configurations strongly influence their nitrogen removal capacity, as the treatment mechanisms depend on variables such as substrate quality and quantity, pH ranges, and the presence of aerobic and anaerobic niches for microorganisms capable of transforming nitrogenous compounds [23]. Nitrogen removal by denitrification can be fostered by recirculating the water or sludge from the aerated process tank into a tank with sufficient substrate such as the primary sedimentation or septic tank with anaerobic conditions [24]. This was only possible for the batch reactors, so only some of the package plants operated in that mode (SBR1 and SBR2) efficiently removed N. Certain degrees of denitrification have also been observed by Johannessen et al. (2012) [25] in package plants that provide organic-rich substrate in anaerobic environments through wastewater recirculation and contact between the nitrate and influent wastewater. Moreover, anaerobic conditions could be
unintentionally provided at the bottom of tanks where chemical precipitation occurs, e.g., in facility TF. However, the alkaline effluent environment in facilities with reactive filter material (e.g., ASF2 and ASF3) could inhibit denitrification due to high pH [26].

Nitrification occurred in most studied package plants, to different degrees. For example, up to 51 and 78% of the NH₄⁺-N was nitrified in facilities SBR1 and SBR2, respectively, while in ASF2 and ASF3 less than 44 and 27% of the NH₄⁺-N was nitrified, respectively. Temperature reportedly affects the ammonia-oxidizing bacterial community, together with other variables such as pH, alkalinity, oxygen concentration, retention time and organic load. Bacterial growth and activities can be retarded or inhibited by low temperatures [27], so nitrification rates may be higher during warmer periods. Temperatures below 8.3 °C reportedly limit nitrification, and little or no growth of nitrifying bacteria reportedly occurs at temperatures below 4°C [23, 28].

Fig. 3. Effluent temperatures and effluent minus influent NO₂⁻⁻⁻-N concentrations in four selected facilities where nitrification occurred to some extent and the sample size was n>7.

No significant correlation was found between the temperature and effluent total N concentrations (p=0.301) or NO₂⁻⁻⁻-N (p=0.619) in the whole dataset and individual facilities (as illustrated by Fig. 3), in accordance with results of previous studies of on-site treatment systems, including package plants (e.g., [29]). However, we found significant strong correlations between the water temperature and the effluent tot-N (r=-0.9) and NH₄⁻-N (r=-0.75), but not NO₂⁻⁻⁻-N (p=0.364) concentrations at facility SBR1 (Fig. 3.). Despite the lack of strong correlations, the trends observed in facilities with efficient BOD reduction and sample size n > 7 indicated a generally inverse relationship between effluent nitrogen (tot and NH₄⁻-N) and water temperature. Measured water temperatures in the studied facilities were lower than the range considered optimal for growth of nitrifying bacteria (28-36°C) and often below the limit for nitrification and growth of nitrifiers [23], which likely explains the low nitrification rates in most facilities. For example, nitrification rates in SBR1 and SBR2 were highest during the warmest periods (<18.6 °C) and lowest in cold periods (e.g., nitrification rates at 3.4, 5.6 and 17 °C at SBR1 were...
negative, 36% and 87%, respectively). Difficulties in establishing a thriving community of nitrifying bacteria and maintaining it stably during cold periods in most plants probably exacerbated disruptions of nitrogen removal. Furthermore, the hydraulic residence time in small package plants is likely too short for high nitrification rates. For example, in an experiment with synthetic wastewater and two reactors in series for nitrification and denitrification, Dinçer and Kargi (2000) [30] found that effluent NH$_4$N content increased from 0.1 to 28 mg L$^{-1}$ when hydraulic residence time decreased from 30 to 4 h, and the optimal hydraulic residence time was 15 h. The residence time in the batch reactors SBR1 and SBR2 was just 7.5 h.

Another important factor is pH, which is optimally alkaline for nitrification reactions, but the reactions consume alkalinity and low pH generally inhibits nitrification [23], although autotrophic bacteria of the genus Nitrosomas, for example, can reportedly provide high nitrification rates at low pH (3.8-4.3) [31]. The pH of wastewater received by SBR1 and SBR2 was within the ranges 6.9-7.9 and 8.1-8.9, respectively, whereas the median effluent pH was 4.9 and 6, respectively, indicating both consumption of alkalinity and low buffering capacity. In conclusion, the nitrification in facilities SBR1 and SBR2 likely lowered the wastewater pH without severely affecting the existing nitrifying bacteria, which afforded considerable nitrification rates. The higher effluent pH of facility SBR2 could account for its higher nitrification rate. Hence controlling the pH in the process tank could potentially avoid inhibition of the nitrification processes.

### 3.2.3 Phosphorous removal in different P-targeting facilities

In eight of the 11 studied plants chemical coagulation and sedimentation were the main phosphorus-removal processes (Table 1). The effluent tot-P concentrations were significantly lower than the influent concentrations in three of the eight facilities that included chemical precipitation treatment (SBR1, SBR2 and TF). The tot-P median concentrations in RBC, SBR3, SBR4, SBR5 were lower in the effluent than in the influent (Table 3), but the limited data did not allow further statistical analysis.

The proportion of dissolved phosphorus was higher in influent than in effluent samples (Table 4) from most facilities (all except RBC, SBR3 and SBR5), indicating that some of the dissolved phosphorus precipitated and the particulate form predominated after the treatment.

#### Table 4 Mean and standard deviations of diss-P/tot-P ratios of influent and effluent samples and removal rates for the facilities with phosphorus precipitation.

<table>
<thead>
<tr>
<th>Facilities</th>
<th>Influent Mean</th>
<th>Influent StDev</th>
<th>Effluent Mean</th>
<th>Effluent StDev</th>
<th>P removal (%)</th>
</tr>
</thead>
<tbody>
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<td>ASC</td>
<td>0.75</td>
<td>0.03</td>
<td>0.65</td>
<td>0.09</td>
<td>13</td>
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<tr>
<td>RBC</td>
<td>0.84</td>
<td>0.02</td>
<td>0.88</td>
<td>0.07</td>
<td>50</td>
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<tr>
<td>SBR1</td>
<td>0.86</td>
<td>0.07</td>
<td>0.27</td>
<td>0.35</td>
<td>78</td>
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<tr>
<td>SBR2</td>
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<td>0.24</td>
<td>86</td>
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<tr>
<td>SBR3</td>
<td>0.71</td>
<td>0.02</td>
<td>0.77</td>
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</tr>
<tr>
<td>SBR4</td>
<td>0.73</td>
<td>0.03</td>
<td>0.33</td>
<td>0.29</td>
<td>95</td>
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<tr>
<td>SBR5</td>
<td>0.86</td>
<td>0.03</td>
<td>0.90</td>
<td>0.05</td>
<td>42</td>
</tr>
<tr>
<td>TF</td>
<td>0.81</td>
<td>0.03</td>
<td>0.26</td>
<td>0.16</td>
<td>95</td>
</tr>
</tbody>
</table>

Coagulation/flocculation of phosphorus is generally affected by temperature, because effects of reducing the temperature on the solubility of aluminum hydroxide species raise the optimal coagulation pH, and change floc characteristics such as size [32]. However, we detected no significant correlation between the wastewater temperature and total phosphorus removal rates ($p = 0.277$), or effluent concentrations ($p = 0.08$). Neither the measured effluent concentrations nor calculated removal rates had a clear monotonic relationship with the wastewater temperature. The lack of
The correlation between temperature and total phosphorus concentration was likely due to the high variability of the data, and effects of other parameters, such as type and dosage of coagulant, pH, alkalinity and ionic strength [33,34].

In the facilities that included chemical precipitation treatment the effluent TSS concentration was strongly correlated with effluent concentrations of tot-P ($r = 0.69$, $p = 0.000$) and particle-bound P, estimated from differences between tot-P and diss-P ($r = 0.78$, $p = 0.000$). Thus, the low phosphorus removal observed in some plants could be due to insufficient sedimentation and subsequent release of particles. Furthermore, the pH strongly influences phosphorus precipitation due to its impact on chemical charges and competition between metal hydrolysis products and natural organic compounds [32]. The optimum pH reported for phosphorus removal (using aluminium coagulants) is 5.5–6.0 [32], so the chemical precipitation of phosphorus may have been impaired in facilities with lower median pH, such as SBR1. The P treatment was also probably compromised in the investigated package plants by the lack of thorough mixing (high turbulence) at the dosing points, which is required for good precipitation of dissolved P [35].

Suboptimal pH, low chemical dosage, insufficient mixing of coagulant, insufficient flocculation and sedimentation, equipment malfunction or lack of coagulant are commonly reported reasons for low P removal in package plants [25,35] and may explain the low removals observed in some of the plants we studied.

Three studied facilities had alkaline filters for P removal (Table 1): ASF1, ASF2 and ASF3, with estimated tot-P removal rates of 3%, 83% and 46%, respectively, based on median influent and effluent measurements. Influent wastewater into ASF1 was clearly affected by dilution, as previously discussed, and was not considered in this analysis. Estimated pair-wise removal rates varied considerably with time at each facility. For example, ASF2 removed between 28% of tot P in February 2020, when there was an unusually high effluent tot-P concentration (8.6 mg L$^{-1}$) and 94% in June 2020, and on both occasions the effluent pH was high (10.1 and 9.7, respectively). ASF3 removed tot P less efficiently, between 23% (in June 2020) and 83% (in October 2019), with effluent pH values (7.8 and 9.7, respectively) lower than those measured in ASF2.

The phosphorus removal capacity of alkaline filters is affected by the residence time, influent phosphorus concentrations, pH and temperature. The removal mechanism mainly involves precipitation reactions after dissolution of calcium ions from the filter material [36–38]. Precipitation of calcium phosphates is favored by high pH [39], which gradually decreases with time during operation [37,40], and as it falls below 9 due to exhaustion of the alkaline filter media, previously precipitated calcium phosphates may dissolve [37]. In this study, the effluent pH measured in facility ASF3 was below 9.9 at the beginning of the sampling campaign (Fig. 4) and decreased over time, indicating that the filter media was reaching exhaustion. The low pH at the end of the sampling campaign (7.3), and higher tot-P concentrations in the effluent than influent indicate that some dissolution of previously formed calcium phosphates may have occurred. Moderately negative correlations (based on combined data from ASF2 and ASF3) were found between the effluent pH and effluent tot-P concentrations ($r = -0.47$, $p = 0.04$) and diss-P ($r = -0.60$, $p = 0.006$), confirming the inverse relation between the parameters.
Fig. 4. Relations between the water temperature (left) and effluent pH (right) with effluent phosphorus concentrations in facilities ASF2 (blue) and ASF3 (red). Sequential regression coefficients (R-Sq) values for the ASF2 and ASF3 data are 30 and 0.7%, respectively.

The dissolved phosphorus to total phosphorus (diss-P/tot-P) ratios were higher in the influent than the effluent of ASF2 (0.89±0.04 and 0.55±0.3, respectively), but the same (0.9) in the influent and effluent of ASF3, with high effluent diss-P concentrations (Table 3), indicating that the soluble phosphorus had not been successfully precipitated, while the particulate phosphorus was removed to a greater extent.

We found a strong correlation between the effluent wastewater temperature and effluent phosphorus concentrations at facility ASF2 (r = -0.71, p = 0.03), but not ASF3 (p = 0.86) (Fig. 4, left panel). Previous laboratory studies have shown that the temperature significantly affects the binding capacity of phosphorus in alkaline filter materials, as precipitation of calcium phosphates is an endothermal reaction, favoured by high temperatures [38,41]. Herrmann et al. (2014) [41] found that total P binding capacities of two tested filter materials increased 1.2- and 1.5-fold when the temperature was increased from 4.3 to 16.5 °C. Because the filter material in ASF3 was only exhausted towards the end of the sampling campaign, changes in its chemistry may have masked temperature’s negative impact on the treatment process. Nevertheless, the highest effluent tot-P concentrations we recorded (6.4 and 8.6 mg L^-1) were at ASF2 and ASF3 during cold months (4.2 and 3.9 °C, in February 2020).

3.2.4 Inactivation of indicator bacteria

High densities of indicator bacteria were found in effluents of most plants, <2.3-5.9 (log_{10}) *E. coli* and <1-4.9 (log_{10}) enterococci per 100 mL, exceeding in most cases European Union limits for acceptable bathing water quality—200 cfu/100 mL for enterococci and 500 cfu/100 mL for *E. coli* [42]—in line with similar studies [7,29]. The removal rates were low, < 2.5 (log_{10}), suggesting that extra polishing treatment steps would be needed to obtain more hygienic effluents. No general correlation was observed between the effluent temperature and densities of either *E. coli* (p = 0.154) or enterococci (p = 0.834), in line with previous studies of conventional activated sludge systems operating at 9–15 °C (Barrios-Hernández et al., 2020). However, regression analyses of variables at specific facilities revealed strong inverse correlations at SBR1 between wastewater temperature and effluent densities of both enterococci and *E. coli* (R-Sq= 70.9 and 48.1%, respectively, n=9). Moreover, at SBR1, the lowest *E. coli* effluent densities (log_{10} 2.4) were recorded in August, when the water temperature was one of the highest measured (about 17 °C), and highest (log_{10} 4.4) in April, when the water temperature was coldest (T_{influent}=3.7 °C, T_{effluent}=5.9 °C).
3.3 Presence of micropollutants in two selected facilities

Ten of the 19 analysed pharmaceutical compounds were detected in at least one of the influent samples taken from facilities TF and ASC (Table S1). Of these, 10 and eight were detected in effluents of the TF and ASC facilities, respectively (Table 5). The detected pharmaceuticals included anti-inflammatory drugs (diclofenac, ibuprofen and ketoprofen), β-blocker drugs (bisoprolol and metoprolol), an ACE inhibitor (enalapril), anticonvulsants (gabapentin and levetiracetam) and the antidepressant venlafaxine.

Effluent concentrations of the β-blocker bisoprolol and anticonvulsant levetiracetam were within ranges of previously reported effluent concentrations, but those of venlafaxine and metoprolol were slightly higher (Table 5). Effluent concentrations of diclofenac and ketoprofen were generally higher than reported concentrations in effluents of conventional wastewater treatment plants (WWTPs), but within ranges of those reported in effluents of on-site sewage systems. In contrast, effluent concentrations of ibuprofen and gabapentin were much higher than those recorded in previous studies. We also found low concentrations of the ACE inhibitor, enalapril, while previously reported concentrations have typically been sub-limit of quantification [44].

The stimulant caffeine and sweetener acesulfame K were detected in all the influent and effluent samples of the two studied facilities. Acesulfame K is a commonly used artificial sweetener with suggested suitability as an indicator of wastewater contamination in the environment due to its persistence [45,46]. Its insensitivity to treatment was observed in this study, as the influent and effluent concentrations did not vary greatly (Table S1). Caffeine is excreted unchanged in the urine but, in contrast to Acesulfame K, it is readily degradable in conventional WWTPs and conventional on-site sewage systems, with high reported removals (>70%) [46–48]. In this study, concentrations in facility TF were clearly lower in the effluent than the influent (Table S1), but not in facility ASC, corroborating the poor performance of ASC’s biological treatment.

Table 5 Summary of concentrations of the selected pharmaceuticals (except enalapril), caffeine and Acesulfame K (µg L⁻¹) detected in effluents in this and previous studies. LOQ= Limit of quantification; ND= Not detected. Median values estimated in this study were only based on detected concentrations.

<table>
<thead>
<tr>
<th>Compound</th>
<th>This study</th>
<th>Other studies</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diclofenac</td>
<td>TF: 0.9, ASC: 1</td>
<td>TF: 0.54-14, ASC: 0.97-1.2, 0.043⁺, 0.50⁺, 0.1² (mean), 3³ (mean), 0.2⁴ (max.), &lt;0.001–0.69⁰, 1.4¹ (max.), 0.03-0.14⁴, 2.4-3.9⁵</td>
</tr>
<tr>
<td>Venlafaxine</td>
<td>TF: 1.3, ASC: &lt;LOQ</td>
<td>TF: 1.3-1.4, ASC: &lt;LOQ, 0.1¹, 0.7² (mean), 0.548⁴ (max.), 0.5-1.3⁴</td>
</tr>
<tr>
<td>Bisoprolol</td>
<td>TF: 0.06, ASC: 0.3</td>
<td>TF: &lt;LOQ-0.06, ASC: 0.2-0.5, 0.02³, 0.7⁴ (mean), 0.42³ (max.), 0.04-1.1⁴</td>
</tr>
<tr>
<td>Ibuprofen</td>
<td>TF: 13, ASC: 56</td>
<td>TF: 6.7-15, ASC: 22-67, 0.007¹, 1.9³, 0.3² (mean), 2.1¹ (max.), ND-55⁰, 17² (max.), 0.1-4.2⁴</td>
</tr>
<tr>
<td>Ketoprofen</td>
<td>TF: 0.6, ASC: &lt;LOQ</td>
<td>TF: 0.006-1.5, ASC: &lt;LOQ, 0.086¹, 1.8³, 0.02⁴ (mean), 1.6¹ (max.), 0.003-3.9², 2.1¹ (max.), &lt;0.003-0.03³, 0.1-0.5⁴</td>
</tr>
<tr>
<td>Metoprolol</td>
<td>TF: 2.8, ASC: 0.7</td>
<td>TF: 1.6-2.9, ASC: 0.7-1.5, 0.07² (mean), 0.8² (mean), 0.003–0.25 ¹, 0.03-0.1², 0.6-1³</td>
</tr>
<tr>
<td>Gabapentin</td>
<td>TF: 0.03, ASC: 52</td>
<td>TF: &lt;LOQ-15, ASC: 52-78, 15.7¹ (mean), 4.3³ (mean), 3-42.6⁴, &lt;0.9-6.5⁰</td>
</tr>
</tbody>
</table>
Diclofenac and metoprolol are considered to be poorly removed (<40%), ketoprofen moderately removed (40-70%), and ibuprofen and caffeine highly removed (>70%) in conventional WWTPs according to a simple classification scheme [46]. Our results are consistent with these general patterns, particularly for diclofenac and metoprolol, whereas ibuprofen was removed to a lesser extent (<45%) than previously reported and only in facility TF (Table S1, Fig. 5.). Caffeine was removed to a lower degree (about 50% in TF, negligible in ASC) than reported in the literature. In contrast to Kasprzyk-Hordern et al. (2009) [50], we did not detect high removal (about 84% was reported in [50]) of the anticonvulsant gabapentin by activated sludge treatment. The antidepressant venlafaxine was found in all the influent and effluent samples of facility TF, in similar ranges, confirming its persistence in aerobic processes and ability to resist wastewater treatment, as previously reported [51].

![Fig. 5. Influent and effluent concentrations of five selected pharmaceuticals in the facilities TF (green markers) and ASC (blue markers). All the compounds were measured in all the samples except gabapentin (below the detection limit in one of the influent samples of facility TF) and venlafaxine (detected in only one of the influent samples of facility ASC).](image_url)
concentration in wastewater effluents and frequency of detection in samples [16]. Use of ibuprofen is considered to result in moderate environmental risk and the predicted non-effect concentration (PNEC) is 1 µg L\(^{-1}\) according to the Swedish pharmaceutical database (FASS, n.d.). The lowest PNEC for diclofenac is 0.05 µg/l according to the Norman Ecotoxicology Database (Norman, n.d.). A suggested environmental quality standard (EQS) of 0.040 µg L\(^{-1}\) for diclofenac is under consideration for inclusion in the EU water legislation (watchlist) [52]. All the effluent diclofenac concentrations recorded in this study exceeded 0.5 µg L\(^{-1}\) (Table S1), implying environmental hazards because of its toxicity to aquatic organisms.

Seven of the 15 analysed phthalates (DEHP, BBP, DEHT, DBP, DEP, DIBP, DMP) were detected on at least one occasion in effluents of the two studied plants (Tables S2a-b, Fig. 6). The highest concentrations measured in the effluents were of the phthalates DEP and DIBP (380 and 270 ng L\(^{-1}\) in TF effluents, 240 and 310 ng L\(^{-1}\) in ASC effluents, respectively). Concentrations of DBP and DEHT were also high in effluents of facility ASC (408 and 380 ng L\(^{-1}\), respectively). The influent and effluent concentrations of the mentioned phthalates were below concentrations reported in effluents of conventional WWTPs [46,47,53], likely due to their smaller scale (e.g., excluding industrial wastewater). The concentrations of phthalates in facility TF were always lower in the effluents than the influents, indicating some removal, but concentrations were higher in some effluents from the plant ASC than the influents (Table S2a).

![Fig. 6. Influent and effluent concentrations of six selected phthalate esters detected in the two studied facilities, ASC (blue markers) and TF (green markers). All the measured concentrations, including blanks, are shown in Table S2a-b.](image)

DEHP is included in the European Union’s list of priority substances [54]. The highest DEHP effluent concentration measured in the present study was 0.27 µg L\(^{-1}\) (Table S2a), which is lower than the EQS
(annual average value) for inland and other surface waters of 1.3 µg L\(^{-1}\) [54] and not shown in the graph due to the high concentrations found in two of the three blanks taken.

Biodegradation by bacteria and fungi, or adsorption, are considered important mechanisms in the degradation and transformation of phthalate esters [53]. The degradation is much slower in anaerobic than in aerobic conditions because of the lack of syntrophic microbial communities, together with suboptimal temperatures, pH, initial phthalates concentrations and carbon sources [53]. The removal rates in conventional WWTPs are reportedly high for most phthalates, including DEP, DBP, BBP, DEHP and DMP, e.g., >90% according to Deblonde et al. (2011) [47] and 73-87% according to Gao and Wen (2016) [53]. We found stronger removal of phthalates in facility TF than in ASC (Fig. 6.), in accordance with the other measured parameters and indicating that removal of phthalate esters will be suboptimal in package plants with suboptimal biological treatment.

Micropollutants’ fate and likelihood of escape in WWTP processes are affected by various characteristics, such as their sorption capacity (affected by their hydrophobicity) and biodegradability (affected by their bioavailability), so non-volatile and polar compounds are most likely to remain untreated [46]. Parameters of the treatment plants, such as sludge and hydraulic retention times, likely influence micropollutants’ removal as they affect key components of the microbial community (e.g., nitrifying bacteria), and time available for biodegradation and sorption processes [56]. The package plants investigated in this study generally had shorter retention times (and thus presumably the contact time for biodegradation and sorption processes) than full scale WWTPs.

The biodegradation and partition processes involved in micropollutants’ degradation can be affected by temperature. Their removal is generally enhanced by temperature increases due to associated increases in microbial activity [57,58], although Suarez et al. (2010) found they had negligible influence in the temperature range 16–26 °C for pharmaceuticals and personal care products. The ambient temperature has positively significant effects on phthalates’ removal in various kinds of treatments and facilities, including an activated sludge process, batch reactors and UASB treatment, according to Gani and Kazmi (2016) [55], who recorded lower average removals at 17 °C than at 30°C, especially for DEHP, DEP and DBP. The low temperatures registered in this study are unfavourable for microbial populations and may affect their enzymatic activities, leading to lower rates of phthalate biodegradation, as Gao et al. (2014) [60] concluded in a study of cold winter conditions’ effects.

Gros et al. (2017) [16] found no clear differences between on-site package plants and medium and large scale WWTPs in concentrations and removals of micropollutants (e.g., pharmaceuticals), and concluded that pharmaceuticals removals in both types of facilities seem to be rather low. However, there are treatment-related variations, e.g., use of trickling filters reportedly results in lower removal of pharmaceuticals (<70%) than activated sludge systems (85%) in full-scale WWTPs [50]. We found that the facility with a trickling filter (TF) seemed to provide higher removal than the activated sludge plant (ASC), which was underperforming, so the results should not be extrapolated. Most package plants do not discharge directly into receiving water bodies, but into ditches or infiltration beds where the effluent is in contact with soil and vegetation, so further treatment processes may occur before the substances reach aquatic environments. Nevertheless, the results indicate that many on-site sewage plants may function poorly and discharge significant amounts of micropollutants into the surrounding environment and receiving waters, in contrast to general results of studies limited to relatively well-functioning sewage systems, as discussed by Blum et al. (2017)[61].
Conclusions

Eleven package plants for on-site treatment of domestic wastewater were investigated during a two-year sampling campaign. We also monitored the occurrence and fates of micropollutants in two of them. The conclusions can be summarized as follows:

- Wastewater temperatures remained above freezing (>1.4 °C) even during periods with the lowest outside temperatures (about -30 °C). Snow melting appeared to have a stronger cooling effect (due to water infiltration into the systems) on the wastewater temperatures than air temperatures.
- No strong correlations were found between effluent BOD or nutrient concentrations and the wastewater temperature, although (for example) the BOD removal appeared to be the lowest in some facilities (e.g., SBR1, SBR2) during the coldest months.
- Nitrogen removal was generally low and apparently largely due to sedimentation of particulate organic nitrogen.
- Nitrification rates were only high (>51%) in two facilities, especially during the warmest periods. The low average temperatures likely prevented high nitrification rates, which could be enhanced by improving the plants’ insulation to keep the water temperature above the limit for nitrification.
- Denitrification was observed, to a limited extent, in two facilities that included water/sludge recirculation of the nitrified N to the septic tank.
- Phosphorus removal rates were good (>71%) in six of the 11 plants, and highest in plants with coagulation. However, no clear relationships between temperature and total phosphorus concentration or removal were established.
- An effect of temperature on the adsorption/precipitation processes in alkaline P filters was confirmed in one facility (ASF2) where the filter pH remained high (>9.5), but not in the plant where the filter media was already likely exhausted (ASF3, pH < 7.3). However adequate operation and maintenance would likely be sufficient to ensure high P removal rates and the effect of cold climatic conditions can be considered negligible.
- Large densities of *E. coli* and enterococci indicator bacteria were found in effluents of most plants, <2.3-5.9 and <1-4.9 (log10), respectively, as the removal rates were low, < 2.6 (log10), suggesting that additional treatment steps are needed to improve sanitation.
- Ten of the 19 analysed pharmaceutical compounds were detected in at least one of the samples taken from the two monitored facilities (10 and 8, respectively, in effluents of the TF and ASC facilities).
- The detected pharmaceuticals included anti-inflammatory drugs (diclofenac, ibuprofen and ketoprofen), β-blocker drugs (bisoprolol and metoprolol), an ACE inhibitor (enalapril), anticonvulsants (gabapentin and levetiracetam) and the antidepressant venlafaxine.
- Seven of 15 analysed phthalates (DEHP, BBP, DEHT, DBP, DEP, DIBP, DMP) were detected on at least one occasion in effluents of the two studied plants.
- Overall, concentrations of the pharmaceuticals measured in the effluents were within or higher than ranges previously reported in effluents of conventional WWTPs, but those of phthalates were below previously reported effluent concentrations.
- No clear removal of persistent compounds like diclofenac, metoprolol and the sweetener acesulfame K was detected, but some removal of ibuprofen, caffeine and the detected phthalate esters was detected in one of the two studied plants.
- Package plants for domestic wastewater treatment may be sources of micropollutants for the receiving environment.
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Authors’ contributions

Brenda Vidal: Methodology, Formal analysis, Investigation, Data curation, Writing – Original draft, Writing – Reviewing and editing, Visualization. Juho Kinnunen: Methodology, Formal analysis, Investigation, Data curation, Writing – Reviewing and editing, Visualization. Annelie Hedström: Conceptualization, Writing – Reviewing and editing, Supervision. Elisangela Heiderscheidt: Conceptualization, Writing – Reviewing and editing, Supervision, Project administration, Funding acquisition. Pekka Rossi: Conceptualization, Writing – Reviewing and editing, Funding acquisition. Inga Herrmann: Conceptualization, Methodology, Writing – Reviewing and editing, Supervision, Project administration, Funding acquisition. All authors approved the final article.

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1. Methods

1.1. Chemical analysis of samples

The samples to determine total and dissolved P, total and dissolved organic carbon (TOC, DOC) and biological oxygen demand (BOD$_7$) were frozen and stored for later analysis in the laboratory. Samples taken from the Swedish sites for DOC and dissolved P analyses were filtered through 0.45 μm filters before freezing. BOD$_7$ was analysed according to the European standard method CSN EN 1899-1 (European Committee for Standardization, 1998). Phosphorus and nitrogen compounds were analysed with a QuAAtro segmented flow analyzer (molybdate method) with digestion with persulfate oxidation according to SS-EN 1189:1996 performance 6.4. The device-specific method number for total and dissolved P, total N, NO$_3$-N and NO$_2$-N (without reduction column) and NH$_4$-N were A-031-04, Q-003-04 and Q-001-04, respectively. The NH$_4$-N was analysed following the ISO 11732 and DIN 38 406 (Part 23, section 2) standard methods.

The samples for microbiological analyses were stored at 5 degrees during sampling. The indicator bacteria *E. coli* and enterococci were analysed in an accredited laboratory within 24 hours based on the methods SS EN-ISO 9308-2:2014 and IDEXX Enterolert® for both microorganisms, respectively. The results were reported as most probable number (MPN) and the detection limit was 10 MPN/100 ml with 95% confidence level.

The temperature and pH were measured on-site during every sampling occasion using a WTW pH330 pH meter with a WTW SenTixx41 pH electrode (Swedish samples) and a WTW multi 350i with Sentix electrode (Finnish samples) calibrated at each occasion (two-point calibration). TSS was determined following the European standard EN 872:2005 (European Committee for Standardization, 2005). The turbidity was measure based on ratio turbidimetric determination using a primary nephelometric light scatter signal (90°) to the transmitted light scatter signal (Model HACH 2100Q). The chloride concentration was measured by liquid determination of anions according to CSN EN ISO 10304-1 and CSN EN 16192 (ISO 10304-1, 2007).

The samples from the Finnish package plants were analysed for total P and PO$_4$ according to the ISO 15681-1:2005 and ISO 15681-1:2005, respectively. IR detection (based on CSN EN 1484, CSN EN 16192, and SM 5310) was used to analyze TOC and DOC (European Committee for Standardization, 1997, 2011). BOD$_7$ was analysed according to the standard method CSN-EN 1899-1:1998. Total N was analysed according to the ISO 11905-1:1998 FIA-technique. The samples were analysed based on the ISO 13395:1997 FIA-technique (IC104) for NO$_2$+ NO$_3$ whereas the Ammonium-N was analysed based on the ISO 3032:1976 (IC106). The indicator bacteria *E. coli* and enterococci were analysed in an accredited laboratory within 24 hours based on the methods SFS 4088:2001 / ROI and SFS-EN ISO 7899-2:2000. The results were reported as colony forming units, CFU per 100 ml.
As the bacteriological data was reported in two different units, CFU and MPN, which were considered equivalent for the purpose of comparing the effluent concentrations. However, differences in the methodology and precision of the results (Gronewold and Wolpert, 2008) are acknowledged and the results were interpreted with caution.

1.2. Micropollutants analysis

The samples were extracted with liquid-liquid extraction (for phthalates) and solid-phase extraction (pharmaceuticals, sweetener and caffeine) and were not pre-filtered. The samples were concentrated and analysed directly after the extraction with LC-MS/MS for pharmaceuticals, sweetener and caffeine and with GC-MS/MS for phthalates. Method and field blanks were used to evaluate potential background levels of target phthalate compounds. Quality control was performed in the laboratory, and included spiked control samples, at least one procedural blank per sample set and internal standards. The recoveries of the spiked controls were used to ensure the method’s performance, by assuring that the known target value(s) had been met. Procedural blanks were taken to ensure that there was no background contamination from the laboratory (vessels, instruments, chemicals, lab air, etc.). For phthalates, three procedural blanks per sample set were used to evaluate the background. The phthalate procedural blanks were subtracted from the sample results. The internal standards included the addition of mass labelled internal standards to the samples before the extraction to account for the losses of analytes during the pretreatment and matrix suppression in the instrumental analysis (LC-MS/MS or GC-MS/MS). The recovery rates of the internal standards were used to verify that the sample pretreatment and analysis procedure worked as it should.

1.3. Investigated facilities

The pictures included in this section are intended to provide a better overview of the on-site facilities sampled in this study.

Figure S1. Package plant model like in SBR1 and SBR2 (left) and package plant TF (right) consisting in a trickling filter for biological treatment, both with chemical precipitation for phosphorus treatment.
2. Results

Table S1. Influent and effluent concentrations of micropollutants from samples taken in two package plants (TF and ASC), during three different occasions.

<table>
<thead>
<tr>
<th>Compounds</th>
<th>Facility</th>
<th>Units</th>
<th>March 2021</th>
<th></th>
<th>June 2021</th>
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<th>August 2021</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
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<td>EFF</td>
<td>IN</td>
<td>EFF</td>
<td>IN</td>
<td>EFF</td>
</tr>
<tr>
<td>Caffeine</td>
<td>TF</td>
<td>µg/L</td>
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<td>180</td>
<td>96</td>
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<tr>
<td></td>
<td>ASC</td>
<td>µg/L</td>
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<td>149</td>
<td>79</td>
<td>110</td>
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<tr>
<td>Acesulfame K</td>
<td>TF</td>
<td>µg/L</td>
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<tr>
<td></td>
<td>ASC</td>
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<tr>
<td>Diclofenac</td>
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<td>13</td>
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<tr>
<td></td>
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<td>Ibuprofen</td>
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<td>ASC</td>
<td>µg/L</td>
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<td>67</td>
<td>54</td>
<td>22</td>
<td>17</td>
<td>56</td>
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<tr>
<td>Bisoprolol</td>
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<td>ng/L</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>5.7</td>
<td>63</td>
</tr>
<tr>
<td></td>
<td>ASC</td>
<td>ng/L</td>
<td>360</td>
<td>510</td>
<td>870</td>
<td>180</td>
<td>140</td>
<td>320</td>
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<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<td>28</td>
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</tr>
<tr>
<td>Clarithromycin</td>
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<td>ng/L</td>
<td>&lt;LOQ</td>
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<td>&lt;LOQ</td>
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<tr>
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<td>ASC</td>
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<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<td>4.3</td>
<td>2.1</td>
<td>1.6</td>
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<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<td>&lt;LOQ</td>
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<td>Fenbendazole</td>
<td>TF</td>
<td>ng/L</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<tr>
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<td>ASC</td>
<td>ng/L</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<td>Fluconazole</td>
<td>TF</td>
<td>ng/L</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<td>&lt;LOQ</td>
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<tr>
<td></td>
<td>ASC</td>
<td>ng/L</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<td>8.1</td>
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<tr>
<td></td>
<td>ASC</td>
<td>ng/L</td>
<td>62000</td>
<td>52000</td>
<td>43000</td>
<td>29000</td>
<td>54000</td>
<td>78000</td>
</tr>
<tr>
<td>Ketoprofen</td>
<td>TF</td>
<td>ng/L</td>
<td>8000</td>
<td>5.7</td>
<td>1000</td>
<td>1500</td>
<td>610</td>
<td>570</td>
</tr>
<tr>
<td></td>
<td>ASC</td>
<td>ng/L</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td>Levetiracetam</td>
<td>TF</td>
<td>ng/L</td>
<td>1500</td>
<td>840</td>
<td>910</td>
<td>360</td>
<td>670</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td></td>
<td>ASC</td>
<td>ng/L</td>
<td>62000</td>
<td>52000</td>
<td>43000</td>
<td>29000</td>
<td>54000</td>
<td>78000</td>
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Table S2a. Influent and effluent concentrations of phthalates (ng/L) analysed in samples taken in two package plants (TF and ASC), during three different occasions.

<table>
<thead>
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<th>Compounds</th>
<th>Facility</th>
<th>March 2021</th>
<th>June 2021</th>
<th>August 2021</th>
</tr>
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<tbody>
<tr>
<td></td>
<td></td>
<td>IN</td>
<td>EFF</td>
<td>IN</td>
</tr>
<tr>
<td>Bis(2-ethylhexyl) phthalate</td>
<td>DEHP</td>
<td>TF</td>
<td>234</td>
<td>69</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>86</td>
<td>120</td>
</tr>
<tr>
<td>Bis(4-methyl-2-pentyl) phthalate</td>
<td>BMPP</td>
<td>TF</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td>Benzyl butyl phthalate</td>
<td>BBP</td>
<td>TF</td>
<td>370</td>
<td>150</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>190</td>
<td>270</td>
</tr>
<tr>
<td>Bis(2-ethylhexyl) terephthalate</td>
<td>DEHT</td>
<td>TF</td>
<td>120</td>
<td>46</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>110</td>
<td>190</td>
</tr>
<tr>
<td>Dibutyl phthalate</td>
<td>DBP</td>
<td>TF</td>
<td>240</td>
<td>96</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>240</td>
<td>260</td>
</tr>
<tr>
<td>Diethyl phthalate</td>
<td>DEP</td>
<td>TF</td>
<td>480</td>
<td>380</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>620</td>
<td>270</td>
</tr>
<tr>
<td>Di-n-hexyl phthalate</td>
<td>DHP</td>
<td>TF</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td>Diisobutyl phthalate</td>
<td>DIBP</td>
<td>TF</td>
<td>450</td>
<td>240</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>310</td>
<td>310</td>
</tr>
<tr>
<td>Diisononyl phthalate</td>
<td>DINP</td>
<td>TF</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td>Diisopentyl phthalate</td>
<td>DISP</td>
<td>TF</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td>Dimethyl phthalate</td>
<td>DMP</td>
<td>TF</td>
<td>43</td>
<td>34</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
<td>16</td>
<td>14</td>
</tr>
<tr>
<td>Dioctyl phthalate</td>
<td>DNOP</td>
<td>TF</td>
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<td>&lt;LOQ</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ASC</td>
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</table>
Table S2b. Phthalates concentrations in blank samples taken with the same sampling equipment during the micropollutants sampling campaign.

<table>
<thead>
<tr>
<th>Compounds</th>
<th>Blank 1</th>
<th>Blank 2</th>
<th>Blank 3</th>
<th>Blank (tap water)</th>
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</thead>
<tbody>
<tr>
<td>Bis(2-ethylhexyl) phthalate</td>
<td>DEHP</td>
<td>260</td>
<td>269</td>
<td>95</td>
</tr>
<tr>
<td>Bis(4-methyl-2-pentyl) phthalate</td>
<td>BMPP</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
</tr>
<tr>
<td>Benzyl butyl phthalate</td>
<td>BBP</td>
<td>42</td>
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<td>&lt;LOQ</td>
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<tr>
<td>Bis(2-ethylhexyl) terephthalat</td>
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<td>Diethyl phthalate</td>
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<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<td>&lt;LOQ</td>
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<td>DISP</td>
<td>&lt;LOQ</td>
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<td>&lt;LOQ</td>
</tr>
<tr>
<td>Dimethyl phthalate</td>
<td>DMP</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<tr>
<td>Dioctyl phthalate</td>
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<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<tr>
<td>Dipentyl phthalate</td>
<td>DPP</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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<td>Dicyclohexyl phthalate</td>
<td>DCHP</td>
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<td>Hexyl-2-ethylhexyl phthalate</td>
<td>HEHP</td>
<td>&lt;LOQ</td>
<td>&lt;LOQ</td>
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</table>

References


Assessing the sustainability of on-site sanitation systems using multi-criteria analysis

Environmental Science - Water Research and Technology

Assessing the sustainability of on-site sanitation systems using multi-criteria analysis†

Brenda Vidal, Annelie Hedström, Sylvie Barraud, Erik Kärrman and Inga Herrmann

Small on-site sanitation systems are widely present in suburban and rural areas in many countries. As these systems often underperform and have an impact on receiving waters, understanding their overall sustainability is of interest for policy and decision-makers. However, the definition and estimation of indicators defining sustainability are challenging, as it is finding the methodological approach to combine qualitative and quantitative indicators into one comprehensive assessment. In this study, twelve indicators defined by environmental, economic, social, technical and health-related criteria were used to compare nine alternatives of on-site sanitation for single households. A non-compensatory method for multi-criteria decision analysis, ELECTRE III, was used for the assessment together with weights assigned to each indicator by a reference group. Several scenarios were developed to reflect different goals and a sensitivity analysis was conducted. Overall, the graywater–blackwater separation system resulted as the most sustainable option and, in terms of polishing steps for phosphorus removal, chemical treatment was preferred over the phosphorus filter, both options being implemented together with sand filters. Assessing the robustness of the systems was a crucial step in the analysis given the high importance assigned to the aforementioned indicator by the stakeholders, thus the assessment method must be justified. The proposed multi-criteria approach contributes to aid the assessment of complex information needed in the selection of sustainable sanitation systems and in the provision of informed preferences.

1. Introduction

Selection of appropriate technology for on-site treatment of domestic wastewater presents a challenge when environmental standards need to be met and solutions must be economically and socially acceptable. In Sweden, the most frequently used treatment systems are drain fields, which account for 30% of all the on-site sanitation systems (OSS), and facilities with only a septic tank and no further treatment (26%). Sand filters (14%) and holding tanks (11%) follow in number, whereas packaged treatment plants represent 2% and urine separation systems 1% of all the OSS. Estimations from the Swedish EPA suggest that approximately 20% of the facilities do not comply with national standards, and similar figures (an estimated 10% to 20% of existing systems) have been reported in the United States.

When assessing the sustainability of wastewater treatment systems to facilitate the selection of suitable solutions, criteria based on the three dimensions of sustainability (environmental, social and economic criteria), with the addition of two more in some cases (technical and health criteria), are often used. This is done to assure integrity and multidimensionality. However, the definition and estimation of indicators defining sustainability are challenging, as it is finding the methodological approach to combine qualitative and quantitative indicators into one comprehensive assessment. Multi-criteria (MCA) methods are tools used to support...
decision-making processes, particularly in order to deal consistently with large amounts of complex information. All MCA approaches require an exercise of judgment and nearly all decisions imply a weighting system of some sort. However, the different MCA methods differ in the way the data is combined and the extent to which they can aid practical decision making.

Previous research following decision-making methodology has addressed the challenges of wastewater treatment alternative selection e.g. Kalbar et al. (2012) but often present results from specific case studies e.g. Lennartsson et al. (2009) or include a limited number of alternatives e.g. Bradley et al. (2002) only evaluates two types of OSS. Indicators of different attributes such as those derived from life cycle assessments, cost analysis, mass balances or quantitative parameters have not been combined in a sustainability framework that is not focused in specific case studies and which includes stakeholders’ views and scenario analysis. Several studies have discussed and proposed indicators based on sustainability principles to evaluate wastewater treatment systems based on literature data. Furthermore, life cycle approaches, and environmental systems analyses and sustainability assessments have been applied. Due to the large number of small-scale and on-site sanitation technologies that currently exist, and despite the criteria and indicators already suggested in the scientific literature, there is a lack of application of such information in a knowledge-based decision support context for OSSs that also incorporate the stakeholders’ views to handle the trade-offs between indicators. The present study aims at assessing twelve sustainability indicators to compare nine OSSs at household scale using the multi-criteria decision analysis method ELECTRE III, and the present study discusses and proposed indicators based on sustainability principles to evaluate wastewater treatment systems based on literature data.

2. Materials and methods

In this study, decision criteria and stakeholders’ views were combined to form a comprehensive judgment of a number of alternatives. The method ELECTRE III was chosen with aid from the workflow schematic for sustainability analysts proposed by Rowley et al. (2012).

2.1 Sustainability criteria and indicators

The sustainability of the OSS was assessed by defining a set of sustainability criteria and related indicators. The criteria were selected based on available scientific literature assessing wastewater systems and were organized into five main categories: Environmental, Economic, Socio-cultural, Technical and Health-related. A number of assessable indicators were defined for each criterion, which were accounted for either qualitatively or quantitatively (Table 1).

A brief description of the indicators is given below, including how they were evaluated, the main sources of the input data and assumptions. Further detailed information is found in the ESI.

Nutrient removal. Nutrient removal referred to the capacity of the system to remove nitrogen (N) and phosphorus (P) from the influent wastewater. Two sub-indicators, namely removal of tot-N and P, were quantified based on previous studies and considered equally important, and thus equal weight was given. The existing data about P reduction in filters varies widely, partly due to the higher P reduction occurring initially, which decreases over time as the P adsorption capacity of the material is exhausted. The retention capacity of the soil is finite and varies with soil mineralogy, organic content, pH, redox potential and cation exchange capacity, affecting the adsorption and precipitation processes involved in the removal of P. For systems with chemical precipitation, P removal is often lower than it could be because the dosage equipment fails or there are problems controlling the addition of chemicals. However, functionality according to design was assumed for all systems, and the uncertainties related to operation and maintenance were considered under the indicator robustness instead. In soil-

<table>
<thead>
<tr>
<th>Table 1</th>
<th>Summary of the sustainability criteria and indicators. VH = very high; H = high; M = medium; L = low; VL = very low</th>
</tr>
</thead>
<tbody>
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<td>Criteria category</td>
<td>Indicator</td>
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<tr>
<td>Environmental</td>
<td>Nutrient removal (tot-N and P)</td>
</tr>
<tr>
<td></td>
<td>Potential for nutrient recycling (N, P)</td>
</tr>
<tr>
<td></td>
<td>Global warming potential (GWP)</td>
</tr>
<tr>
<td></td>
<td>Cumulative energy demand (CED)</td>
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<td>Energy recovery</td>
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<td>Economic</td>
<td>Capital cost</td>
</tr>
<tr>
<td></td>
<td>Operation &amp; maintenance cost</td>
</tr>
<tr>
<td>Socio-cultural</td>
<td>Social acceptance</td>
</tr>
<tr>
<td>Technical</td>
<td>Robustness</td>
</tr>
<tr>
<td>Health</td>
<td>Risk of pathogen discharge</td>
</tr>
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</table>
based systems and package plants, the tot-N removal depends on nitrification and (limited) denitrification processes that occur on-site. In the source separation alternatives (S1 and S2), the fractions that contain most of the N and P (>90% of the N is found in urine and feces) are collected and treated elsewhere, thus reducing the amount of nutrients discharged on-site.

**Potential for nutrient recycling.** Potential for nutrient recycling referred to the potential agricultural reuse of the different waste fractions produced in the systems in relation to the nutrients content (N and P) in each fraction as calculated for the nutrients removal indicator. The sludge from the septic tanks, the sand from the sand filters, the Polonite® filter material from the P-filters, the BW and the urine were considered potential sources of nutrients. This indicator was quantified based on the current practice in Sweden when data was available, e.g. about 34% of the generated sludge is reused as soil conditioner (Statistics Sweden, 2018), or based on assumptions when data was not available, e.g. 100% reuse of the blackwater and urine. Although the P-saturated filter material Polonite® has a high fertilizing potential as indicated in pot experiments, its potential for nutrient recycling was considered low (5% was assumed) because at present, it is rarely applied to farmland. There are no practical arrangements for recycling exhausted sand from the sand filters and drain fields and therefore the recycling of these materials was quantified to 0%.

**Global warming potential (GWP).** Global warming potential (GWP) accounted for the greenhouse gas (GHG) emissions in kg CO₂-equivalents (eq.) released during: the production of the alternatives’ components and materials (e.g. tanks, pipes, filter materials) and their transport (e.g. distance from the production site), system installation and operation (e.g. electricity consumption) and maintenance (e.g. collection of septic sludge, replacement of chemicals and P-filter), as well as the post-treatment of the fractions that were not treated on-site (indirect nitrous oxide emissions from ammonia emissions during storage of sludge, blackwater and urine). The end-of-life phase was excluded from the analysis. The calculations were based on LCA methodology standards following the global warming potential [v1.0.1, January 2015] impact assessment method and the ELCD 3.2 database (European Reference Life Cycle Database, 2016).

**Cumulative energy demand (CED).** Cumulative energy demand (CED) referred to the primary energy used during the production, transport and installation of the components and materials, during operation and maintenance of the alternatives and the post-treatment of the fractions that were not treated on-site (sludge, blackwater and urine). The calculations were based on the same LCA methodology standards as described for GWP.

**Energy recovery.** Energy recovery referred to the possibility to obtain energy in the form of biogas produced from the collected septic sludge. This indicator was evaluated qualitatively with a three-point ordinal scale that classified the energy recovery of the alternatives as low, medium or high, and was estimated proportionally to the volume of sludge produced in each alternative based on the composition of the different wastewater fractions according to Jönsson et al. (2005).

**Capital cost.** Capital cost was based on the investment to purchase the different components and services, and the manpower required for the installation of each OSS alternative multiplied by the annuity factor, which considers the amortization time and an interest rate of 4% (assumed). The lifetime of the components of each system and the amortization time were considered to be the same. The present value of the components and services were taken from the main distributors’ websites (e.g. Avloppscenter) or directly from the producers’ websites, excluding the value-added tax.

**Operation and maintenance (O&M) cost.** Operation and maintenance (O&M) cost referred to the yearly cost for the operation and maintenance of the alternatives, which included the collection and transport of blackwater, urine and the sludge from the septic tank; electricity use; purchase of consumables (chemicals, P-filter) and components (change of pump); and check-up service including effluent sampling and analysis. The present values of the components and services were taken from the main distributors’ websites or directly from the producers’ websites, excluding the value-added tax. Because the cost of emptying the sludge from the septic tanks and the holding tanks varies across the country, a representative average price was used.

**Social acceptance.** Social acceptance was defined as the user-friendliness of the alternatives with regard to the convenience, effort and degree of complexity of operating the system, from the user’s perspective. The indicator was assessed qualitatively using as a reference the alternatives considered the most socially accepted, namely A1 and A2, which were chosen because they represent the most common OSSs installed in Sweden, users are familiar with them and they are considered convenient due to their simplicity. The other alternatives were assessed in comparison to A1 and A2, in terms of how the “inconvenience” for the users increased when adding different components to the OSS. Chemical dosing equipment needs to be monitored frequently (e.g. refill the dosing tank up to a few times a year) and requires more effort from the users than e.g. changing a P-filter every 2–3 years. Holding tanks for BW could get full and cause nuisance to the users as they might not be able to use the toilets, whereas urine-diversion toilets have been reported to cause problems with odors and inconveniences with the maintenance and cleaning and users require pre-knowledge about the system.

**Robustness.** Robustness was defined using two sub-indicators, namely the risk of failure of the system and the adaptability to flow fluctuations. The sub-indicator “risk of failure” accounted for the possibility of the system to encounter a technical problem that may hinder its treatment capacity, and the likelihood of such an incident to happen. Failure was defined as the lack of adequate functioning, both partly...
or completely, of the system operating under normal conditions. The following risks were considered: the risk of the soil-based treatment units not being constructed correctly, which is a common problem; the risk of filter clogging; the risk of chemical dosing failure; or the risk of failure of the automatic equipment in the package plants, such as aeration equipment and sensors. The sub-indicator “adaptability to flow fluctuations” accounted for the capacity of the system to adapt to changes in the quantity of the flow, e.g. increase in the average water consumption because of greater presence of users, or periods of absence of users when the system is not in use; and the quality of the inflow, e.g. changes in temperature. The indicator was assessed qualitatively and the “risk of failure” was considered to be of higher importance (2/3) than the “adaptability to flow fluctuations” (1/3) because of the more severe implications of the former.

Risk of pathogen discharge. Risk of pathogen discharge was based on a qualitative assessment of the capacity of the OSS to remove pathogens from wastewater prior to discharge in the surrounding environment. The assessment was based on the number of barriers included in the systems that potentially have pathogen removal capacity and thus decrease the pathogens load (e.g. filter media, chemical precipitants). The receiving waters were also taken into account by decreasing one point in the scale, e.g. if the wastewater was discharged to a surface water system, or further infiltrated into the surrounding soil profile (drain fields) and thus posed a risk of groundwater contamination. It was assumed that surface water is the preferred type of receiving water body over groundwater because contamination is more difficult to detect, measure and remedy in the latter.

2.2 Description of the compared sanitation systems (alternatives)

Nine OSS alternatives (A1–P2) were selected and compared, including conventional widely used systems, namely sand filters and drain fields, as well as package plants and less conventional options including source separation systems (Table 2). The selection of alternatives was based on relevant literature and discussions with practitioners. The alternatives were grouped after the main and most relevant treatment process or distinguishable characteristics, as some treatment options are found under different types of systems e.g. the greywater from source-separation systems is treated in a soil-based unit. Hence, the alternatives are not completely exclusive to the type of system they are named after, and the grouping was merely made for clarity. The alternatives are described in more detail in the ESL.

The alternatives with sand filters (A1, A3, A4, S1 and S2) included a distribution chamber placed between the septic tank and the sand filter and an inspection chamber situated after the sand filters as recommended in existing guidelines. Alternatives with drain field (A2 and A5) had no inspection chamber because the wastewater continues infiltrating through the soil (no outlet).

Ultra-low-flush vacuum toilets with 0.6 L per flush (EcoVac®) and low-flush urine-diverting toilets with 0.3 for small flush and 2.5 L for big flush (EcoFlush®) were included in the source separation options S1 and S2, respectively.

2.3 General assumptions and study boundaries

Data was collected from the scientific literature, reports, national statistics, LCA databases and information from suppliers of treatment facilities. The OSSs were assumed to be of standard design for one household with an average of three persons. The functional or reference unit of the analysis was the overall sustainability score of an on-site sanitation alternative for one household per year. A selection of relevant assumptions are listed in Table 3.

The system boundaries for the LCA-based indicators GWP and CED (Fig. 1) included the treatment of BW with

<table>
<thead>
<tr>
<th>Table 2</th>
<th>Summary of sanitation alternatives</th>
</tr>
</thead>
<tbody>
<tr>
<td>Type of system</td>
<td>No.</td>
</tr>
<tr>
<td>Soil-based</td>
<td>A1</td>
</tr>
<tr>
<td>A2</td>
<td>As A1, but the wastewater continues infiltrating and percolating through the underlying soil instead of being collected under the sand filter</td>
</tr>
<tr>
<td>A3</td>
<td>As A1, but with additional polishing step for phosphorus (P) removal (alkaline P-filter) using the filter media Polonite®</td>
</tr>
<tr>
<td>A4</td>
<td>Chemical precipitation unit installed inside the house (under the sink) and dosed when water flows. Flocculation and sedimentation occurs in the septic tank (larger volume than alternatives A1–A3). Subsequent sand filter as A1</td>
</tr>
<tr>
<td>A5</td>
<td>As A4, but the wastewater continues infiltrating and percolating through the underlying soil as A2</td>
</tr>
<tr>
<td>Source separation</td>
<td>S1</td>
</tr>
<tr>
<td>S2</td>
<td>Urine-diverting toilet and collection of GW and feces in a septic tank with subsequent sand filter. Urine collected in a container and transported to a centralized facility for hygienization (6 months’ storage)</td>
</tr>
<tr>
<td>Package plants</td>
<td>P1</td>
</tr>
<tr>
<td>P2</td>
<td>A single unit buried underground operating in a 3-phase semi-continuous regime with activated sludge process, with equalization tank, aeration tank and chemical dosing</td>
</tr>
</tbody>
</table>

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1% urea, e.g. about one year of storage depending on the temperature,\(^{40,41}\) storage of urine for six months,\(^{40}\) and storage of anaerobically digested and dewatered sludge for six months (as in e.g. Kjerstadius et al. 2016\(^{42}\)); that is, when the wastewater fractions can be safely reused.\(^{40}\)

The sludge in the septic tanks is collected once a year based on current Swedish practice\(^{43}\) and transported to the nearest WWTP located 50 km away (distance assumed).

The different components produced in Scandinavia (septic tanks, distribution and inspection chambers, package plants) were assumed to be transported for an average distance of 500 km, and Polonite\(^\text{®}\) filter material was assumed to be transported by cargo ship for 300 km and truck for 800 km from the production site in Poland. Transport of the construction materials, e.g. sand or gravel, to the sites was included (50 km assumed), but not the transport of the smaller-sized components such as pumps, dosing equipment or dosing chemicals. The emissions from transport when making service visits, e.g. once a year for package plants, were disregarded.

2.4 The ELECTRE III method

ELECTRE III is a robust method that uses pseudo-criteria instead of true criteria, as the latter have strict preference for the best performance without accounting for any uncertainty.\(^{22}\) The pseudo-criteria are calculated based on preference thresholds that define a “buffer zone” between strict preference and indifference when comparing the performance of two alternatives. The added flexibility to the comparisons, as it takes into account uncertainty in the input data,\(^{44}\) makes it suitable for this study. The detailed description of the computation is found elsewhere,\(^{21,45}\) as well as the main advantages and disadvantages.\(^{46-48}\)

ELECTRE III uses a non-compensatory aggregation approach, meaning that there is no possibility of offsetting a bad score on an indicator by good scores on another indicator. The use of a non-compensatory method intrinsically implies that the study has a strong sustainability perspective, as different forms of capital are not substitutable.\(^{22}\)

In this study, the nine alternatives were assessed using twelve indicators in an evaluation matrix where the best outcome was represented by the maximum evaluation on each indicator. First, a pairwise comparison was carried out. Each alternative \(a\) was compared to another \(b\) according to two major concepts namely the concordance and the discordance. An outranking relation \(S\) between \(a\) and \(b\) was stated (\(a\ S\ b\)) \(\{\text{if there were enough arguments to decide that } a \text{ was at least as good as } b \}\) \{majority principle measured by a concordance index \(C(a,b)\), “while there was no essential reason to refute the relation” \[\text{measured by discordance indices } D_i(a,b)\].\(^{49}\)

To calculate the concordance index \(C(a,b)\), the alternatives were evaluated against each indicator by pairwise multiplying the partial concordance values \(c_i(a,b)\) obtained when comparing alternative \(a\) to \(b\) by the corresponding weights. The larger \(C(a,b)\) is, the stronger the evidence that \(a\) is preferred over \(b\).\(^{49}\) Preference \((p_i)\) and indifference \((q_i)\) thresholds were defined for each indicator and used to calculate the concordance values \(c_i(a,b)\). The indifference threshold \((q_i)\) allows one alternative strategy to be considered “insignificantly worse” than another alternative with respect to a given indicator even though its evaluation may be (slightly) lesser in value. The preference threshold \((p_i)\) determines if the value of one alternative on the indicator \(i\) is “strongly preferred” over another alternative on the indicator \(i\). Both thresholds can be expressed as a constant number or as a percentage.

To calculate the discordance index \(D_i(a,b)\), the definition of veto thresholds \((v_i)\) can be used, which expresses the possibility of the alternative \(a\) to be discredited if it is exceeded by the performance of \(b\) by an amount greater than the veto threshold, regardless of the other indicators. No veto threshold \((v_i)\) was used in this study and therefore the discordance index \(D_i(a,b) = 0\) for all pairs of alternatives.

Then, an index called the degree of credibility of the statement \(a\ S\ b\) (\(\delta(a,b)\)) is calculated by aggregating the concordance index and discordance indices. The degree of credibility \(\delta(a,b)\) indicates the extent to which \(a\) outranks \(b\). Because in this study the discordance index was zero, the
credibility of the outranking relation was equal to the concordance index $C(a,b)$. Two preliminary rankings were then established based on the values of $\delta(a,b)$, namely descending and ascending preorders or distillations. In the descending distillation, the process ranks alternatives from the best to “less good” alternatives, whereas in the ascending distillation the alternatives are ranked from the “least bad” to the worst. A final ranking is the result of the intersection of the two distillations.

The ELECTRE III-IV software version 3.x was used for the computations. Further detailed descriptions of the methodology can be found in the scientific literature.  

2.5 Weighting of indicators

From a mathematical perspective, the weights used in non-compensatory aggregation methods such as ELECTRE III represent importance coefficients as they describe the perceived relative importance of the criteria. To weigh the importance of the selected indicators, the panel method was applied. A reference group was formed with six representatives from different relevant stakeholders: the highest responsible environmental authority for OSS (Swedish Agency for Marine and Water Management), the Swedish Homeowners Association, the Federation of Swedish Farmers, the Swedish Waste Management Association and two representatives of advisors and communicators, one from the National Platform for On-Site Sanitation and one from the Centre for Water Development in Norrtälje. The reference group assigned weights first through an online questionnaire and then during a group discussion in which they could endorse the weights already given or modify them. The reference group was asked to discuss the indicators, to rank them from the most to the least important, and to give individual weights to each indicator. The most important indicator was allocated 100 points whereas the other indicators were assigned points (from 0 to 100) depending on how important they were considered in relation to the most important one. As the group discussion did not result in a consensus, the arithmetic mean of the normalized weights was used as in eqn (1).

$$W_i = \frac{1}{n} \sum_{i} \frac{w^*}{W^*} \times 100$$

where:
- $W_i =$ weight of indicator $i$,
- $w^*$ = points allocated by a stakeholder for each indicator, between 0 and 100,
- $W^*$ = total points given by a stakeholder to all the indicators,
- $n =$ number of stakeholders.

2.6 Scenarios and weighting factors associated

Scenario 0 was the baseline scenario with the initial assumptions and weights given by the stakeholders as described in previous sections. Additionally, three scenarios reflecting plausible settings of interest based on socio-demographic factors were assessed. These scenarios were developed to apply the proposed methodology and as an analogy to case studies as they describe specific but representative local conditions. Scenario 1, representative of e.g. areas in northern Sweden, was characterized by areas with surface waters without special protection status according to Swedish legislation and thus the removal of nutrients was of less importance than in scenario 0. Moreover, the scenario 1 areas were characterized by low population density with scattered houses and small areas of farmland with very low potential for recycling nutrients to the soil. For scenario 1, the indicators nutrients removal and potential for nutrients recycling were given the lowest importance (weight), while the rest of the indicators remained in the same order of importance.

Scenario 2, in opposition to scenario 1, represented areas with sensitive eutrophicated receiving waters as described in Swedish legislation and high importance was given to the...
indicator nutrient removal. Scenario 2 areas were characterized by larger population density and considerable presence of farmland where the nutrients from the OSS could be potentially recycled. For scenario 2, the indicators nutrient removal and potential for nutrient recycling were given the highest importance and alternatives A1 and A2 were removed from the analysis, as they generally do not comply with the existing guidelines on nutrients removal.54,55

Scenario 3 represented a change in political strategy, with higher demands on energy recovery and with special focus on climate change mitigation (e.g., lowering the emissions and the energy use). For scenario 3, the indicators energy recovery, CED and GWP were given the highest weights, while the other indicators remained in the same order of importance.

The weights of the three scenarios were modified based on Simos’ card method to establish weights.52,56 Simos’ card method consists of placing the indicators (which are written on cards for better visualization) in order of importance with the possibility to add blank cards in between the indicators to represent larger differences. The schematic representation of the cards’ order can be found in the ESL.5

2.7 Sensitivity analysis

Several parameters and assumptions in the input data were modified to conduct a sensitivity analysis of the baseline scenario: the lifetime of the Polonite® filters, the importance of the removal of different nutrients, the cost for treatment of blackwater and urine storage, the potential for nutrients recycling in alternative S1 and the robustness of the alternatives.

In this study, the lifetime of Polonite® filters was assumed to be three years based on distributors’ recommendations (2–4 years).57 However, previous full-scale studies e.g. Vidal et al. (2018) have shown that the alkaline material can become saturated or clogged after less than three years of use.24 Because precise estimations of the lifetime of such alkaline filters are complex due to the changes in P load, flow dynamics and weathering reactions affecting the removal mechanisms,58 the lifetime of the Polonite® filters was decreased from three to two years in the sensitivity analysis.

The removal of nutrients was assumed to be of the same importance for both N and P, and a single weight was assigned by the reference group for the indicator nutrients removal without nutrient specification. However, there are no requirements for N removal for wastewater systems for less than 10 000 PE in Europe,59 and the input of P to the Baltic Sea should be reduced to a larger extent (41%) than the N (13%) to combat eutrophication.60 In the sensitivity analysis, more importance was given to the removal of P (100% of the weight) than to the removal of N (0%).

The cost for the treatment of blackwater (with urea) and the urine (storage) was not included in the analysis, as it is generally not covered by the homeowners. However, if the costs were to be included as suggested in previous reports,26,16 the yearly O&M cost would increase. The investment cost for the treatment systems has been reported to be generally low,61 and mainly covers the eventual installation of grids, pumps, storage tanks, coverage for already existing manure tanks and the routine sampling. An increase in the yearly O&M cost was assessed in the sensitivity analysis, based on the price of urea of €300 per metric ton63 and the cost reported for urea sanitization of BW,65 between €35–110 per household and year (approximated, considering inflation), which covered for infrastructure investment, spreading of the sanitized BW and sampling. In this study, the O&M cost was intended to reflect a possible management fee that municipalities could introduce to the homeowners and was set to €20 for urine and €50 for BW, considering that the spreading on farmland was not included in the study boundaries.

The BW was assumed to be reused at a 100% rate for the estimation of the indicator potential for nutrients recycling. However, this assumption may not be realistic, e.g., not all municipalities have the infrastructure for treatment and reuse. In the sensitivity analysis, the BW was assumed to be taken to a centralized WWTP and reused to the same extent as the sludge fraction (i.e., 34% instead of 100%). The contributing emissions to the GWP were modified considering that the BW was not treated with urea (which adds extra nitrogen) but mixed in the wastewater treatment plant, and the emissions of ammonia nitrogen during storage were doubled from 5% to 10% total nitrogen, as for sludge.

The robustness of the alternatives was considered one of the most important indicators for the reference group. However, determining qualitatively the performance of each alternative in terms of robustness was challenging and the assessment could vary based on available data. In the sensitivity analysis, the performance of alternatives S2, P1 and P2 was increased from low to medium robustness, and the indifference and preference thresholds were increased from 0.5 to 1 and from 1 to 2 respectively, to include the uncertainty associated to the assessment of the indicator.

3. Results

3.1 Results of the performance matrix

The indicators’ performance estimated for each alternative are shown in Table 4. The nutrients removal was highest in the source separation alternatives (S1 and S2) for both N and P because most of the nutrients (90% of the N and 80% of the P) are contained in the feces and urine.55 The tot-N removal was low for most of the systems, ranging from 30% to 40%, except for the source separation systems S1 and S2, which had a tot-N removal of 95% and 85% respectively. The alternatives with a P removal step, either by the use of Polonite® filter (A3 and P1) or the addition of chemicals (A4, A5, P2) had high P removal (90%) together with the source separation options S1 and S2, which had 90% and 80% P removal, respectively. However, A1 and A2 had the lowest P removal (40%) as they are not designed to be a stand-alone treatment for long-term P removal26,54 but are instead intended to degrade carbonaceous material.
Performance matrix of the different alternatives with respect to each indicator

<table>
<thead>
<tr>
<th>Criteria</th>
<th>Indicators</th>
<th>Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Environmental</td>
<td>Tot-N removal %</td>
<td>% of the dry weight</td>
</tr>
<tr>
<td></td>
<td>P removal %</td>
<td>% of the dry weight</td>
</tr>
<tr>
<td>Potential for N recycling %</td>
<td>N</td>
<td>% of the dry weight</td>
</tr>
<tr>
<td>Potential for P recycling %</td>
<td>P</td>
<td>% of the dry weight</td>
</tr>
<tr>
<td></td>
<td>Global warming potential</td>
<td>kg CO2-eq. per household per year</td>
</tr>
<tr>
<td></td>
<td>Cumulative energy demand</td>
<td>MJ per household</td>
</tr>
<tr>
<td>Economic</td>
<td>Capital cost</td>
<td>€ per household per year</td>
</tr>
<tr>
<td></td>
<td>Operat. &amp; maint. cost</td>
<td>€ per household per year</td>
</tr>
<tr>
<td>Socio-cultural</td>
<td>Robustness</td>
<td>High, Medium, Low</td>
</tr>
<tr>
<td></td>
<td>Social acceptance</td>
<td>Very high, High, Medium, Low</td>
</tr>
<tr>
<td>Technical</td>
<td>Robustness</td>
<td>High, Medium, Low</td>
</tr>
<tr>
<td>Health</td>
<td>Risk of pathogens discharge</td>
<td>Very high, High, Medium, Low</td>
</tr>
</tbody>
</table>

The alternatives with chemical P removal (A4, A5, P2) had the highest energy recovery based on the larger volumes of sludge produced after the addition of chemicals and the likely higher content of organic matter present in the sludge. The alternatives with conventional septic tanks and further treatment (A1, A2, A3, S2) and the packaged plant with Polonite® filter (P1) had medium energy recovery. Alternative S1 had the lowest production of sludge and hence the lowest energy recovery, because the GW treated on-site produces smaller volumes of sludge as compared to the mixed wastewater and no biogas production was assumed for the BW treatment.

In the same line, the highest potential for nutrients recycling was attained by the source separation systems for both P and N. Alternatives S1 and S2 had 90% and 79% tot-N recycling potential respectively, in contrast to the recycling potential of about 2% tot-N of the rest of the alternatives. In terms of P recycling potential, the results ranged from the higher potential of alternatives S1 (81%) and S2 (53%) to the moderate potential of the alternatives with chemical P removal (29% for A4, A5 and P2) and the low potential for the rest of the alternatives (<9%).

The GWP and the CED were the lowest for A2 followed by A1, the standard drain field and sand filter, due to the lower use of components and consumables (extra tanks, chemicals, P-filter). The GWP for A2 and A1 was 35 and 54 kg CO2 eq. per household per year, respectively, whereas the largest values were attained by P1 and P2, with an annual emission of 104 and 95 kg CO2 eq. per household, respectively. Moreover, the CED was the highest for the alternative P2 and P1 with an annual CED of 8562 and 7627 MJ per household, more than three times the lowest CED which was attained by A2 (2403 MJ per household). For most of the alternatives, the largest contributors for both indicators were the production of the tanks and the treatment of the sludge which had a CED of approximately 1 MJ kg⁻¹ of sludge and a GWP of 0.01 kg CO2 eq. kg⁻¹ of sludge considering anaerobic digestion and dewatering processes and the Swedish electricity mix. However, the largest contributor in terms of GWP and CED for the package plants was the electricity use, as P1 and P2 consume 450 and 550 kW h per year in comparison to the rest of the alternatives whose only electricity consumption was that of the pump (7.5 kW h per year).

The alternatives with chemical P removal (A4, A5, P2) had the highest energy recovery based on the larger volumes of sludge produced after the addition of chemicals and the likely higher content of organic matter present in the sludge. The alternatives with conventional septic tanks and further treatment (A1, A2, A3, S2) and the packaged plant with Polonite® filter (P1) had medium energy recovery. Alternative S1 had the lowest production of sludge and hence the lowest energy recovery, because the GW treated on-site produces smaller volumes of sludge as compared to the mixed wastewater and no biogas production was assumed for the BW treatment.

Alternatives A1 and A2 had the lowest capital costs because of their simplicity and smaller number of components used compared to the rest of the alternatives. The source separation alternatives S1 and S2 required investments in double tanks, one for BW in S1 and one for urine in S2, and their yearly investment cost differed in approximately €100, the investment in adapted toilets being the main contributor to the difference. The cost of the ultra-low-flush vacuum toilet considered in the GW–BW separation option was much higher (€1407) than the urine diversion toilet (€456) which was only slightly more expensive than a conventional toilet. Alternatives A4 and A5 had medium capital costs of €564 and €605 per year respectively, reflecting the inclusion of the chemical dosing equipment in contrast to A1–A3 which
did not have such component. Alternative A3, P1 and P2 had
the highest capital cost; the costs associated to the Polonite®
filter bag and tank contributed in A3, whereas the purchase
of the package plants constituted the main cost for P1 and
P2. A similar pattern was observed in the indicator O&M cost.
The alternatives with the lowest yearly cost for O&M were A1
and A2 with €166, followed by the source separation alternatives
S2 (€253) and S1 (€287), which have a yearly emptying of
two tanks instead of one. The soil-based systems with polishes
had polishing step had nearly the same O&M cost, €377 for A3
and €380 for A4 and A5, the main difference being that the
Polonite® filter was exchanged every third year whereas the
chemicals needed to be purchased every year. The package
plants P1 and P2 had high yearly O&M costs due to the man-
age contracts with the providers, which included routine
maintenance such as cleaning, replacement of worn parts
and sampling of sludge and effluent water.

In terms of social acceptance, the conventional systems
(A1 and A2) had very high acceptance because of their con-
venience and low complexity as reported in the literature.6 For
the package plants (P1 and P2), acceptance was high despite
the complexity of the plants. This was because the design of
the plants made them convenient for the operators, who did
not have to monitor them regularly as management and
maintenance were assumed to be carried out by trained per-
nsonnel and were included in the management contracts (at
least once a year). However, the alternatives with chemical
dosing equipment installed inside the households (A4 and
A5) had medium acceptance, because of the inconvenience of
more frequent monitoring (e.g. refilling the dosing tank),
which may require greater effort from the homeowners than
e.g. changing a P-filter every 2–3 years as in A3. The GW and
BW system (S1) had higher acceptance than the system with
urine diversion (S2), as generally reported in the literature.65
Urine-diversion systems have been found to cause problems
with odors and inconveniences (e.g. extra cleaning) and users
require pre-knowledge about the system.56,67

The robustness was high for A1 and A2 considering that
these systems generally work well if they are correctly
designed and loaded,23 the main risk being the clogging of

<table>
<thead>
<tr>
<th>Indicators</th>
<th>Indifference threshold (q)</th>
<th>Preference threshold (p)</th>
<th>Definition approach</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tot-N removal</td>
<td>10</td>
<td>40</td>
<td>Data uncertainty</td>
</tr>
<tr>
<td>P removal</td>
<td>20</td>
<td>40</td>
<td>Data uncertainty</td>
</tr>
<tr>
<td>Potential for N recycling</td>
<td>20</td>
<td>30</td>
<td>Data range and uncertainty</td>
</tr>
<tr>
<td>Potential for P recycling</td>
<td>10</td>
<td>40</td>
<td>Data range and uncertainty</td>
</tr>
<tr>
<td>Global warming potential</td>
<td>10%</td>
<td>20%</td>
<td>EU target of 20% reduction of GHG emissions by 2020a</td>
</tr>
<tr>
<td>Cumulative energy demand</td>
<td>1000</td>
<td>3000</td>
<td>Data range and uncertainty</td>
</tr>
<tr>
<td>Energy recovery</td>
<td>0.5</td>
<td>1</td>
<td>Change in category</td>
</tr>
<tr>
<td>Investment cost</td>
<td>50</td>
<td>100</td>
<td>Data range</td>
</tr>
<tr>
<td>Operation and maintenance cost</td>
<td>50</td>
<td>100</td>
<td>Data range</td>
</tr>
<tr>
<td>Social acceptance</td>
<td>1</td>
<td>2</td>
<td>Change in category, high uncertainty</td>
</tr>
<tr>
<td>Robustness</td>
<td>0.5</td>
<td>1</td>
<td>Change in category</td>
</tr>
<tr>
<td>Risk of pathogens discharge</td>
<td>0.5</td>
<td>1</td>
<td>Change in category</td>
</tr>
</tbody>
</table>

a: (Eurostat, 2016).72

The GW and BW separation (S1) also had medium robust-
ness because even though only the GW is treated on-site, the
holding tank for BW does not adapt to flow fluctuations in
the same way as a septic tank with an outlet and the alterna-
tive requires monitoring of two tanks instead of one. The
urine diversion system (S2) and the package plants (P1 and
P2) had lower robustness based on the added complexity of
the systems; the urine-diverting toilets may present problems
with the blockage of the urine-conducting pipe69 or ventila-
tion malfunctioning, whereas the package plants generally
had an increased risk of failure due to the presence of e.g.
moving parts, sensors or electrical control systems27 and
they are often sensitive to operational disturbances.36

The GW and BW separation system (S1) had the lowest risk
of pathogen discharge to receiving waters because the feces,
which is the fraction that contains the largest pathogen load
in wastewater, was stored in a holding tank and collected
and treated in a separate facility. The results for each alternative
varied depending on the number of technical treatment bar-
riers that were included in the sanitation systems, as
discussed by e.g. Stenström et al. (2013).70 The package plants
(P1 and P2) and the sand filters with Polonite® filter (A3) and
chemical P-removal (A4) had two barriers and thus a lower risk
of pathogen discharge than the alternative A1 with only sand
filter (one barrier) and S2 (one barrier for fecal fraction), or
the alternatives A2 and A5 with drain fields (one barrier). The
risk of pathogens discharge was the highest for the drain field
without further treatment (A2) due to its single-barrier filter
material and because the receiving body was the groundwater
instead of surface water which would be more preferable.

3.2 Definition of the thresholds used in ELECTRE III method

Indifference and preference thresholds were defined
(Table 5) as required for the implementation of the ELECTRE
III method.74
For a quantitative indicator $i$, the indifference $q_i$ and preference $p_i$ thresholds were defined as an absolute value based on the uncertainty associated to the data and the range of the values across the alternatives of the indicator in question. Only the indicator GWP had both thresholds as percentage values because of the reference used.72

The thresholds established for a qualitative indicator $i$ were defined in terms of the number of categories on the scale that separated two alternatives, e.g. if low = 1, medium = 2, high = 3, then $p_i = 1$ (change in category). The higher $p_i$ ($p_i = 2$) for the qualitative indicator social acceptance reflected the greater uncertainty associated with the evaluation of the indicator.

### 3.3 Weighting the indicators

The normalized weights given to the indicators by the six members of the reference group are plotted in Fig. 2. Two indicators, namely robustness and risk of pathogen discharge received the highest normalized weights (15.5 out of 100 for each one). All members except member D assigned the highest weight of 100 points to at least one of these two indicators. Similar patterns concerning stakeholders’ preferences have been reported in the literature regarding sustainable wastewater infrastructure. For example, in Zheng et al. (2016) the stakeholders gave the highest weights to objectives related to safe (hygienic) disposal of wastewater and protection of the water resources, followed by the costs (both capital and running costs) and lastly, the social acceptance of the end users.73

### 3.4 Ranking of alternatives

The pairwise comparison between alternatives according to the different indicators resulted in the dense ranking shown in Table 6 for the baseline scenario 0. Alternative S1 (GW–BW separation) outranked the other alternatives. The outcome can be explained with the good performance of S1 on the top three most important indicators (e.g. with highest weights, Fig. 2), as the alternative had a medium robustness, the lowest risk of pathogen discharge and the highest removal of nutrients. The soil-based alternatives with sand filter (A4, A1 and A3) outranked the alternatives with drain fields, because of the drain fields’ lower performance in the indicator risk of pathogen discharge in comparison to the sand filters, given that the performance in terms of robustness was similar.

The chemical removal of P in A4 outranked the Polonite® filter as polishing step in A3 despite their similar performance in terms of robustness (both showed medium robustness), risk of pathogen discharge (low risk in both cases), and nutrients removal (30% N removal and 90% P removal for both alternatives). Their ranking was influenced by

| Scenario 0 | With original weights from reference group (e.g. northern Sweden) | S1 > A1, A4 > A3 > A2 > A5 > S2 > P2 > P1 |
| Scenario 1 | Lowest importance to nutrients-related indicators | A1, A4 > A2, S1 > A5 > A3 > S2 > P2 > P1 |
| Scenario 2 | Highest importance to nutrients-related indicators (e.g. areas with farmland) | S1 > S2 > A4 > A5 > A3, P2 > P1 (A1, A2)* |
| Scenario 3 | Highest importance to energy recovery, CED and GWP (e.g. change in political strategy) | A4, A5 > A1 > A3, P2 > A2 > S1 > S2 > P1 |

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*Alternatives A1 and A2 were excluded from the ranking in scenario 2 because they generally do not fulfill the Swedish guidelines in terms of nutrients removal.
differences in the potential for P recycling, which was higher for A4 which had larger volumes of sludge produced after the chemical P removal step (34% reuse) compared to the low P recycling from the Polonite® filters (5% reuse). Furthermore, the capital cost was higher for A3 (£761 per year) than for A4 (£564 per year), whereas the O&M costs were nearly the same (£377 and £380 respectively).

The alternative with urine diversion (S2) was ranked in position 6. Despite having high performance in terms of nutrients recycling, surpassed only by S1, all the soil-based alternatives generally performed better than S2 in terms of robustness and risk of pathogen discharge, which influenced the final ordering. The two package plants (P1 and P2) attained the last positions in the ranking, due to their weak performance on the indicator robustness and their high investment and O&M costs.

### 3.5 Scenario analysis

Scenario 1, where the indicators related to the removal and potential recycling of the nutrients were given the lowest importance, resulted in small changes in the final ranking of the alternatives compared to scenario 0 (Table 6). The standard sand filter option (A1) shared the first position together with option A4 [chemical P removal + sand filter], which was second in scenario 0. By giving the lowest weight to the two nutrients-related indicators, the other indicators increased their importance accordingly, which influenced S1 (GW-BW separation) to remain in the top positions although its good performance in terms of nutrient removal and recycling had minor impact in the ranking.

On the other hand, when the highest importance was given to the nutrient treatment and nutrient recycling potential (scenario 2), the source separation systems S1 and S2 clearly outranked the remaining alternatives (Table 6) because of their good performance on the indicators in focus.

Scenario 3 benefited the soil-based alternatives that included chemical removal of P, due to their higher potential to recovery energy and moderate GWP and CED. The source separation alternatives S1 and S2 dropped to the end of the ranking because of their relatively high GWP, mainly due to the use of extra tanks, transport of larger volumes and the N2O emissions during treatment and storage of the BW and urine. The package plants had similar GWP and CED, but differed mostly in the potential to recover energy, as P2 produced larger volumes of sludge.

### 3.6 Sensitivity analysis of the baseline scenario (scenario 0)

Decreasing the lifetime of the Polonite® filters in A3 and P1 by one year increased the O&M cost, the GWP and CED (Table 7.1). The increase in relation to the baseline scenario was larger for A3 than P1, because of the lower initial performance in the three indicators, however A3 retained the third position in the ranking.

Increasing the O&M cost for the BW and urine treatment did not affect the final ranking of the alternatives (Table 7.2), likely because the weight of the indicator was not so high. Even when the additional cost was doubled to €100 per household per year, the ordering remained unaffected. Increasing the importance of P removal to the detriment of N only affected the middle-ranked alternatives (Table 7.3). For example, A5 (drain field with chemical removal of P) outperformed alternatives A2 and A3 despite the fact that both A5 and A3 had 90% P removal. The lower costs possibly benefited A5. The changes from 100 to 0% in the reuse of BW (Table 7.4) decreased the potential to recover nutrients for S1 but also reduced the GWP because of the urea avoided. However, these changes did not affect significantly the ranking because the indicator potential for nutrients recycling had a low weight. The changes in the indicator robustness in terms of the performance (Table 7.5) or the indifference and preference thresholds (Table 7.6) did not affect significantly the top ranking of the alternatives. However, an increase in the robustness from low to medium and an increase in the indifference and preference thresholds proved to be beneficial for the urine diversion option S2 which outperformed three more alternatives as compared to scenario 0, and detrimental for A5, which dropped to the last position.

For all the tested changes, the first three solutions remained the same, indicating that the ranking is reasonably

### Table 7 Results of the sensitivity analysis

<table>
<thead>
<tr>
<th>Modified parameter</th>
<th>Ranking</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>None</td>
<td>S1 &gt; A1, A4 &gt; A3 &gt; A2 &gt;</td>
<td>Increase in O&amp;M cost, GWP and CED in A3 (28%, 16% and 7% respectively) and in P1 (19%, 7% and 3% respectively)</td>
</tr>
<tr>
<td>1. Decrease in the lifetime of Polonite® from 3 to 2 years</td>
<td>S1 &gt; A1, A4 &gt; A3 &gt; A2 &gt;</td>
<td>Increase O&amp;M cost in S1 and S2, when assuming a municipal fee of €30 and €20 hh⁻¹ y⁻¹ for BW and urine management respectively</td>
</tr>
<tr>
<td>2. Include the cost for BW and urine treatment</td>
<td>S1 &gt; A1, A4 &gt; A3 &gt; A2 &gt;</td>
<td>There are no requirements for N removal for wastewater systems for less than 10 000 PE</td>
</tr>
<tr>
<td>3. Change the weight of the importance of P (100%) and N (0%) removal</td>
<td>S1 &gt; A1, A4 &gt; A3 &gt; A2 &gt;</td>
<td>BW is collected and treated in a WWTP together with sludge instead of urea; lower potential to recycle nutrients and lower GWP</td>
</tr>
<tr>
<td>5. Increase robustness of S2, P1 and P2 from low to medium</td>
<td>S1 &gt; A1, A4 &gt; A3 &gt; A2 &gt;</td>
<td>Considering they are managed properly and less failures occur</td>
</tr>
<tr>
<td>6. Change indifferent and pref. thresholds for indicator Robustness</td>
<td>S1 &gt; A1, A4 &gt; A3 &gt; A2 &gt;</td>
<td>To reflect the uncertainty in the evaluation of robustness, the indiff. and pref. thresholds were increased to 1 and 2 respectively</td>
</tr>
</tbody>
</table>

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robust. The aforementioned alternatives dominated the ranking due to their superior performance in comparison to the rest of the options as the method applied is based on outranking relations.

4. Discussion

Assessing the sustainability of on-site sanitation systems requires understanding of the existence of trade-offs between sustainability indicators and the priorities and/or objectives of the decision-makers, e.g., operators or stakeholders. The addition of weights representing the stakeholders’ preferences implicitly adds subjectivity to the analysis, a characteristic feature of multi-criteria analysis since not all the indicators have the same importance depending on goals and background conditions. The weights given by the stakeholders (Fig. 2) defined a prioritization of the indicators. This set of weights was then modified to test various scenarios. The definition of the indicators (including their number, description and estimations) considered in this study determined the results, as the final ranking depended on how many times the alternatives outranked or became outranked by each other based on their performance. The uncertainties and assumptions related to the calculation of the alternatives’ performance were dealt with, to some extent, in the methodology, e.g., by using thresholds (Table 5), and in the sensitivity analysis (Table 7).

Blackwater separation ranked highest

The alternative with GW–BW separation (S1) outranked the other alternatives in Scenario 0 (Table 6) because it had the lowest risk of pathogen discharge and medium robustness, the best performance in terms of nutrients removal and recycling, and moderate costs. The results from the scenarios study showed that an increase in importance in the indicators energy recovery, CED and GWP, as in Scenario 3, had higher impact in the position of S1 in the final ranking (A4 and A5 would then be the most sustainable alternatives) than a change in the nutrients-related indicators, as in scenarios 1 and 2. The CED and GWP in S1 were generally higher than for all the soil-based alternatives and urine diversion, mostly due to the larger volume transported (6 m³ instead of the conventional 2.2 m³ tanks) and for the higher emissions related to urea hygienization. However, reliable estimations of emissions from BW sanitation are needed to reduce uncertainties in the calculation of their GWP.17

Including the treatment of the BW in the yearly cost to be paid by the homeowners, or assuming that the BW is not sanitized with urea but transported to a central WWTP, did not affect the first position of S1 in the final ranking (Table 6). The results indicated that BW separation was still preferred despite the introduction of an additional fee (e.g., €50) by the municipalities in order to cover the investment and operational costs of the treatment with urea, showing that there is an economic margin if management fees are to be introduced by the local authorities.

Furthermore, even if the BW was not sanitized with urea but treated in a centralized WWTP (Table 7.4), alternative S1 outranked the others, suggesting that the option to have GW–BW separation could be chosen proactively even if the municipal infrastructure for urea sanitization is not yet available. Other treatment options such as anaerobic digestion could also be considered if the aim is to increase the energy recovery in the form of biogas and reduce the emissions.7

Wet composting of BW, on the other hand, would generally require some energy input for stirring and aeration12 and to increase the temperature,7,13 and the O&M costs can be twice that of urea treatment.63

In comparison, the urine diversion alternative S2 had a higher risk of pathogen discharge since the feces were treated in a sand filter on-site. However, scenario 2 showed that urine diversion is a sustainable alternative when it is important to remove and recycle both N and P. Furthermore, diverting the urine gives the homeowner the possibility to reduce energy, emissions and costs related to collection and storage if used locally, an option not available with the other alternatives.12 The use of the wastewater fractions as fertilizers was not included in the scope of the study. However, urine is a cleaner fertilizer compared to BW, with significantly lower cadmium content, e.g., 0.6 mg cadmium per kg P compared to 11 mg in BW.17 A comprehensive assessment with a transition theory perspective reported that BW systems generally perform better than urine diversion systems because of technical malfunctioning of the latter or because the urine diversion toilets are less socially accepted than the low-flush or vacuum toilets used in the GW–BW separation systems.85

When the robustness of S2 was medium as for S1, or the preference threshold was widened, the urine diversion system improved its position in the final ranking of alternatives (Table 7.5 and 6). Improving the technology used in urine diversion systems and decreasing the failures associated to e.g. clogging of pipes would increase the robustness of the system and hence the overall sustainability.

In the baseline scenario, the ranking suggests that BW separation (S1) or a sand filter with (A4) or without chemical removal of P (A1) would be the most sustainable options of the alternatives studied. For example, A4 had a higher risk of pathogen discharge compared to S1, higher O&M costs and considerably lower removal of N. Systems like A4 require the installation of dosing equipment and a larger septic tank than a conventional one (4 instead of 2.2 m³) whereas for S1, double tanks and the installation of a low-flush vacuum toilet are required. Although the robustness of both systems was assessed as “medium,” it must be noted that the robustness of A4 will be so only if the dosing equipment is correctly managed.12

The soil-based options outranked the package plants

Soil-based systems A1–A5, outranked the urine diversion alternative S2 and the package plants (P1 and P2) in the baseline scenario (Table 6). Sand filters without P removal (A1)
outranked drain fields (A2), and the chemical removal of P (A4) was favored over the removal with Polonite® filter (A3). Sand filters and drain fields are considered to be robust systems, and economical and socially accepted partly due to their simple construction and passive functioning in comparison to package plants. However, they are not exempt of pathogenic risks and generally do not comply with the existing environmental guidelines in terms of P reduction as already concluded in previous studies. The low P removal in these systems is mainly due to the various physico-chemical mechanisms involved in the sorption (into Fe and Al oxide phases) and precipitation (of Al, Fe and Ca phosphates) reactions occurring in the filter material and soil, but also the lack of optimal design and construction in terms of hydraulics and clogging risks. Comprehensive data on their general performance and failure rate is missing or presents high variability. Hydraulic overloading and failures because of design, installation and maintenance problems (e.g. clogging) are the main reasons for their poor performance.

Alternatives with a drain field (A2 and A5) were penalized with respect to the health indicator because of their higher risk of pathogens reaching the groundwater unnoticed, as compared to the systems with an outlet pipe discharging to surface waters. If groundwater contamination was not a potential problem, A1 and A2 would not outrank each other as they had similar performance in most of the indicators.

In this study, simpler systems were assessed as “better” than advanced ones such as package plants on the indicators social acceptance and robustness based on studies reporting performance of different commercial package plants and monitoring reports. This assumption contributed to the low ranking of the alternatives with package plants P1 and P2. However, the recent development of on-site technology for package plants, which often include sensors and alarm systems to assure treatment efficiency, can make these options to actually be considered robust and reliable when managed properly resulting in high nutrients reduction. Moreover, the land area requirements were not included as indicator in the study but their inclusion would have benefited the package plants as they are more compact and typically require less area than e.g. sand filters or drain fields. Additionally, package plants can be installed in areas where bedrock, soils or fluctuating groundwater tables limit the implementation of soil-based systems. The two package plants showed large similarities in the performance of most of the indicators. Only in indicators potential for P recycling and energy recovery was P2 significantly superior to P1.

Comparing polishing steps for phosphorus removal

When the presence of sensitive receiving waters requires adequate P treatment, chemical removal of P in the septic tank and subsequent sand filter was preferred over a Polonite® filter placed after a sand filter. Other studies, reported similar results when comparing chemical precipitation systems with reactive filter material (Filtralite® and Filtra P). Weiss et al. (2008) also investigated the sensitivity of the changes in the P-filter material’s lifespan, showing that the linear relation between the filter material’s use and its lifespan explained the significant increase in the energy use of the alternatives.

The main differences between the polishing steps in terms of indicator performance were related to the potential to recycle P and recover energy, GWP and robustness. Polonite® filters can be reused on farmland although data regarding the use of by-products from OSSs is still poor, making it difficult to make accurate assumptions. The development of a legal and institutional framework for the collection and reuse of alkaline filter materials like Polonite®, which is considered a waste product after its usage and hence is managed by the corresponding authorities, would benefit alternatives like A3 and P1. Although not included in the scope of the study, there are differences in the quality of the recyclable fractions (sludge and filter material) between both options. In chemical precipitation systems, both contaminant metals and P can be found in the sludge since the metals are bound to particulate material, making it less attractive from the recycling point of view. Metals can also accumulate in Polonite® filters, although most of their content probably deposits in the septic tank and the low concentrations accumulated in the filter material would likely not restrict their use as fertilizers. Alternative A4 had around 23% lower GWP than A3. About half of the emissions during the construction phase in alternative A3 (33 out of 73 kg CO2-eq.) originated from the production of the filter material and the extra tank. However, the lower maintenance requirements make the use of Polonite® filters more convenient for the users and slightly more robust than the chemical P removal, although both A3 and A4 were assessed as having medium robustness.

Reliability of the results and limitations of the study

The reliability of the ranking procedure was analyzed through changes in the weights, which defined different scenarios, and a sensitivity analysis of selected assumptions. Increasing or decreasing the importance of the nutrients-related indicators affected the ranking only to a minor extent (only S2 improved significantly in scenario 2), whereas modifying the indicators GWP, CED and energy recovery in scenario 3 did have a greater impact on the ranking, as e.g. alternative S1 worsened its position to the third least preferred option and P2 outranked four alternatives, thus improving its position. The performance of the top alternatives on the indicators GWP, CED and energy recovery, and the indicators’ discriminatory impact, likely explains the influence in the ranking in Scenario 3. For example, S1 had better performance on the indicators with higher importance for the stakeholders, namely robustness, risk of pathogen discharge and nutrient removal, than on the indicators favored in scenario 3.

The indicator robustness influenced the final ranking greatly because of the high weight given by the reference group, but its assessment was challenging due to the variability of the
existing data and the difficulty in assessing it qualitatively. The method followed to assess the robustness of the systems was in accordance to how the stakeholders usually perceive the indicator e.g. the simpler the system is, the better and more robust. However, the validity of the method could be questioned since simpler systems also have considerable problems associated to the construction and installation, which prevent them from achieving good treatment rates.

In terms of indicators choice, the over-representation of the environmental dimension in the study was reduced during the analysis as the stakeholders placed the lowest weights to three of the five environmental indicators (Fig. 2) hence the distribution of weights was considered to be well spread among the different sustainability dimensions. Besides, the categorization of the indicators under the five main dimensions (Environmental, Economic, Socio-cultural, Technical and Health-related) could be done differently as some indicators are of different nature. For example, the Environmental indicators could be grouped into two categories: “Nutrient-related indicators”, which would include the Nutrient removal and Potential for nutrient recycling in a context of water quality and resource recovery; and “Energy-related indicators”, including the CED, GWP and Energy recovery, relevant in a context of climate change and energy efficiency.

Complete independence among indicators is difficult to verify and most analysts assume that the criteria are not all independent. Often, the most suitable criteria for a judgement of alternatives are interconnected and present multiple interactions between them. Given that all indicators may not be completely independent, the selection of an appropriate aggregation method gains great importance as some methods are more susceptible to interference than others. For example, the weighted sum (a compensatory method) is sensitive to the presence of dependent criteria in the form of ‘double-counting’ in contrast to the ELECTRE III method which uses a non-compensatory aggregation approach. The indicators used in the present study were considered to avoid double-counting as they represented separate aspects of value as described in Dodgson et al. (2009). Furthermore, the interrelationship between criteria can be assessed using different methodologies capable of handling criteria interactions and synergies, which was not included in the present study. Some methods proposed in the literature for modelling criteria interactions are decision making trial and evaluation laboratory (DEMATEL) and analytical network process (ANP) which can be combined and used as hybrid techniques to determine relationships between criteria. These models could be further applied to the present study for understanding criteria interactions together with the ELECTRE III method, as shown in previous studies dealing with multicriteria decision making.

Several issues were not included in the study boundaries, which probably had an impact on the results. The estimations for the energy recovery were based on sludge volumes as an indication, rather than on composition and content with regard to the potential for biogas production, which might have resulted in an over-simplification of the process. Furthermore, the varying nutrients’ plant availability of the different fractions (sludge, Polonite® filter material, BW, urine), as discussed elsewhere, was not considered in this study. The energy and resources that would be saved by replacing mineral fertilizers with sanitation by-products was also not taken into account, although their use contributes significantly to the energy and emissions balance.

Ordinal scores, as those used to assess the qualitative performance indicator, are well handled by compensatory methods such as ELECTRE III as they are not converted into cardinal scores, which introduce uncertainty in compensatory aggregation methods. Moreover, data uncertainty was managed by the use of indifference and preference thresholds. The focus and priorities of the decision-makers (represented by the reference group in this study) affected the ranking of alternatives as seen in the evaluation of scenarios, but only to some extent since a general pattern can be extracted from them (Table 6), as discussed in the above sections.

Finally, the optimal on-site sanitation solution will also depend on the local individual conditions (e.g. space availability, soil type and conditions, slope, groundwater table) and the operators’ personal preferences and economy.

5. Conclusions

This study assessed the sustainability of nine on-site sanitation systems following a multi-criteria approach with defined indicators and weights assigned by a group of stakeholders. The ranking of the alternatives was robust and generally changed little being considerably more sensitive to changes in the weights (scenarios) than to changes in the performance (sensitivity analysis), meaning that there is margin for data variability.

Conventional soil-based systems without polishing step generally do not comply with the existing Swedish guidelines in terms of P reduction. However, in this study, they outranked other alternatives capable of fulfilling these recommendations, indicating the importance of setting clear goals and requirements that apply in a decision-making process. When removal of P is required due to sensitive receiving waters, BW separation (S1) or chemical removal of P (A4) were preferred over Polonite® filters (A3) given that additional infrastructure needs to be implemented to facilitate the use of source-separation systems. Furthermore, in areas where nutrient removal is important (scenario 2), S1 and urine diversion (S2) were the most sustainable options. Sand filters generally outranked drain fields, which is in line with the current recommendations in terms of preferable receiving water body. Package plants have the potential to be robust systems when the technology is operated adequately and are favored in comparison to simple sand filters or drain fields where nutrients removal is prioritized. In scenario 3, the soil-based alternatives with chemical removal A4 and A5 obtained the first positions of the ranking, whereas the source separation alternatives worsened their
positions. Since the ranking was influenced by the performance on indicators related to emissions and energy use, further research including the substitution of synthetic fertilizers would be needed to obtain a more complete picture. The results also showed that the sustainability of urine diversion systems would increase considerably if they were more user-friendly and robust, e.g. lower failure associated to clogging of pipes and odors.

Improved estimations and data on the performance of the OSSs, emissions and social acceptance are needed for more accurate evaluations and estimations of the indicators. Determining the most sustainable alternatives will depend on the trade-offs and main focus or objectives of the decision-maker, as well as on the existing regulations and local conditions. Overall, the methodological approach of ELECTRE III proved to be suitable for the assessment of sanitation alternatives with regard to their sustainability, as both qualitative and quantitative indicators were used in this study. Furthermore, the use of thresholds contributed to dealing with data uncertainty. The methodological framework and resulting ranking of alternatives could be used to support decision-making processes concerning sanitation systems.

Conflicts of interest

There are no conflicts to declare.

Acknowledgements

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Supplementary material

Assessing the sustainability of on-site sanitation systems using multi-criteria analysis

Brenda Vidal 1,*, Annelie Hedström 1, Sylvie Barraud 2, Erik Kärrman 3 and Inga Herrmann 1

1. Detailed description of system alternatives

A1. Septic tank and sand filter

Wastewater from the household is collected in a three-chamber septic tank made of fiberglass-reinforced polyester (FRP) and pumped to a sand filter constructed below ground surface. The wastewater is spread using perforated distribution pipes and treated through a variety of physical, chemical and biochemical reactions and processes as it infiltrates. The effluent is collected at the bottom of the filter by drainage pipes. A conventional sand filter constructed according to the Swedish standard was assumed (Swedish EPA, 2003a), with a layer of filter media of 0.8 m (sand/gravel between 0 to 8 mm in size) and a surface area of 30 m². Both distribution and drainage pipes are embedded in a 0.2 m coarse gravel layer. The sludge from the septic tank is transported to the nearest WWTP for anaerobic treatment.

A2. Septic tank and drain field

A design similar to A1 is assumed. However, the wastewater is not collected after the buried infiltrative material, a layer of coarse gravel of 35 cm (Swedish EPA, 2003b), but instead continues infiltrating and percolating through the underlying soil. It was assumed that the properties of the soil allow infiltration of the wastewater.

A3. Septic tank, sand filter and P-filter

Same design as in alternative A1 with the addition of a polishing step for phosphorus (P) removal (alkaline P-filter), using the filter media Polonite® after the sand filter. The water is transported from the outlet of the sand filter to the P-filter by gravity.

A4. Septic tank with chemical P removal, sand filter

Wastewater from the house is collected in a three-chamber septic tank. A chemical precipitation unit with polyaluminum chloride is installed inside the household (e.g. under the kitchen sink) and dosed continuously. The flocculation and sedimentation occurs in the septic tank and therefore the tank volume is larger (4 m³) than in alternatives A1-3. The septic sludge is assumed to contain more P than in A1-3. The lifespan of the sand filter was assumed to be higher (25 years instead of 20 years as in A1-3), because some of the suspended solids and the BOD are removed during the flocculation process resulting in a lower load to the sand filter (Palm et al., 2002; Weiss et al., 2008).

A5. Septic tank with P precipitation and drain field

Same design as in A4 though in this case the wastewater is not collected after the filter media, a layer of gravel (between 12 to 32 mm in size) of 35 cm (Swedish EPA, 2003b), but instead continues
infiltrating into the soil. It was assumed that the properties of the soil allow infiltration of the wastewater.

S1. Greywater and blackwater separation

This system is based on the separate collection of greywater (GW) and blackwater (BW). The GW was assumed to be collected in a septic tank followed by a sand filter. The lifespan of the sand filter was assumed to be higher than the conventional one (25 years instead of 20), because most of the nutrients and some BOD are already removed with the BW. The sludge from the GW tank is transported to a WWTP. The BW is collected in a holding tank which is emptied once a year. For collection, storage and treatment reasons it is important that the volume of the WCs’ flush is small and therefore the use of a low-flush toilet (about 1 L per flush) was assumed in the analysis. The collected blackwater is transported to a central treatment facility using urea treatment (1% urea) for hygienization.

S2. Greywater and feces, urine diversion (UD).

This system is based on the separate collection of GW, urine and feces fractions from separating toilets. The GW and the feces are collected together in a septic tank and subsequently conveyed to a sand filter. The sludge from the septic tank is transported to a WWTP. The urine is collected in a container and transported to a centralized facility for hygienization (six-month storage).

P1. Package plant 1

This package plant consists of one single unit buried underground to which the wastewater from the household is transported by gravity. The mechanical treatment occurs in the first three chambers of the plant, followed by two bioreactors with aerators that provide oxygen to the water. A fraction of the water is returned to the first chamber. The effluent infiltrates through an alkaline P-filter with Polonite® material placed in a bag in the center of the plant. The sludge is collected from the three chambers that act as septic tanks, and it is transported to the nearest WWTP for anaerobic treatment.

P2. Package plant 2

This package plant consists of one single unit buried underground were the wastewater from the household is transported to by gravity. The plant operates in a 2-phases semi-continuous regime. The main phase corresponds to a continuous process of activated sludge, and the second phase corresponds to the regulation of excess sludge and self-cleaning of the plant when the level of wastewater in the equalization tank is low. The raw wastewater is first collected in an equalization tank and then pumped to an aerated water-processing tank with activated sludge where primary settlement occurs and a chemical dosing equipment is installed. A small sand filter is used as a final polishing step. All the sludge is collected in a separated tank inside the unit and further collected and transported to the nearest WWTP.
Table S1.1 Summary of components included in each alternative.

<table>
<thead>
<tr>
<th>Components</th>
<th>Sand filter</th>
<th>Drain field</th>
<th>Sand filter + P-filter</th>
<th>Chemical P removal + sand filter</th>
<th>Chemical P removal + drain field</th>
<th>Greywater/Blackwater separation</th>
<th>Urine diversion</th>
<th>P1 (P-filter)</th>
<th>P2 (chem. precipitation)</th>
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<tbody>
<tr>
<td>Abbreviation --&gt;</td>
<td>A1</td>
<td>A2</td>
<td>A3</td>
<td>A4</td>
<td>A5</td>
<td>S1</td>
<td>S2</td>
<td>P1</td>
<td>P2</td>
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<td>Septic tank 2.2 m³</td>
<td>x</td>
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<td>Septic tank 4 m³</td>
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<td></td>
</tr>
<tr>
<td>Septic tank 1.2 m³</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Holding tank 6 m³</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Storage tank 3 m³</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sand filter (20 years lifespan)</td>
<td>x</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Sand filter (25 years lifespan)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Drain field (20 years lifespan)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Distribution box</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td></td>
<td>x</td>
<td></td>
</tr>
<tr>
<td>Inspection box</td>
<td>x</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Polonite ® bag</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
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<tr>
<td>Tank for Polonite ® bag</td>
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<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Pump</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Chemical dosing equipment</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
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<tr>
<td>Chemicals (PIX)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td></td>
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<tr>
<td>Urea</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Vacuum toilet</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Urine diversion toilet</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Package plant</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>x</td>
</tr>
</tbody>
</table>
2. **Input data and assumptions**

Table S2.1 Assumptions for the estimations of the nutrients removal for each alternative.

<table>
<thead>
<tr>
<th>Alternatives</th>
<th>P</th>
<th>N</th>
<th>Main sources and comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>A1. Septic tank + Sand filter</td>
<td>40%</td>
<td>30%</td>
<td>40±20% P and 25±10% N (Olshammar et al., 2015); 50% P and 50% N in sand filters (Urban Water, 2011); 10-80% P and 10-40% N in sand filters (Palm et al., 2002); 8-16% P removal in sand filters (Eveborn et al., 2012)</td>
</tr>
<tr>
<td>A2. Septic tank + Drain field</td>
<td>40%</td>
<td>35%</td>
<td>50±30% P and 30±10% N (Olshammar et al., 2015); 96% P and 40-60% N in drain fields (Urban Water, 2011)</td>
</tr>
<tr>
<td>A3. Septic tank + Sand filter + P-filter (Polonite®)</td>
<td>90%</td>
<td>30%</td>
<td>75±20% P in P-filters (Olshammar et al., 2015); 96.7% in column study with Polonite® (Gustafsson et al., 2008); 89-97% P removal in column and full scale systems with Polonite® (Renman and Renman, 2010); 43-99% P reduction in full-scale Polonite® filters (Vidal et al., 2018)</td>
</tr>
<tr>
<td>A4. Septic tank + chemical P removal + sand filter</td>
<td>90%</td>
<td>30%</td>
<td>90% P removal with chemical precipitation when it functioned (Hellström and Jonsson, 2006); 85% P removal in septic tanks with chemical precipitation</td>
</tr>
<tr>
<td>A5. Septic tank chemical P removal + drain field</td>
<td>90%</td>
<td>35%</td>
<td>Combination of previous assumptions</td>
</tr>
<tr>
<td>S1. Greywater to sand filter; Blackwater to urea treatment</td>
<td>90%</td>
<td>95%</td>
<td>80% P and 90% N are removed with the BW collection; around 10-20% of remaining N is expected to be reduced in the sand filter (Lennartsson et al., 2009)</td>
</tr>
<tr>
<td>S2. Greywater + faeces to sand filter, urine diversion to storage</td>
<td>80%</td>
<td>85%</td>
<td>A reduction of 35-50% P and 50-70% N is expected when urine is diverted (Palm et al., 2002). Upper values were assumed.</td>
</tr>
<tr>
<td>P1. Package plant with P-filter Polonite®</td>
<td>90%</td>
<td>40%</td>
<td>80±5% P and 40±5% N for package plants in general (Olshammar et al., 2015). See references in A3 for Polonite® filter material.</td>
</tr>
<tr>
<td>P2. Package plant with chemical precipitation</td>
<td>90%</td>
<td>40%</td>
<td>80±5% P and 40±5% N for package plants in general (Olshammar et al., 2015). See previous references for chemical precipitation of P.</td>
</tr>
<tr>
<td>Basic process</td>
<td>Cumulative Energy Demand (CED)</td>
<td>Global Warming Potential (GWP)</td>
<td>Data sources and assumptions</td>
</tr>
<tr>
<td>-------------------------------------------------------------------------------</td>
<td>-------------------------------</td>
<td>-------------------------------</td>
<td>-----------------------------</td>
</tr>
<tr>
<td>Excavation of 1 m³ of sandy soil with 15 kW excavator</td>
<td>19.3</td>
<td>1.3</td>
<td>Oekobaudat 2016 database (BMUB, 2016), Process 9.1.01 “Bagger 15K Aushub”</td>
</tr>
<tr>
<td>Transport of 1 t of good over 1 km by garbage truck (collecting route)</td>
<td>8.00</td>
<td>0.07</td>
<td>Adjusted from (Sonesson, 1996)</td>
</tr>
<tr>
<td>Transport of 1 t of good over 1 km by medium weight truck</td>
<td>1.0</td>
<td>0.07</td>
<td>ELCD 3.2 database, Process “Lorry transport”</td>
</tr>
<tr>
<td>Transport of 1 t of good over 1 km by heavy weight truck</td>
<td>0.7</td>
<td>0.05</td>
<td>ELCD 3.2 database, Process “Articulated lorry transport”</td>
</tr>
<tr>
<td>Transport of 1 t of good over 1 km with sea cargo</td>
<td>0.2</td>
<td>0.01</td>
<td>ELCD 3.2 database, Process “Electricity grid mix, consumption mix, at consumer, AC, 230V/52”</td>
</tr>
<tr>
<td>Production of 1 kg of gravel and sand</td>
<td>0.006</td>
<td>0.0007</td>
<td>Report from the IVL Swedish Environmental Institute (Stripple, 2001), pp 48.</td>
</tr>
<tr>
<td>Production of 1 kg of macadam (12-24, 16-32 mm)</td>
<td>0.07</td>
<td>0.001</td>
<td>ELCD 3.2 database, Process “Lorry transport”</td>
</tr>
<tr>
<td>Production of 1 kg of material-separating layer (macadam 4-8 mm)</td>
<td>0.02</td>
<td>0.0004</td>
<td>Based on Erlandsson (2010)</td>
</tr>
<tr>
<td>Production of 1 MJ of electricity in Sweden</td>
<td>2.1</td>
<td>0.02</td>
<td>ELCD 3.2 database, Process “Electricity grid mix, consumption mix, at consumer, AC, 230V/52”</td>
</tr>
<tr>
<td>Production of 1 kg of PE (HD)</td>
<td>77.0</td>
<td>1.95</td>
<td>ELCD 3.2 database, Process “Polyethylene high density granulate (PE-HD), production mix, at plant”</td>
</tr>
<tr>
<td>Production of 1 kg of PP</td>
<td>75.0</td>
<td>1.98</td>
<td>ELCD 3.2 database, Process “Polypropylene granulate (PP), production mix, at plant”</td>
</tr>
<tr>
<td>Production of 1 kg of sheet moulding reinforced component glass-fiber polyester</td>
<td>41.3</td>
<td>1.8</td>
<td>ELCD 3.2 database, Process “Electricity grid mix, consumption mix, at consumer, AC, 230V/52”</td>
</tr>
<tr>
<td>Production of 1 kg of steel</td>
<td>12.8</td>
<td>1.0</td>
<td>ELCD 3.2 database, Process “Steel sections (ILCD), production mix, at plant, blast furnace route / electric arc</td>
</tr>
<tr>
<td>Production of 1 kg of PVC</td>
<td>60.4</td>
<td>1.6</td>
<td>ELCD 3.2 database, Process “Polyvinylchloride resin (S-PVC), production mix, at plant, suspension”</td>
</tr>
<tr>
<td>Production of 1 m² of geotextile</td>
<td>2.33</td>
<td>0.11</td>
<td>Calculated based on specific heat capacity of the opoka rock assumed 0.97 kJ/kg*K (Romushkevich et al., 2004)</td>
</tr>
<tr>
<td>Production of 1 kg of PAX</td>
<td>2.67</td>
<td>0.1</td>
<td>Calculated based on specific heat capacity of the opoka rock assumed 0.97 kJ/kg*K (Romushkevich et al., 2004)</td>
</tr>
</tbody>
</table>
opoka rock is heated at 900 °C (Brogowski and Renman, 2004) in kilns (80% efficiency assumed) fueled with natural gas (Sammeli, personal communication, June 2017). Energy needed (ELCD3.2) to produce 1 kg of natural gas= 55.8MJ/kg; natural gas LHV= 44.1MJ/kg and HHV=52.21MJ/kg. Emissions calculated based on ELCD3.2.

Treatment of 1 kg of sludge in average WWTP

Based on energy use and emissions from Hospido et al. (2005) for thickened mixed sludge (including the polymer manufacture) with a dry matter content of 10 g L⁻¹, although the sludge from septic tanks is more diluted and normally has lower dry matter content e.g. around 4-6% (Hedström and Hanaeus, 1999).

<table>
<thead>
<tr>
<th>Component name</th>
<th>Comment</th>
<th>Data sources and assumptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>N₂O emissions from BW (treated with urea 1%) and urine storage</td>
<td>NH₃-N emissions: 5% tot-N</td>
<td>as in Karlsson and Rodhe (2002) assuming covered storage. Same factor used for both BW and urine due to their similar low viscosity and dry matter content (Spångberg et al., 2014). 95% of NH₃-N in the BW (Jönsson et al., 2005) and 100% in the urea (46% N), as the urea is degraded into ammonium by urease-producing bacteria (Nordin, 2010)</td>
</tr>
<tr>
<td>Indirect N₂O emissions: 1 % NH₃-N emitted to air</td>
<td>(IPCC, 2006)</td>
<td></td>
</tr>
<tr>
<td>N₂O emissions from sludge storage</td>
<td>NH₃-N emissions: 10% tot-N</td>
<td>as for semi-solid manure (Karlsson and Rodhe, 2002)</td>
</tr>
<tr>
<td>Indirect N₂O emissions: 1 % NH₃-N emitted to air</td>
<td>(IPCC, 2006)</td>
<td></td>
</tr>
</tbody>
</table>

Table S2.4 Summary of assumptions for the calculation of the indicators capital cost (components + man power) and operation & maintenance costs.

<table>
<thead>
<tr>
<th>Capital cost</th>
<th>Prices without VAT</th>
<th>Comments and references</th>
</tr>
</thead>
<tbody>
<tr>
<td>Man power (1 worker + machinery)</td>
<td>80 €/ h</td>
<td>5, 3.5 and 2 days of work (6 hours/day) assumed for installing sand filters/drain fields, package plants, and holding tanks, respectively. Personal communication with Mikael Samuelsson (Sep 2018) <a href="http://www.samuelssonmekaniska.se/kontakt.php">http://www.samuelssonmekaniska.se/kontakt.php</a></td>
</tr>
<tr>
<td>Man power (1 worker)</td>
<td>48 €/ h</td>
<td></td>
</tr>
<tr>
<td>Tanks</td>
<td>Septic tank 2.2 m³</td>
<td>1384 €</td>
</tr>
<tr>
<td>Component</td>
<td>Cost</td>
<td>Description</td>
</tr>
<tr>
<td>----------------------------------</td>
<td>-------</td>
<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Septic tank 4 m³</td>
<td>2160 €</td>
<td>For mixed wastewater and chemical removal of P.</td>
</tr>
<tr>
<td>Septic tank 1.2 m³</td>
<td>920 €</td>
<td>For greywater.</td>
</tr>
<tr>
<td>Equipment for sand filter field</td>
<td>120 €</td>
<td>Assumed based on</td>
</tr>
<tr>
<td>Basic package</td>
<td>120 €</td>
<td>Assumed based on</td>
</tr>
<tr>
<td>Soil test</td>
<td>750 €</td>
<td>To check the suitability of the soil for infiltration. Assumed based on</td>
</tr>
<tr>
<td>Sand and gravel</td>
<td>12 €/ton</td>
<td>Assumed based on</td>
</tr>
<tr>
<td>Polonite® bag</td>
<td>600 €</td>
<td>Assumed based on</td>
</tr>
<tr>
<td>Tank for filter</td>
<td>1320 €</td>
<td>Assumed based on</td>
</tr>
<tr>
<td>Dosing equipment</td>
<td>776 €</td>
<td>Assumed based on</td>
</tr>
<tr>
<td>Components for chemical P removal</td>
<td>180 €</td>
<td>Assumed based on</td>
</tr>
<tr>
<td>Toilets (only the difference in cost with standard)</td>
<td>345 €</td>
<td>Assumed average price based on</td>
</tr>
</tbody>
</table>
respect to standard toilets was included in the calculations)

|-----------------------------|--------|--------|--------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|

**Operation & Maintenance cost**

<table>
<thead>
<tr>
<th>Electricity cost</th>
<th>0.078 €/kWh</th>
<th>Electricity network + supplier. <a href="https://www.vattenfall.se/elavtal/elpriser/rorligt-elpris/">https://www.vattenfall.se/elavtal/elpriser/rorligt-elpris/</a></th>
</tr>
</thead>
<tbody>
<tr>
<td>Annual sampling of effluent</td>
<td>76 €</td>
<td>By an external accredited company, for BOD7, Tot-P, Tot-N, E.coli and pH. <a href="https://www.vattenproptagning.se/analys/kontrollpaket-enskilt-avlopp">https://www.vattenproptagning.se/analys/kontrollpaket-enskilt-avlopp</a></td>
</tr>
</tbody>
</table>

**Package plant 1**

<table>
<thead>
<tr>
<th>Service agreements</th>
<th>176 €/year</th>
<th>Includes one visit per year, change of wear parts and effluent sampling (pH and P concentrations). Personal communication with Karl-Gustav (Feb 2018) <a href="http://www.ecot.se">http://www.ecot.se</a></th>
</tr>
</thead>
<tbody>
<tr>
<td>Operation and maintenance</td>
<td>240 €/year</td>
<td>Change of Polonite® every third year and electricity are included. <a href="http://vaguiden.se/marknadsoversikt-2016/">http://vaguiden.se/marknadsoversikt-2016/</a></td>
</tr>
</tbody>
</table>

**Package plant 2**

<table>
<thead>
<tr>
<th>Service agreements</th>
<th>240 €/year</th>
<th>Includes one visit per year, change of wear parts and sludge and effluent sampling (pH and P concentrations). <a href="http://vaguiden.se/marknadsoversikt-2016/">http://vaguiden.se/marknadsoversikt-2016/</a></th>
</tr>
</thead>
<tbody>
<tr>
<td>Operation and maintenance</td>
<td>120 €/year</td>
<td>Electricity, chemicals and consumables included. <a href="http://vaguiden.se/marknadsoversikt-2016/">http://vaguiden.se/marknadsoversikt-2016/</a></td>
</tr>
</tbody>
</table>

**Emptying septic/holding tank**

<table>
<thead>
<tr>
<th>Size (m³)</th>
<th>Cost (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt; 3 m³</td>
<td>87 €</td>
</tr>
<tr>
<td>3.1 – 6 m³</td>
<td>121 €</td>
</tr>
<tr>
<td>6 – 9 m³</td>
<td>170 €</td>
</tr>
</tbody>
</table>

Prices can vary between 60 – 170 € [http://husagare.avloppsguider.se/sluten-tank-och-kompaktfilter-f%C3%B6r-bdt-.html](http://husagare.avloppsguider.se/sluten-tank-och-kompaktfilter-f%C3%B6r-bdt-.html) Assumed values taken from Bollnäs municipality for extractions with hoses up to 10 meters long [http://www.bollnas.se/index.php/slamtoemningstaxa](http://www.bollnas.se/index.php/slamtoemningstaxa)

The capital cost was calculated based on the present value (PV) of the different components, materials and services needed for the installation of the OSS, multiplied by the annuity, which was accounted based on the following formula (Brealey et al., 2012):

\[
PV = \frac{C \left(1 - \left(1 + r\right)^{-n}\right)}{r}
\]

Where \( r \) = interest rate (assumed 4%) and \( n \) = amortization time (years) of each component

Table S2.5 Assumptions for the assessment of the qualitative indicator *social acceptance*

<table>
<thead>
<tr>
<th>Alternatives</th>
<th>Social acceptance</th>
<th>Notes and references</th>
</tr>
</thead>
<tbody>
<tr>
<td>A1. Septic tank + Sand filter</td>
<td>Very high</td>
<td>It is one of the most common systems in Sweden (Olshammar et al., 2015), it does not require much effort to operate and maintain it</td>
</tr>
<tr>
<td>A2. Septic tank + Drain field</td>
<td>Very high</td>
<td>It is one of the most common systems in Sweden (Olshammar et al., 2015), it does not require much effort to operate and maintain it</td>
</tr>
<tr>
<td>System Description</td>
<td>Acceptance</td>
<td>Notes</td>
</tr>
<tr>
<td>--------------------</td>
<td>------------</td>
<td>-------</td>
</tr>
<tr>
<td>A3. Septic tank + Sand filter + P-filter (Polonite®)</td>
<td>High</td>
<td>The P-filter needs to be changed every 2-4 years (3 years assumed).</td>
</tr>
<tr>
<td>A4. Septic tank + P precipitation + sand filter</td>
<td>Medium</td>
<td>The dosing equipment for the precipitation of P is placed inside the house, normally under the sink. Re-filling of the chemical must be done regularly, and it has been reported in the literature that often house owners forget to check it up and refill it (Hellström and Jonsson, 2006).</td>
</tr>
<tr>
<td>A5. Septic tank + P precipitation + drain field</td>
<td>Medium</td>
<td>The dosing equipment for the precipitation of P is placed inside the house, normally under the sink. Re-filling of the chemical must be done regularly, and it has been reported in the literature that often house owners forget to check it up and refill it (Hellström and Jonsson, 2006).</td>
</tr>
<tr>
<td>S1. Greywater to sand filter; Blackwater to urea treatment</td>
<td>High</td>
<td>The only inconvenience for the house owners is the fact that they have to empty two tanks, the septic tank with GW and the holding tank with BW, instead of only one. But the emptying can be done at the same time, e.g. once a year. This system has generally been more socially accepted than urine-diversion (McConville et al., 2017).</td>
</tr>
<tr>
<td>S2. Greywater + faeces to sand filter, urine diversion to storage</td>
<td>Low</td>
<td>For the diverting toilet to function properly, individual acceptance and knowledge is required. The diverting toilet could involve more frequent cleaning and maintenance and problems with odors and blockages may arise (Larsen et al., 2009). Moreover, it could be less convenient for visitors or guests.</td>
</tr>
<tr>
<td>P1. Package plant with P-filter Polonite®</td>
<td>High</td>
<td>It does not have the highest social acceptance because it requires more monitoring than the soil-based systems, as it contains many different components. The risks related to equipment failure (e.g. aeration, sensors, etc.) are higher and will require more attention from the owner (e.g. call the technicians to come).</td>
</tr>
<tr>
<td>P2. Package plant with chemical precipitation</td>
<td>High</td>
<td>It does not have the highest social acceptance because it requires more monitoring than the soil-based systems, as it contains many different components. The risks related to equipment failure (e.g. aeration, sensors, etc.) are higher and will require more attention from the owner (e.g. call the technicians to come).</td>
</tr>
</tbody>
</table>
Table S2.6 Assumptions for the assessment of the qualitative indicator robustness. Definition of robustness was based on previous descriptions from Spiller et al. (2015).

<table>
<thead>
<tr>
<th>Alternatives</th>
<th>Robustness</th>
<th>Notes and references</th>
</tr>
</thead>
<tbody>
<tr>
<td>A1. Septic tank + Sand filter</td>
<td>High</td>
<td>These systems work well if they are correctly designed and loaded, but they often lack good design and construction from the start, which influences their initial and long term performance (Palm et al., 2002; Vidal et al., 2018). The P-removal decreases with time as the sand gest exhausted (Wilson et al., 2011).</td>
</tr>
<tr>
<td>A2. Septic tank + Drain field</td>
<td>High</td>
<td>These systems work well if they are correctly designed and loaded, but they often lack good design and construction from the start, which influences their initial and long term performance (Palm et al., 2002; Vidal et al., 2018). The P-removal decreases with time as the sand gest exhausted (Wilson et al., 2011).</td>
</tr>
<tr>
<td>A3. Septic tank + Sand filter + P-filter (Polonite®)</td>
<td>Medium</td>
<td>If the preceding sand filter is clogged it will affect the subsequent P-filter by clogging it. However, The P-adsorbing compounds are part of the filter media in contrast to the chemical precipitation options where it needs to be added (Jenssen et al., 2010).</td>
</tr>
<tr>
<td>A4. Septic tank + P precipitation + sand filter</td>
<td>Medium</td>
<td>The dosing equipment often presents problems e.g., precipitants are not refilled and hinders the P removal, hence frequent monitoring is necessary for it to be reliable (Hellström and Jonsson, 2003). If the correct dose of the coagulants and the pH are not sufficient at all times, the high removal of P will not be consistent (Jenssen et al., 2010), and the performance could be unstable.</td>
</tr>
<tr>
<td>A5. Septic tank + P precipitation + drain field</td>
<td>Medium</td>
<td>The dosing equipment often presents problems e.g., precipitants are not refilled) and hinders the P removal, hence frequent monitoring is necessary for it to be reliable (Hellström and Jonsson, 2003)</td>
</tr>
<tr>
<td>S1. Greywater to sand filter; Blackwater to urea treatment</td>
<td>Medium</td>
<td>The low flush toilets might cause problems (e.g., clogging) as compared to normal toilets</td>
</tr>
<tr>
<td>S2. Greywater + faeces to sand filter, urine diversion to storage</td>
<td>Low</td>
<td>The urine diverting toilet might cause problems (e.g. clogging, odors)</td>
</tr>
<tr>
<td>P1. Package plant with P-filter Polonite®</td>
<td>Low</td>
<td>The P-adsorbing compounds are part of the filter media in contrast to the chemical precipitation options where it needs to be added (Jenssen et al., 2010).</td>
</tr>
<tr>
<td>P2. Package plant with chemical precipitation</td>
<td>Low</td>
<td>The dosing equipment often presents problems (e.g., precipitants are not refilled) and hinders the P removal, hence frequent monitoring is necessary for it to be reliable (Hellström and Jonsson, 2003)</td>
</tr>
</tbody>
</table>
Jonsson, 2003). If the correct dose of the coagulants and the pH are not sufficient at all times, the high removal of P will not be consistent (Jenssen et al., 2010), hence the performance could be unstable.

Table S2.7 Assumptions for the assessment of the qualitative indicator *risk of pathogen discharge*. Estimations based on the number of barriers present in each system alternative and considering previous research on pathogens removal in sand filters (Herrmann et al., 2017; Kauppinen et al., 2014), P-filters (Herrmann et al., 2017; Jenssen et al., 2010; Nilsson et al., 2013), source separation systems (Larsen and Maurer, 2011; Palm et al., 2002) and package plants.

<table>
<thead>
<tr>
<th>Alternatives</th>
<th>Risk of pathogen discharge</th>
<th>Notes and references</th>
</tr>
</thead>
<tbody>
<tr>
<td>A1. Septic tank + Sand filter</td>
<td>Medium</td>
<td>1 barrier (sand filter) (Herrmann et al., 2017; Kauppinen et al., 2014) reported great variability in the quality of the effluent of the sand filters investigated.</td>
</tr>
<tr>
<td>A2. Septic tank + Drain field</td>
<td>High</td>
<td>1 barrier (soil) but the treated water infiltrates directly into the groundwater, which increases the risk of affecting it negatively as it is often the local source of drinking water (Palm et al., 2002).</td>
</tr>
<tr>
<td>A3. Septic tank + Sand filter + P-filter (Polonite)</td>
<td>Low</td>
<td>2 barriers: the sand filter and P-filter; (Jenssen et al., 2010) reported good results in saturated filter beds with Filtralite®P, a light-weight expanded clay aggregate with high P sorption capacity. (Nilsson et al., 2013) reported high reduction rate of Enterococci and E.coli, although it decreased with time. The bacteria reduction in Polonite® could be attributed to the high pH, “followed by straining in smaller pores created by clogging”</td>
</tr>
<tr>
<td>A4. Septic tank + P precipitation + sand filter</td>
<td>Low</td>
<td>By removing particulate matter with the chemical precipitation the pathogen removal also increases as they are often associated to particles (Jiménez et al., 2009).</td>
</tr>
<tr>
<td>A5. Septic tank + P precipitation + drain field</td>
<td>Medium</td>
<td>1 barrier (soil) but the treated water infiltrates directly into the groundwater, which increases the risk of affecting it negatively as it is often the local source of drinking water (Palm et al., 2002). By removing particulate matter with the chemical precipitation the pathogen removal also increases as they are often associated to particles (Jiménez et al., 2009).</td>
</tr>
<tr>
<td>S1. Greywater to sand filter; Blackwater to urea treatment</td>
<td>Very low</td>
<td>The only discharge of wastewater is from GW, which does contain pathogens (Larsen and Maurer, 2011); however most of the pathogen load is contained in the BW which is collected and taken for centralized sanitation somewhere else.</td>
</tr>
<tr>
<td>S2. Greywater + faeces to sand</td>
<td>Medium</td>
<td>1 barrier (sand filter) as in A1. Most of the pathogens are contained in the feces as the urine is</td>
</tr>
<tr>
<td>Method</td>
<td>Barrier Level</td>
<td>Description</td>
</tr>
<tr>
<td>-----------------------------------------------------------------------</td>
<td>---------------</td>
<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>filter, urine diversion to storage</td>
<td></td>
<td>considered almost sterile</td>
</tr>
<tr>
<td>P1. Package plant with P-filter Polonite ®</td>
<td>Low</td>
<td>2 barriers, the biological step (with aeration) and the P-filter</td>
</tr>
<tr>
<td>P2. Package plant with chemical precipitation</td>
<td>Low</td>
<td>1 barrier, the biological step (with aeration) but also the P-precipitation</td>
</tr>
</tbody>
</table>
### 3. Summary of input data for the assignation of weights

Table S3.1 Weights given by the different members of the reference group for each indicator and arithmetic mean for each indicator (normalized weight).

<table>
<thead>
<tr>
<th>Indicators</th>
<th>Members of the reference group</th>
<th>Normalized weight</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>A</td>
<td>B</td>
</tr>
<tr>
<td>Robustness</td>
<td>75</td>
<td>85</td>
</tr>
<tr>
<td>Risk of pathogen discharge</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>Nutrients removal (N and P)</td>
<td>65</td>
<td>85</td>
</tr>
<tr>
<td>Capital cost</td>
<td>30</td>
<td>40</td>
</tr>
<tr>
<td>O &amp; M cost</td>
<td>35</td>
<td>40</td>
</tr>
<tr>
<td>Potential for nutrients recycling</td>
<td>55</td>
<td>60</td>
</tr>
<tr>
<td>Social Acceptance</td>
<td>50</td>
<td>80</td>
</tr>
<tr>
<td>Cumulative energy demand</td>
<td>20</td>
<td>30</td>
</tr>
<tr>
<td>Global Warming Potential</td>
<td>25</td>
<td>20</td>
</tr>
<tr>
<td>Energy recovery</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td><strong>Total weights</strong></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Figure S3.1 Representation of Simos cards method for the assignment of weights for each scenario (1, 2 and 3). The empty white boxes represent blank cards as described in the methodology (Figueira and Roy, 2002; Simos, 1990). The original weights of Scenario 0, the baseline scenario, were given by the stakeholders but were also transformed with the Simos card method for further definition of the scenarios.
References


Paper V

Vidal, B., Sharp, L., Hedström, A., Herrmann, I.

Examining the centralization heuristic in Swedish peri-urban and rural wastewater management

Manuscript
Abstract

Global questions such as population growth, climate change, eutrophication or scarcity of resources challenge the existing wastewater management systems. The centralized wastewater management paradigm is based on a linear model with strong lock-ins where wastewater is collected, treated and disposed. Alternative local or decentralized sanitation solutions with focus on resource recovery, such as urine or blackwater separation could be considered in newly built peri-urban and rural areas to increase the recovery of resources and tackle some of the current global challenges. The aim of this study was to examine the heuristics in the Swedish wastewater sector by investigating which aspects are considered by municipalities in the planning process when certain areas need (improved) sanitation services. Interviews with representatives from local municipalities and water utilities, official statistics and scientific literature were use as source material. Centralising the wastewater systems by building long sewer networks and investing in the upgrade of the existing main wastewater treatment plants was found to be part of the sector’s heuristics. Influential reasons mentioned by the interviewees included the robustness of the system, simplicity of operation and maintenance and protection of the receiving waters. Planning wastewater systems differently as, for example, implementing a decentralized solution or a source-separating system, was driven by resource constraints (e.g., water shortage), sustainability profiling, strong leadership and political will. Including alternative systems with focus on resource recovery during the planning phase was rarely done, although one example is presented. Identified barriers hindering implementation of alternative sanitation solutions included legislative factors (lack of requirements and law interpretation), technical factors (immature technologies, uncertainties), organizational factors (lack of initiative, competence and experience), economic factors (financial limitation, lack of incentives) and time constraints. The paper further highlights the need of having common visions based on strong drivers and real needs, the engineers’ perceptions of decentralized wastewater treatment systems and the importance of including alternative solutions during the initial assessment or evaluation phase when different options are compared.

Keywords: decentralized, innovative systems, alternative sanitation, small scale, source-separating, urine diversion, blackwater, greywater, decision-making, wastewater management
Examining the centralization heuristic in Swedish peri-urban and rural wastewater management

1. Introduction

When wastewater treatment strategies were first developed in the 20th century, treatment plants were located close to the urban areas where the concentration of polluted water was causing the most acute problems. Since then, ‘progress’ has been associated with the continued roll-out of wastewater services into increasingly rural landscapes. Typically, localities were first served by a basic and very local treatment system, but as further development occurred treatment became more centralized by expansion of the sewer networks, which enabled delivery of better secondary and (more recently) tertiary treatment (Burian et al., 2000; Söderholm, 2013).

In line with this narrative, over 88% of the Swedish population are now connected to a public sewer network (Statistics Sweden, 2021). Of the 1700 Swedish wastewater treatment plants (WWTPs), 25% serve a small population (less than 2000 person equivalents (PE)), while about 12% are slightly larger (serving <10 000 PE (SWWA, 2016). In addition, small WWTPs for less than 200 PE serve almost a million households in Sweden. About 40% of these small facilities were estimated to fail organics and phosphorus removal requirements, with issues related to infiltration/inflow into sewers, chemical dosage and mechanical faults (Palmér Rivera, 2006). In line with previous trends (Burian et al., 2000; Söderholm et al., 2022), many people would expect further centralization of wastewater treatment in the future and continuation of the decline in the proportion of small (<200 PE) facilities as WWTPs built during high-income countries’ infrastructural expansion of the 1970s start to require renovation and upgrade.

The conventional centralized wastewater management paradigm is based on a linear model, in which wastewater is collected, treated and disposed of. WWTPs have distinct economies of scale, as unit costs decrease with increases in treatment capacity, so large plants are widely prioritized over small ones (Maurer, 2013). However, sewer systems also have diseconomies of scale related to the size of the catchment area, as systems in larger catchments require longer pipes (Maurer, 2013). This is significant as the collection of wastewater accounts for about 60% of the total budget of a centralized sewage system, according to Eggimann et al. (2016) (Eggimann et al., 2016) and this component can be reduced by decentralized systems.

Studies in other high-income countries, e.g., Switzerland, have found that the degree of centralization is often higher than optimal, as the costs of associated sewers and pumping exceed the economies of scale of the WWTPs (Eggimann et al., 2015). The optimal degree of centralization for wastewater infrastructures is rarely considered in planning, although this clearly indicates a need to rethink the current sanitation approach. Moreover, there are increasing needs to improve nutrient recirculation, energy reuse, resource efficiency, climate change mitigation, and resilience in times of crisis. These challenges are adding to the rationale for more distributed wastewater systems (Eisa et al., 2022; Hoffmann et al., 2020; Larsen et al., 2021) and changing the current centralization paradigm, which has come under increasing criticism (Beck, 2013; Hoffmann et al., 2020; Öberg et al., 2020). The decision to centralize, upgrade or build a new plant offers an opportunity to implement solutions that follow a new paradigm for water management, adapting the system to local opportunities and needs, and closing loops of resources (water, energy, nutrients) via a fit-to-purpose approach.

Decentralized sanitation systems, which include on-site solutions, cluster types and community types, can minimize transport distances and volumes and, if nutrients are recycled from wastewater to
farmland, decrease energy requirements for production of commercial fertilizers (Hoffmann et al., 2020; Libralato et al., 2012; Rittmann, 2013). Such systems can reduce stormwater and groundwater inflows to pipe systems, due to the shorter pipe lengths, and source-separating options such as urine diversion or blackwater/greywater separation can be more easily implemented in them (Tchobanoglous and Leverenz, 2013). Off-grid or decentralized systems also have higher flexibility and thus higher responsivity to changes in conditions, e.g., shifts in water consumption patterns and technological developments (Eggimann et al., 2016). The flexibility and potentially lower per unit costs are key strengths of decentralized systems (Maurer, 2013; Spiller et al., 2015), which are already competitive in specific niches, like urine diversion. However, they also have inherent limitations, including the need for flow equalization, higher energy/resources inputs per unit of treated effluent and higher physical footprints (Tchobanoglous and Leverenz, 2013).

The implementation of decentralized wastewater technologies is often hindered by barriers that are not only technological but also related to the engineering culture and heuristics in the sector, where not all wastewater treatment solutions are considered equally (Etnier et al., 2007). Additionally, institutional setups, weak organizational models (Larsen et al., 2016) and users’ behavior and acceptance (Brands, 2014) rather than technological limitations, have been identified as main sources of failure in the implementation and performance of decentralized systems. There had been a shift in the sanitation discourse in the last three decades, as source-separating systems like urine diversion were increasingly recognized as “alternative” sanitation components (mostly implemented in eco-villages) and feasible options to meet the stricter environmental legislation (Kvarnström et al., 2006). However, sanitation systems with high nutrient recovery capacities, such as urine or blackwater separation, have not yet been widely adopted and are still marginalized (McConville et al., 2017; Söderholm et al., 2022). To identify reasons for this, we recognize a need to explore the paradigms involved. This term refers to the ideas that generate decision-guiding heuristics, i.e., standardized rules of thumb and practices (Nalau et al., 2021). The validity and relevance of heuristics are rarely tested or questioned, and although they may not be based on empirical evidence, they are often used in planning and implementation processes, sometimes generating inappropriate solutions for highly complex problems (Nalau et al., 2021). We hypothesize that the decentralization paradigm has had some sway and is influencing discussions about wastewater treatment, but the centralization heuristic still dominates. This study investigates whether this hypothesis is correct, and if so why the centralization paradigm still prevails. The study was prompted by recognized needs to question whether the centralization of wastewater treatment of peri-urban and rural areas should continue, given technological developments and changing environmental expectations (Beck, 2013; Brands, 2014; Hoffmann et al., 2020; Larsen et al., 2016; Öberg et al., 2020).

We understand that the continuing trend in Sweden, and likely other countries, is to centralize the water systems of rural and peri-urban areas by building long pipe networks and upgrading existing main treatment plants for mixed wastewater (end-of-pipe technology). Here we ask why that trend prevails although innovative technologies for wastewater management with higher capacities for nutrient recovery and circularity have been developed. To address this issue, we address three questions. First, “Is it true that centralization is continuing?” Second, “Why are centralizing practices chosen and decentralization options rejected?” Third, “which technologies for wastewater treatment are commonly chosen in centralized or decentralized systems?” Major aims of the study were to elucidate how Swedish municipalities plan their wastewater systems and take decisions when the systems in certain areas need an upgrade or new ones must be developed. To address the questions and meet these aims we analyzed publicly available data and conducted personal interviews with representatives from local municipalities and water utilities. The methodology applied and results obtained are presented and discussed in following sections, after a brief description of the Swedish water sector.
Municipalities are wholly responsible for the planning, construction and operation of water and wastewater facilities within their municipal water jurisdictions in Sweden. In most municipalities (61%) a municipal authority is solely responsible for water services within their boundaries (61%). However, in other arrangements, authorities formed through multi-municipality associations provide water services for 22% of municipalities, multiservice utilities serve 14%, and a municipal water company serves the remaining 3% (SWWA, 2016). Fees for the services are the main sources of revenue for the water management organizations, accounting for about two-thirds of their total revenues, and state grants make a smaller contribution (SWWA, 2021). As access to safe water and sanitation is considered a basic requirement for an acceptable standard of living, the water services are not subjected to privatization or commercialization (i.e., profits cannot be made from provision of water services) and constitute a “natural monopoly” according to existing Swedish legislation (Söderholm et al., 2022).

The effluent from municipal sanitation systems is subject to licensing rules as expressed in the Swedish Environmental Code that succeeded the Environmental Act in 1999 (Swedish government, 1999). The Code is a framework that contains most of the environmentally relevant legislation. The specific legislation concerning wastewater collection, treatment and disposal is included in the Public Water Services Act (Swedish government, 2006). Paragraph 6 of the Act provides the municipalities the opportunity to expand the public water services, but the Act also implies that under certain conditions municipalities must provide water services in certain areas, regardless of costs. The legislation does not, however, specify which technology should be implemented (it is technology-neutral).

2. Material and methods

2.1. Statistical data collection and analysis

Data on population growth and numbers of treatment plants, classified in terms of size (between 2 000 and 10 000 PE) and municipality in Sweden, were obtained from publicly available databases covering 18 years (2000-2018) compiled by Statistics Sweden (SCB). Information on smaller plants (< 2 000 PE) in the Swedish Water and Wastewater Association’s VASS database, voluntarily provided by municipalities, was explored but not included in the analysis due to low quality and reliability.

2.2. Data collection and analysis

2.2.1. Selection and characterization of the selected case studies

Seven cases were selected for detailed analysis of factors affecting decisions taken regarding existing or soon-to-be-built wastewater systems, each in a different municipality. For this, we followed a purposive sampling strategy, selecting cases to illustrate varying approaches to wastewater management. These included ”typical cases” of common systems, e.g., connecting an area with a sewer network to a WWTP, and ”deviant cases”, e.g., innovative approaches to wastewater management including source separation (Flick, 2014). The intention was to cover a wide range of approaches for solving common problems affecting local wastewater systems, with varying degrees of decentralized and centralized options, system sizes and geographical locations. The authors’ pre-knowledge of the cases, their relevance and representativeness, and the availability of the interviewees to participate, were applied as criteria of “convenience” (Flick, 2014) in selection of the cases.
Table 1. Overview of the case studies included in the investigation. LPS stands for low-pressure sewer.

<table>
<thead>
<tr>
<th>Case study</th>
<th>Type of new system</th>
<th>Previous wastewater system</th>
<th>Reasons to change the existing system</th>
<th>Current technical solution</th>
<th>Size</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>Decentralized</td>
<td>Conventional local on-site</td>
<td>Area development following ruling</td>
<td>Local sewers and on-site</td>
<td>1 200 PE</td>
</tr>
<tr>
<td></td>
<td></td>
<td>wastewater systems, for</td>
<td>that better wastewater systems</td>
<td>facility with chemical</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>individual houses and</td>
<td>were needed in the area and the</td>
<td>precipitation and biological</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>clusters incl. holding</td>
<td>municipality had to take over the</td>
<td>treatment in two sand</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>tanks, septic tanks,</td>
<td>system</td>
<td>filters</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>sand filters and drain</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>B</td>
<td>Semi-centralized</td>
<td>Unexploited land publicly</td>
<td>City development project in an area</td>
<td>Separation of blackwater</td>
<td>12 000 PE</td>
</tr>
<tr>
<td></td>
<td>source separation</td>
<td>owned</td>
<td>with limited raw water sources for</td>
<td>(treated centrally) and</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>drinking water</td>
<td>greywater (local treatment</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>and reuse)</td>
<td></td>
</tr>
<tr>
<td>C</td>
<td>Semi-centralized</td>
<td>Four septic tanks with no</td>
<td>Field inventories showed that the</td>
<td>New sewers and connection</td>
<td>About 23 households</td>
</tr>
<tr>
<td></td>
<td></td>
<td>further treatment</td>
<td>systems were obsolete and needed a</td>
<td>to an existing small</td>
<td>(Ca. 100 PE)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>upgrade</td>
<td>municipal WWTP</td>
<td></td>
</tr>
<tr>
<td>D</td>
<td>Semi-centralized</td>
<td>Small on-site systems,</td>
<td>The property owners requested</td>
<td>New sewers including</td>
<td>1500 PE</td>
</tr>
<tr>
<td></td>
<td></td>
<td>including drain fields,</td>
<td>inclusion in the municipal</td>
<td>gravity and LPS sewers,</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>sand filters, package</td>
<td>jurisdiction area and provision with</td>
<td>pumping stations and</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>plants and holding tanks</td>
<td>drinking water and wastewater</td>
<td>connection to a new small</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>services</td>
<td>treatment plant built in</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>parallel to an existing</td>
<td></td>
</tr>
<tr>
<td>E</td>
<td>Centralized</td>
<td>Four small local plants</td>
<td>Ageing infrastructure, end of</td>
<td>New sewers and connection</td>
<td>Four plants,</td>
</tr>
<tr>
<td></td>
<td></td>
<td>providing various treatments,</td>
<td>technical lifespan</td>
<td>to the main municipal WWTP</td>
<td>70-1500 PE</td>
</tr>
<tr>
<td></td>
<td></td>
<td>including trickling</td>
<td></td>
<td>in an urban area</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>biofiltration, activated</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>sludge processes, and</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>chemical precipitation</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>F</td>
<td>Centralized</td>
<td>Local plant with</td>
<td>Ageing infrastructure, end of</td>
<td>New sewers and connection</td>
<td>2 500 PE</td>
</tr>
<tr>
<td></td>
<td></td>
<td>conventional treatment</td>
<td>technical lifespan, sensitive local</td>
<td>to the main municipal WWTP</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>(mechanical and chemical</td>
<td>receiving waters</td>
<td>in the urban area (10 km)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>precipitation)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>G</td>
<td>Centralized</td>
<td>Individual on-site</td>
<td>Local receiving waters were</td>
<td>New sewer network including</td>
<td>About 400 households</td>
</tr>
<tr>
<td></td>
<td></td>
<td>wastewater treatment systems</td>
<td>affected by eutrophication; the</td>
<td>LPS systems at each</td>
<td>(Ca.1600-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>of various types</td>
<td>municipal water utility took over</td>
<td>property and connection</td>
<td>2000 PE)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>responsibility to provide water and</td>
<td>to the main municipal WWTP</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>wastewater services</td>
<td>in the urban area</td>
<td></td>
</tr>
</tbody>
</table>
2.2.2. Semi-structured interviews and questionnaire

Representatives of the selected municipalities and their water utilities were contacted and invited to participate in the study. The interviewees included water/civil engineers in positions of responsibility, such as heads of water divisions, managers, strategists and project developers who had been directly or indirectly involved in the focal decision-making processes of municipal administrations and water utilities. Upon agreement, they were sent a short description of the study and information about the planned management of acquired data by email. They were subsequently interviewed online during May 2021 and April 2022. The interviews lasted 45–60 minutes and were recorded and thereafter translated into English and summarized. A thematically semi-structured expert interview guide was used. This included questions about specific local conditions, the process of choosing the focal technical system, challenges encountered, sustainability aspects, common practices in the sector, awareness and perceptions of new technologies for local reuse of resources and barriers for implementation. The guide was followed in every interview, with minor adaptations.

After every interview, the interviewees received a questionnaire listing a number of potential drivers for choices of systems. They were asked to assign a score to each of the drivers, indicating their importance for choosing the system selected by the respective municipality or utility, ranging from 0 (irrelevant) to 3 (main reason). Free text space was also provided for them to add information, if desired, about aspects not covered in the questionnaire and/or explanatory comments. The interview guide and the questionnaire are included in the Supplementary Material.

2.2.3. Data extraction, analysis and interpretation

The empirical material was analyzed following an inductive-deductive approach, in which observations are used to establish generalizations about investigated phenomena, and a hypothesis is tested by assessing its validity in diverse case studies (Flick, 2014; Hyde, 2000). The tested hypothesis was that in a high-income country like Sweden there is a trend to centralize the (waste)water systems by building long pipe networks connecting smaller areas to central infrastructure. Thematic analysis was applied, by processing data obtained from the interviews in the form of blocks of information and grouping them in categories or ‘themes’ based on keywords and the contents of interviewees’ responses (Flick, 2014). The themes were then used to illustrate patterns found in the interviewees’ responses. The themes covered drivers, common practices, technical decision-making, sustainability conceptualization, perceptions of and barriers hindering implementation of sanitation systems with enhanced resource recovery capacities, and the current wastewater paradigm. The results were validated in efforts to ensure credibility, plausibility and reliability through refinement of the themes and interpretations of the interviewees’ responses via iterative comparisons with the raw data. Quotations in the Results and Discussion sections are used to provide evidence and illustrate respondents’ common or differing perceptions with regard to a theme.

3. Results

3.1. Trends in the wastewater sector during the period 2000 – 2018

The number of small treatment plants (2 000 to 10 000 PE) in Sweden has decreased in recent decades, while the number of bigger treatment plants (> 100 000) has increased slightly but steadily, from 18 in 2010 to 23 in 2018, according to the national statistics (Figure 1). Hence, the available official data supports the hypothesis that there has been a general trend to decommission smaller plants. Temporal data about WWTPs smaller than 2 000 PE were difficult to obtain, and of low to medium quality when
available. However, the limited available data showed a similar trend in many regions during the period covered (2000–2018). Data about private single on-site sanitation systems are generally scarce, and not included in the national databases, although areas with individual on-site systems are often centralized when they are incorporated into municipal wastewater jurisdictions.

The reasons for decommissioning small plants can vary. Depopulation may be occurring in some rural areas of Sweden (Statistics Sweden, 2020) although the country’s population is generally increasing (Figure 1). This is clearly likely to reduce numbers of small local WWTPs and promote their decommission. However, the most plausible reason for the loss of large numbers of small WWTPs is that they are in areas where water services are being centralized and connected to a municipal network that conveys wastewater to a main WWTP. In addition, the 50–60 years investment cycles for many small plants built during the major infrastructural development period of the 1960s and 70s is ending (Libralato et al., 2012; Söderholm et al., 2022). Thus, many municipalities must decide whether to renew or upgrade their ageing existing infrastructure, or decommission it and connect households or areas it served to the main network.

![Figure 1](image.png)

**Figure 1** Variations in numbers of WWTPs of indicated size classes (in person-equivalents, PE) in Sweden with year (2000-2018) and population density, as indicated by the left and right y axes, respectively. The statistics vary from year to year, which may indicate limitations in the consistency of the data.
3.2. Drivers for choosing wastewater systems

Important parts of the conducted interviews covered issues related to the choice of solution and approach (centralized or decentralized) in each specific case study. A summary of the main drivers influencing the decision to opt for a specific system is represented in Figure 2. All the interviewees responded that the decision taken was the result of an evaluation during an optioneering phase where other options were considered, to a greater or lesser extent depending on the example.

Figure 2. Main drivers and other important drivers (scored 3 and 2 in the questionnaire, respectively) for implementation of the systems considered in Cases A–G. Factors scored as main drivers in at least three of the cases, and secondary drivers in most of the remaining cases, are marked in bold. The asterisked driver standard known procedure refers to perception of the solution as being well known, safe and with low risk of failure based on previous experience in the sector.
Despite the differences in implemented systems, four interviewees mentioned that systems’ robustness was an important driver, meaning that a local sand filter, a local small treatment plant, connection to a main municipal WWTP through a new pipe network, and a source separation system with double piping were all regarded as robust options. Similarly, “simple operation and maintenance” and perceptions of the systems as “standard known solutions” that were safe and with low risk of failure were considered main or important drivers in almost all the case studies. The exceptions were case E (where it was argued that all solutions were welcome, not only the standard ones) and B. The latter was the most innovative case, involving implementation of several alternative technologies (such as vacuum toilets and double piping) and likely the one with the highest uncertainties.

Existing capacities of the municipal network and WWTP were often mentioned as main or important drivers in the centralization examples, supporting the common practice of centralizing certain areas whenever possible. In contrast, lack of capacity in the main WWTP or municipal sewers contributed to implementation of local solutions, such as those of cases C, D and (to less extent) B. Lack of capacity (and time to build the connecting sewer network) provided the possibility to consider other alternatives in the optioneering phase. The centralization option would have been considered in cases C and D if the municipal systems (including the treatment plants and network) had sufficient capacity. The other example with a decentralized solution, case study A, was also designed with the intention to connect the area to the nearest municipal plant in the future, if needed.

Local risks associated with the sensitive ecological and/or chemical status of the receiving waters, including surface and groundwater, were also regarded as important or main drivers in several cases, in which both centralization and decentralized approaches were selected (Figure 2). Individual on-site wastewater systems are considered to contribute nutrient and pathogen loads to the environment because of their varying treatment performance due to problems with their planning, construction and operation (Vidal et al., 2019). Conveyance of pollution from such source points to a local or even distant plant operated and controlled by the municipality is regarded as a better option for the environment than individual on-site systems built and operated by the property owners. This also applies to bathing and drinking water quality, and hence people’s health.

Costs were also often scored as important drivers (and main driver in one case). Investment costs of the centralization options, including building low sewer pipes and pumping stations, were not always lower than those of upgrading or building a local decentralized option, although the estimates depended on the time horizon considered. In that regard, the long-term benefits of centralizing wastewater treatment in certain areas, including future development prospects, were considered to outweigh the higher investment costs. Generally, the perception was that the operational costs would be lower when operating one main plant instead of many small ones.

The decision to implement a source separation system in an area without the benefits of lower investment or operational costs clearly shows the importance of other drivers. In case B, the main drivers indicated for implementing a source separation system were mostly related to mitigation of water shortage, political visions and willingness, and long-term sustainability goals. Similarly, in an investigation of the implementation of source separation systems in pilot areas, Skambraks et al. (2017) concluded that their possible long-term benefits outweighed their potentially higher operating costs.

The presence of a key leader or driving force was mentioned in two cases of (semi)decentralized options. In case A, an individual leader played a key role by raising the alternative of building a local sand filter, using his knowledge and expertise regarding such systems:

“The solution (a local sand filter) was considered an alternative thanks to the investigator, a technical consultant with specific competence, who had knowledge of sand filters and saw the possibility to evaluate an alternative solution to the conventional municipal solutions (a new traditional WWTP or connecting sewage pipe). If the consultant had not
In case B the regional government acted as a main driving force by requesting establishment of a system focused on sustainability and resource efficiency. Consequently, the issue was highlighted by the media and aroused general interest in the public discourse. The water officials investigated the possibilities and took decisions based on the political vision for the region, which included having a sustainable/green profile and contributing to the UN’s Sustainable Development Goals.

3.3. Heuristics in the wastewater sector

All of the studied examples involved a certain degree of centralization of smaller systems, except case A, where a wholly local solution was implemented. Three cases provided examples of full centralization of small plants (E, F) or services in areas with individual on-site wastewater systems (G). Two cases (C and D) exemplified centralization by connection to a relatively small plant in the vicinity of the study area (semi-centralization, as defined in this study) rather than to a main urban WWTP. This was mainly because the main WWTP was working at full capacity and/or the main sewer was too far away to connect, at least at a reasonable cost. In a contrasting example of semi-centralization, in case B a local system for greywater treatment and reuse was planned to be implemented together with a centralized solution for the blackwater fraction (conveyance to the main urban WWTP).

Responses of our interviewees indicate that a general perception in the organizations involved in the studied cases was that services in as many areas as possible should be connected to the main municipal system if there was no clear reason to avoid this option. Main reasons for such centralization included enhancement of the systems’ performance in terms of costs, maintenance, staffing requirements, and environmental impact. It was also argued that large treatment plants can provide better treatment control through (for example) more stable biological treatment and chemical dosing, as they have more homogenous and regular wastewater flows than smaller plants. This is because variations in seasonal flow and wastewater characteristics are stronger in small plants. Another important aspect raised by three interviewees was that it may be difficult for smaller plants to meet increasingly strict legal quality requirements, such as removal of pharmaceuticals that require advanced treatment. In that regard, it was considered more economically efficient to invest in improving main treatment plants by adding further treatment or polishing steps, rather than trying to improve (a large number of) small plants.

As for any other urban technological system, the long-term perspective is important when building wastewater systems. The future development and growth of constructed areas was considered an important aspect, and the general perception was that municipal water and wastewater systems increase possibilities for areas, making them more attractive and suitable for permanent housing. Local property owners often welcomed the centralization initiatives (according to the interviewees), as they would allow them to benefit from the municipal services, despite living in rural areas. However, the interviewees also acknowledged the high costs of building sewage networks and pumping stations, which sometimes acted as “opposing forces to centralization”, as expressed by one participant. The connection tariffs are very expensive, which sometimes reduces property owners’ willingness to be connected to the main system, especially if they have already invested in a functioning treatment system, or the property affected is a sporadically used vacation house.

The main goal of municipal wastewater operators, as expressed by most interviewees, is to do the best they can with the available financial resources, which implies taking decisions that are fair for the society (as the systems are financed by water tariffs), the environment and resources.

Some interviewees recognized that centralization is a common, unofficial practice and rather straightforward decision, often dictated by municipal strategy, but others indicated that their
organizations did not follow an explicit strategy and generally based decisions on case-by-case evaluations.

“It is an unofficial practice, we would say. Whenever possible, we would connect certain areas to the main wastewater treatment plant instead of building something local. That would be the cheapest and easiest to operate.”

“I wouldn’t say that we have an explicit strategy that all the areas should be centralized. We do test different local solutions, like local wetlands for wastewater treatment, and we’re aware of how expensive it is to build long pipe networks.”

Nevertheless, most of the interviewees believed that centralization was the most common approach in Sweden, whenever possible and as long as the connection rate was high enough:

“It is difficult to say if there’s a trend in Sweden, but for example in Stockholm (the area operated by Stockholm’s water and waste utility SVOA) there are only two main WWTPs, so obviously the direction is towards centralization. If that’s right or wrong, I don’t know, but the main thoughts are to keep the costs down and have better control over the discharges.”

Interviewees also indicated that some municipalities were exploring other options, such as the municipality or water utility taking over responsibility for the operation and maintenance of private owners’ small local wastewater systems. However, there are some economic and organizational challenges associated with this approach, as the funding needs to be clarified and some form of subsidy may be needed. Similarly, in new development areas, where establishment and maintenance of a sustainable and innovative profile are important drivers, the municipalities and water utilities were apparently more likely to consider decentralized options with alternative or innovative technology for wastewater treatment, as mentioned by two participants. Future residents of such housing projects may be more conscious of climatic impacts and more willing to pay higher costs for water infrastructure if it reduces their footprint, the interviewees reasoned.

3.4. The decision to upgrade process

The need to change an existing wastewater system or plan a new one can provide an opportunity to reassess the planning strategies and consider something ‘different’ from what had been previously implemented, if assessed as more advantageous.

Common reasons for changing an existing system, based on the interviews, can be summarized as:

- A need to upgrade or renovate existent local (individual or community) treatment plants because they are inadequate or reaching the end of an investment cycle,
- Requirements for new or more adequate systems for new development areas,
- Local receiving waters being affected by eutrophication, and
- Incorporation of new areas into municipal water jurisdictions, which must be provided with water services according to existing legislation and reasons listed above.

Once a need arose, all the interviewees confirmed that an official investigation or assessment, to a greater or lesser extent, was generally carried out to select a suitable solution (optioneering phase). The assessments were mainly based on analysis of costs, including the investment, operational and maintenance costs. One interviewee explained in detail how the cost of connecting an area was compared to the cost of upgrading a local system (if present), in a so-called ‘centralization analysis’, intended to show when centralization became cheaper than upgrading. However, the comparison of costs seemed to vary and the cheapest option was not always the one implemented. For example, investigations by several municipalities had shown that connecting to the main network was the most expensive option,
but still concluded it was worth it in a long-term perspective and considering future developments in the area. Practical issues related to the operation and maintenance of the systems, distances to the main municipal network and local conditions, e.g., groundwater level, elevation, and geographical distances, were often included in the assessment, according to several interviewees. Asset management aspects related to the simultaneous installation of drinking water pipes, for example, were also mentioned as aspects considered during the investigations. One interviewee shared an example of an assessment, involving a simple system analysis in which three main options were compared and discussed, including connection to a municipal WWTP and construction of a local treatment facility. As well as economic, operational and maintenance indicators, other parameters were also considered, such as phosphorus and microbiological discharges, stability of the treatment processes, risks of odors and noise, transport of sludge, and flexibility for handling varying loads and flows.

When an area was classified as a ‘Paragraph 6’ (incorporated into a municipal water jurisdiction, with accompanying municipal obligation to provide water services), the decision sometimes appeared to be obvious for some municipalities. Then there was little discussion about different alternatives, especially if there were problems with the local receiving water, as illustrated by the following quotation from an interview:

“In that case (when the area becomes a Paragraph 6 area) we connect to the main network. So, there’s not much discussion about the technical systems that will be implemented. For the municipality it’s mainly if they’ll build gravity sewers or use pumps. That’s the study they do in the design phase.”

In their investigations, the municipal wastewater operators rarely appeared to include options that were less conventional or innovative than connection to the main municipal network. This is despite all the participants declaring that their municipalities and water companies were open to collaboration and trying out innovative solutions. However, there were two exceptions. In case A, the opportunity was taken to install a local sand filter, a conventional system at small scale but not commonly considered when building larger local systems. This was selected as the most suitable solution following assessment, because of its higher robustness and flexibility, adequate treatment level and lower cost. Case B provides another example of an alternative (and innovative) solution to the conventional planning of wastewater systems, where separation of the greywater and blackwater at source was the chosen option in an assessment with a clear vision: to implement a wastewater system that would help to decrease overall water consumption.

Aspects not considered or included during the technical decision-making were also mentioned during the interviews. These included CO₂ emissions and carbon footprints, possibilities to recycle nutrients and risks associated with the vulnerability of the (centralized) systems in difficult times, e.g., if a crisis or war caused a main sewer to break causing disturbances and local environmental impacts, or the interruptions of distributions of resources, e.g., chemicals.

3.5 Sustainable wastewater systems: Definitions and perceptions

When asked what a sustainable wastewater system would be like, the interviewees gave various answers, ranging from general to highly specific features, including wastewater treatment efficiency and pipe network parameters. In total, the answers covered environmental, social and economic aspects, considered the three main pillars of sustainability, but no single participant included all three aspects in their answer.

Three interviewees mentioned decreasing flows of wastewater to be treated, by decreasing water usage and infiltration and inflows into pipe networks as a good starting point when planning sustainable systems:
“A system without infiltration and inflow, that requires as little operational efforts as possible, would be a sustainable system.”

Gravity sewers were generally considered more sustainable options than pressurized sewers, due to the lower operational requirements (and higher operator-experience). Aspects related to energy efficiency, including more effective pumps and pumping stations, and reducing electricity use, were often mentioned. Similarly, systems designed to enable reuse of resources such as water, e.g., using treated effluent for irrigation or aquifer replenishment, recycling nutrients for farmland application, and recovering energy in the form of heat or biogas, were considered good examples of sustainable wastewater systems. Rainwater harvesting for flushing toilets (instead of using drinking water) was also raised as an idea for effective use of water resources by one interviewee. Another definition of a sustainable wastewater system was:

‘A system where we reuse all the resources as much as possible and as little as possible can be classified as a waste product.’

Lowering the footprint in terms of climate emissions or issues around climate adaptation were not specifically raised by any interviewee, although indirect examples about emissions from transport of sludge or use of chemicals were mentioned. During other parts of the interviews, two mentioned that climate impact assessments were sometimes included in the investigations. In contrast, the Swedish Water and Wastewater Association has developed a vision to make the wastewater sector climate neutral by 2030, with strong emphasis on generating biogas from sludge anaerobic digestion plants to compensate for the emissions from plants’ operations (SWWA, n.d.). At the time of writing, a new climate emissions assessment tool is being piloted with 18 different water organizations and utilities (SWWA, n.d.).

In addition, more technical considerations were used to define sustainable wastewater systems, such as robustness, low maintenance and operational requirements, simplicity and reliability. Essentially, from this perspective, a sustainable system is one that functions as designed.

From a social perspective, comfort and convenience were mentioned by two interviewees who highlighted the superiority of water toilets in comparison to dry or incinerating toilets, which are available options for holiday houses. Public health protection was specifically mentioned by one interviewee in response to the sustainability question, and raised by other interviewees during different parts of the interviews, as it concerns one of the main objectives of municipal wastewater operators: to protect the public health and the environment.

Clearer legislation and guidance by the relevant authorities have suggested importance for implementing sustainable wastewater systems, as municipalities’ small budgets and resources constrain consideration of the sustainability of systems they are building. One interviewee highlighted the potential value of wastewater advisors, i.e., independent public agents who can support property owners with information and expertise when selecting appropriate on-site technology for their specific contexts. Clearly, the resulting enhancement of owners’ knowledge could improve their decision-making and hence the wastewater sector’s sustainability.

Economic sustainability was only mentioned specifically by one interviewee, but all the interviewees discussed (at some points in the interviews) the economic constraints when planning and managing wastewater systems specially alternative systems which may have higher costs.

3.6. Sustainable and resource efficient wastewater systems: Barriers for implementation

The interviewees were asked during different moments of the interviews about their perceptions of resource-efficient systems and systems focusing on nutrient recovery. Not all the interviewees had a
clear idea of what resource-efficient systems implied, although most mentioned biogas recovery or sludge reuse. The discussions about barriers that may be preventing the development and implementation of alternative systems based on resource-efficiency were rich and covered various perspectives, as illustrated in Figure 3.

A barrier mentioned by four interviewees was the existing legislation. From a legislative perspective, the authorities have no incentive or specific requirements to implement systems that are resource-efficient. The main tasks and responsibilities imposed by the current legislation regarding wastewater management (Swedish government, 2006) are to treat wastewater and dispose of the effluents in a safe manner from public health and environmental perspectives, at the lowest necessary cost. Some interviewees suggested that the legislation requires updating to include additional (specific) responsibilities or duties that promote circularity and implementation of systems that enhance recovery and reuse of resources. Likewise, having clearer goals and visions with specified guidelines and requirements would help the municipalities to know what they can do, and are expected to do. Similarly, the responsibility for implementation was raised by several interviewees, who questioned which department/utility or other actors should initiate projects that prioritize nutrient recovery and reuse.

The general perception was that implementing such systems would be costly and municipal wastewater operators could not shoulder the economic liabilities alone. They would need to cooperate with other actors, e.g., waste or environmental departments, in order to financially support (and justify) such projects, and for that clearer legislation would be needed. Accordingly, the Swedish Water and Wastewater Association has recently been working on a proposal for the government to revise and update the current legislation and recommendations to promote circularity and reduce obstacles and misinterpretations of the law that hinder implementation of climate- and environmentally-friendly wastewater technology (SWWA, 2022).

Two interviewees reflected on their own role and responsibilities as water professionals, implying that they could be more open and actively treat new alternative wastewater systems opportunities:

“Perhaps it’s us who are the barriers, we who do not think about it (implementing alternative solutions). It’s difficult when we haven’t done it before and don’t have experience. It’s difficult to think ‘new’, and there are also risks.”

In a similar vein, a third participant mentioned that if the water professionals lack knowledge of alternative solutions they will not be seriously considered in the technical investigations and choices of systems to implement. Personal and organizational attitudes towards new systems were also mentioned as potential barriers, as there is “resistance to change”, as expressed by one interviewed participant. Thus, there are needs for both effective technical solutions and political will to implement them.

Lack of competence and experience of the current water engineers were barriers or challenges suggested by three interviewees. On the positive side, new approaches to wastewater planning and management raise new challenges and requirements for everyone involved to acquire new knowledge, as highlighted by one respondent. The also promote cooperation with project partners and consultants to solve complex problems and develop mutual support.
Figure 3. Barriers hindering the implementation of alternative wastewater systems focused on resource recovery according to the interviewed participants.

Uncertainties related to the technologies’ maturity and readiness to upscale were also concerns:

“*We make investments with a 100-year-old perspective, and that can also be a barrier for implementing new systems because we don’t know if they will function for that long.*”

Some interviewees also questioned the system-thinking perspectives associated with new recovery solutions, as the local reuse of nutrients they enable (for example) may be at the expense of higher local transportation (and linked emissions) of the recyclable products like sludge, blackwater or urine.

Costs were commonly mentioned barriers, as it is difficult to justify the extra initial costs of some new alternative systems, especially for small municipalities. A representative of one municipality mentioned the problem of differences in final costs for similar solutions depending on the location, implying that what works in a big city may not be feasible in rural regions. A commonly expressed idea was that wastewater operators are limited by the income they obtain through tariffs and taxes, and everything they do with that money must be carefully justified. Another mentioned aspect was the lack of economic incentives for innovation, since the municipalities and water utilities do not have profits or margins in the revenues to invest in development, and there is no external competition that can put pressure on them:
"We’re always a monopoly driven by water tariffs, so there is no competition like in the free market. If there was a free market, there would be another type of pressure on innovation."

Time was also considered a constraint as emphasized by two respondents, who transmitted the overwhelming situation that many municipalities and water utilities encounter as they have many ongoing projects in parallel, and decisions often need to be taken fast to find solutions that fulfill the basic legal requirements.

In terms of spatial barriers and geographical distances, new innovative systems may be difficult to retrofit in existing areas or big cities, limiting such systems to new areas or small areas located far from a public sewage network, as discussed by three interviewees. Building new alternative systems in parallel to existing systems, especially at large scale, was regarded as highly challenging. It is not a prerequisite that they must necessarily be decentralized, but it is a consequence of the challenges around their implementation and the existing infrastructure.

Lastly, a representative of one municipality mentioned that there were lower possibilities to recirculate nutrients locally in their region than in most areas of Sweden, due to scarcity of farmland, and hence lower incentives to implement a solution with high local nutrient recovery capacity.

3.7. Perceptions of the current paradigm, is there a need to change it?

Many interviewees perceived the water sector as being traditional and conservative, old-fashioned at times, and locked in “old ways of doing things”, which seemed to hinder the adoption of new approaches and ways of thinking.

A general perception was that a new way of thinking about and understanding the resources was needed in the sector. According to one interviewee, the paradigm had already started to shift towards thinking more sustainably when planning water systems, particularly in terms of their climatic impacts. Most participants regarded a shift from conceiving of wastewater as a waste product to seeing it as a source of water, nutrients and energy as essential. A shift from treatment to reuse was also favorably regarded, although concerns were raised about some issues that still need to be overcome, like removal of pharmaceutical residues in wastewater. The following quotation reflects common thoughts about the need to change the way of thinking and current paradigm in the wastewater sector:

“I think a new way of thinking is necessary. It’s long-term work to “adjust”, you have to start on a smaller scale and increase the context in the long run. Education, information, research and support are needed (for the transition). Good communication is also important, to raise awareness and market new solutions, show good examples. Having opportunities for funding applications to test new technologies can also be important.”

Two interviewees in case studies B and F, stated that their thinking about wastewater planning was different some years ago, and they would have provided different answers then. Centralization is not always clearly the preferred option anymore and there is much more thinking now about sustainability aspects when planning future water systems.

There are considerable conflicts among the municipal wastewater operators’ objectives. For example, there is a need and willingness to move towards recycling and recovery of resources, but also stricter requirements regarding applications of wastewater by-products, such as sludge on farmland, because of their contents of contaminants (e.g., metals, pathogens, pharmaceuticals and other micropollutants). These conflicts hinder possibilities to shift the wastewater paradigm further, as mentioned by one participant who also suggested a need to re-assess what is important, considering the risks.

Several interviewees argued that a significant shift in the wastewater sector and adjustment to new ways of planning will require a long time, because of the infrastructural lock-ins of the urban system and
water professionals’ mindset. It was suggested that a generational shift would be needed to change the mindset of water system planners, and increase the flexibility of the thinking as water engineers start looking at newer opportunities and solutions. One interviewee suggested that drivers for any paradigm change are likely to come from the research community, or other actors outside the public sector. This is because the routine duties of municipal operators leave too little time to think about and develop innovative new solutions.

A key question is how much of the wastewater system should be changed. Three interviewees raised concerns about challenges posed by a complete transition in the sector. They argued that it would be difficult to change the whole system, but new approaches and thinking could be implemented in newly built areas or, similarly, started at a small scale then gradually increased. These views are shared by the Swedish Water and Wastewater Association, which has no power to “force” but can encourage municipalities to implement double piping for separated streams, for example (Finnson, personal communication). Lastly, the need to look at the whole urban water cycle was emphasized by one interviewee who suggested that stormwater should start to be regarded as a useful water resource with reuse potential.

4. Discussion

Planning wastewater systems

This study supports the hypothesis that in high-income countries like Sweden the redevelopment of rural wastewater facilities continues to follow a centralized approach for economic, legislative, environmental and technical reasons. The results also indicate that heuristics and historical developments influence the decisions. Planning with consideration of the optimal degree of centralization for wastewater infrastructures seldom applied in practice likely because of the complexity it entails, resulting in higher centralization rates than the economically optimal (Eggimann et al., 2015).

The interviewees discussed common practices when planning wastewater systems in terms of environmental-technical sustainability, reflecting an engineering culture – more effective treatment control, operation and maintenance, reducing costs and pollution management. This discourse dominated their perceptions, whereas aspects related to circularity of resources and climate emissions were often only weak considerations. Previous studies have discussed how an engineering culture can hinder application of sustainable approaches in urban planning in practice, in contrast to working within organizations with broader views and willingness to embrace options outside a confined discourse (Cettner et al., 2014a). The lack of systems thinking in the wastewater facility planning process has also been suggested as a barrier and consequence of the standard engineering curriculum and/or typical engineering culture (Etnier et al., 2007) in a sector where traditional wastewater professionals with a “Business as Usual” mentality are common (Capodaglio et al., 2017). Systems thinking refers to defining the boundaries of a system to include all the significant activities and relationships (of flows and resources) and understanding those relations. Recognition of narrow system boundaries often limits development of an orientation towards broader systems that include other components of urban cycles (or so-called ‘city metabolism’), such as energy and nutrients, and hence limits inclusion of economic benefits, such as nutrient recovery, in the cost calculations. The planning of different municipal services like water and energy often takes place in silos where data exchange and multidisciplinary cooperation is not promoted (Okwori et al., 2021).

The centralized approach to water and wastewater management have served society well and developed since the 1900s by improving the health of urban citizens. However, with increasing global problems related to rapid urbanization, climate change and resource use, there are increasingly urgent needs to
develop more sustainable approaches to management of water resources (Estévez et al., 2022; Gikas and Tchobanoglous, 2009; Hoffmann et al., 2020). However, those global pressures or drivers do not appear to be strong enough to overcome the existing socio-technical regimes that stabilize the current prevalent systems, as learned from this interview study. The technology has also changed but practice is not keeping up with the technological advances and whether this is appropriate caution or over conservatism needs to be investigated.

What drives planning differently?

Case B provides an example of constrained resources (specifically drinking water) or pressure (drought and water scarcity) pushing municipalities and water utilities to think more broadly and consider other options during their planning processes. Cettner et al. (2014) regarded environmental pressure as one of eight factors associated with receptivity to change in stormwater management, and a strong potential driver of major change in planning procedures.

The planning of wastewater systems often includes assessment of possible options’ effects on water quality, e.g. nutrient discharges, but rarely impacts on watershed water quantity (Etnier et al., 2007), which could include shortages or excesses triggered by combined sewer overflows. Their inclusion would be more likely to result in fair and impartial consideration of alternative options, including (semi)decentralized systems, with (for instance) implementation of local water reclamation and reduction of risks of problems with stormwater overflow, infiltration and inflow due to shorter sewer networks. The growing scarcity of water resources and increased focus on recovery and reuse of resources, together with the development of new metropolitan or isolated communities, putatively affect decentralization processes (Libralato et al., 2012), and all of those factors seem to have influenced the decision in case B. Water scarcity may have been a strong factor in case B, but without adequate macro-level drivers e.g., political initiative and will, and conducive interactions between institutions, planning the alternative system in practice would not have been possible. Hence, other drivers as discussed in section 3.2 enabled planning of the alternative system and differed from those in the other cases (Figure 2). They included clear goals, visions (a strong sustainability profile), and strategies initiated by politicians and driven by water professionals following previous Swedish showcases such as the H+ project in Helsingborg (further described in Lennartsson et al. (2019)). Similarly, Skambraks et al. (2017) found that strong environmental goals at national and local government levels were the main drivers for investment in new and integrated systems. Such goals are needed to overcome the challenges commonly encountered during the implementation of source separation systems, which often arise from the involvement of several municipal sectors. Moreover, a common vision and strong leadership are important elements of two (of three) theoretical components of a sustainability transition framework developed by Lennartsson et al. (2019). The first component or factor consists of shared views, problems, goals and collective visions, and the second is the presence of committed, responsible actors that can act as ‘frontrunners’ promoting sustainable actions. As noted by (Lennartsson et al., 2019), difficulties in establishing a common vision and lack of a strategic unit may significantly hinder implementation of alternative systems. In addition, guiding policies and goals should be listed in steering documents and included in budgets to ensure that they are considered, prioritized, and consequently provide leverage for local administrations to implement alternative systems (Lennartsson et al., 2019).

In a similar illustrative example, a blackwater separation system was implemented in Munga, an area hosting about 279 households in central Sweden, following an investigation that included consideration of this alternative in accordance with a municipal policy document. The local policy adopted by the City Council (Västerås city, 2013) included areas in and outside the municipal water jurisdiction and specified that the wastewater solutions must have long-term sustainability, be robust and contribute substantially to the local recycling of resources from technical, financial and environmental perspectives (Kärrman et al., 2017). Another source separation system implemented in Södertälje municipality (blackwater wet composting with urea hygienization) was also enabled by strong political will to establish a pilot facility in the area following a ‘recirculation’ policy dictated by the municipal environment committee (Miljö nämnd), influenced by problems with severe eutrophication of the local lakes (Kärrman et al., 2017). The policy documents are similar in both cases, one of the differences
being that in Västerås municipality the policy applied to all water jurisdictions and in Södertälje only to individual on-site treatment facilities outside the municipal water jurisdiction.

Case study A illustrated the implementation of a local decentralized option: a traditional treatment option with a sand filter. A distinguishable driver for choosing that option was the role played by a key person. This was the ‘suggestion’ and evaluation of the alternative solution together with conventional options (a new traditional WWTP or connection pipe) by a consultant in the assessment or ‘optioneering’ phase. This is a crucial phase in which traditional and well-established options can be compared to innovative or alternative ones, and new opportunities can arise. Key people leading change is another factor associated with receptivity to change (Cettner et al., 2014b), as commitment, personal knowledge and experience can be highly influential human factors in the decision-making process. Removal of barriers so that all solutions are consider impartially in order to implement the most effective ones will require robust optioneering phases, engineers acting as leaders rather than followers (Etnier et al., 2007), and hence a change in thinking in the sector. Lennartsson et al. (2019) also discuss the importance of a guiding policy document (either vague or very specific), strong commitment, and leadership providing clear direction for proceeding towards implementation. Champions can have an instrumental role as proactive, innovative and inspired individuals that drive change in overcoming internal resistance to it and were regarded as crucial for the implementation of innovative source-separation systems by Lennartsson et al. (2019). They also found that there were no obvious champions in other projects that did not include significant innovation or transformational change. However, the cited authors also questioned the dependence of sustainability transition processes on champions, especially in larger municipalities where implementation processes and procedures can be more stringent. People that understand and believe in the transition are, nevertheless, needed to anchor a vision and drive it forward as mentioned in sections 3.6 and 3.7 in the analysis of drivers and barriers for implementing alternative systems and driving change.

Perceptions of decentralized wastewater treatment systems

Decentralized wastewater systems are often associated with individual on-site facilities that are ineffective or old plants that are coming towards the end of their technical lifespan and need to be upgraded. Bad experiences from the past (and present) of decentralized systems have probably influenced these perceptions, as discussed during the interviews, and decentralized systems are still often “pointed as an example of a bad idea” (Capodaglio et al., 2017) regardless of the number of successfully implemented examples (Gardner and Sharma, 2013; Larsen et al., 2021). Reasons for negative experiences include inappropriate dimensioning, ineffective operation or maintenance, and in some cases being “born technologically obsolete” (Capodaglio et al., 2017). Centralized sanitation systems are portrayed as superior and more efficient in comparison to smaller and decentralized ones, which affects the development and establishment of alternative infrastructure that could support decentralized systems (like source-separation) (Barquet et al., 2020). The wastewater sector is characterized by large-scale infrastructure with long-term investment horizons that result in considerable lock-in effects (Barquet et al., 2020; Söderholm et al., 2022). These lock-ins represent structural barriers for resource reuse such as phosphorus, and can slow down the pace of change (Barquet et al., 2020).

Etnier et al. (2007) identified five main barriers to the creation of an environment in which all wastewater treatment solutions are considered fairly and impartially in engineers’ evaluations and implementations of decentralized wastewater technologies. Some of these barriers are similar to those discussed in section 3.6, such as the engineers’ lack of knowledge of decentralized systems (or even their unfavorable perceptions) and the lack of system thinking applied to wastewater issues. The regulatory system’s lack of support for decentralized systems (in the USA) and the financial reward for using centralized systems were also suggested by Etnier et al. (2007) as major barriers and could be relevant in a Swedish context, as discussed from a historical perspective by Söderholm et al. (2022).

Decentralized wastewater treatment systems include not only individual on-site systems for single households, but also systems with various degrees of semi-centralization, satellite small-grid and hybrid
systems, in contrast to grid-dominated systems (Hoffmann et al., 2020; Libralato et al., 2012; Tchobanoglous and Leverenz, 2013). According to these commentators, when both centralized and decentralized systems are feasible, the most appropriate technology should be selected in a case-based approach, i.e., the option that is most economically affordable, environmentally sustainable and socially acceptable. A further advantage of fair assessment of decentralized technologies is that it gives planners opportunities to consider implementation of a source separation system, e.g., urine, blackwater or greywater separation, and resource-saving systems, e.g., low flush/vacuum toilets to enhance resources and energy recovery. Source-separating technologies can also be centrally implemented, as decentralization is not a prerequisite but rather a consequence of the existing infrastructure and the complexity of retrofitting new systems into the existing one. Lastly, an interesting alternative option raised by some interviewees would be for the municipality to take over the responsibility to operate (without ownership) individual on-site wastewater systems, to ensure better practices and more control, as already implemented in other countries or regions e.g., Flanders (Van Den Broek, personal communication). In such cases the key considerations are not related to the technology or scale of the systems, but their management.

**Barriers to the implementation of wastewater systems focused on resource efficiency and recovery**

This study has not investigated the empirical validity of barriers reported in the interviews. For instance, interviewees often claimed that current legislation does not incentivize strongly enough the implementation of systems with high nutrient recovery capacities (for example). However, we did not attempt to validate or disprove whether this is true, or it depends on how the legislation is interpreted. Rather, we documented perceptions of professionals regarding wastewater decision-making in Sweden and used those perceptions to map the heuristics and create a list of barriers. However, some of the identified barriers, e.g., lack of visions and goals, clear regulations, policy, and financial support, have also been reported by previous authors e.g., Lennartsson et al. (2019) and McConville et al. (2017). They have also been noted by relevant actors in the water sector, such as the Swedish Water and Wastewater Association (SWWA, 2022). McConville et al. (2017) explored mechanisms hindering expansion of source separation systems, through investigations including case studies and interviews with sector experts. They identified barriers including immaturity of technology, weak national advocacy coalitions, risk aversion, lack of institutional capacity and other organizational challenges. Their findings corroborate the barriers suggested by our interviewees. One barrier noted by McConville et al. (2017), related to the “low economic value of recovered nutrients”, was only tangentially discussed by our participants. However, it appears to be a fundamental factor for the success of alternative systems based on resource recovery, as failure to reuse recovered resources is a common cause of failure. Higher initial capital costs are often mentioned as major challenges that are not always based on empirical foundations, as discussed during the interviews. However, legal and institutional uncertainties, as well as lack of capacity, may be even greater obstacles.

The Swedish legislation is neutral in terms of which technology or system should be applied in every municipal case, hence there is no impediment to implement source-separating systems in that regard. The Swedish Environmental Code clearly promotes closing the nutrients loops, as it states that “reuse and recycling, as well as other management of materials, raw materials and energy are encouraged with a view to establishing and maintaining natural cycles”. However, that interpretation of the Code appears to be seldom applied. Moreover, the fact that municipal services are often managed in silos makes it difficult to justify the costs of alternative systems as the benefits mostly fall outside the water sector (e.g., nutrients or energy recovery and use). Neither the rules and legislation that specifically apply to the wastewater sector, nor the organizational and business models are built for creating circular flows of wastewater resources (Kvarnström et al., 2022), causing inertia to change.
According to the Swedish Water and Wastewater Association (Finnson, A. 2022, personal communication), implementing systems focused on resource recovery would require long-term (10–20 years) contracts between wastewater operators and farmers interested in reusing recyclable wastewater fractions (e.g., urine or blackwater) in agriculture. Other required actions include pressurizing fertilizer companies to include a certain percentage of recycled nutrients in their products, similar to the existing legislation regarding ethanol contents of fuels for combustion engines, and the development and dissemination of relevant research results (Finnson, A. 2022, personal communication).

Traditionally, the wastewater sector has dealt with the symptoms of poor treatment e.g., eutrophication as problems, rather than as using opportunities and drivers to improve resource recovery and promote circularity, which calls for a change in the thinking and planning approach. Simply, the problems we are facing in urban water management “stem from mixing and dilution” (Larsen et al., 2015), in addition to the entrenched and locked-in linearity of systems associated with the current paradigm in the sector (Barquet et al., 2020; Fam et al., 2010). This paradigm must be changed, according to many of the interviewees, and has already started to change, according to a few of them.

5. Conclusions

The number of the smallest treatment plants in Sweden for which official information is available (2 000 to 10 000 PE) have decreased in recent decades (from 288 to 251), while the number of larger treatment plants (> 100 000 PE) has increased slightly but steadily (from 18 to 23). The available data supports the hypothesis that there is a general trend to decommission smaller plants. Centralizing the water systems by building long sewer networks and upgrading existing main treatment plants (end-of-pipe technology) was part of the sector’s heuristics according to the interviewees.

The drivers or reasons for implementing specific systems were shared by most cases. Robustness of the systems was an important driver for five interviewees for example. However, very different solutions (e.g., a local sand filter, a local small WWTP, connection to a main municipal WWTP, and a source separation system) were regarded as robust in the different cases. Optioneering, involving comparison of several possible solutions, included only in once case (B) and alternative sanitation system with capacity for resource recovery. Upgrading an existing local plant, building a new one or connecting to a main WWTP were the main options considered. Local risks associated with the sensitive ecological/chemical status of the receiving waters were also main drivers in several cases. The costs were repeatedly mentioned as important, but not decisive, drivers since more expensive options were sometimes implemented. The limited capacity of the sewer network and main WWTP were contributory drivers for implementing local semi-decentralized and/or resource-recovery solutions, as was the presence of a key leader or driving force supporting alternative solutions.

Better treatment control, making the system more effective in terms of operation, cost, use of staff and minimization of environmental impacts were some main arguments for centralizing wastewater management in some areas. The uncertainties related to future legal requirements also contributed to the preference for centralized solutions in some cases, as well as the financial advantages of improving one large facility rather than many small facilities.

Paradigmatic changes in sanitation systems (e.g., introduction of source-separating alternatives or decentralized systems) can be driven by resource constraints (e.g., scarcity of drinking water), environmental pressure (e.g., the need to ameliorate climate change), key leadership and experience, as well as clear goals, common visions and strategies. However, the drivers must be strong enough to break the inertia and create change in a sector characterized by strong lock-ins. Barriers to the implementation of alternative sanitation systems (with high capacities for resource recovery) suggested by the interviewees were of economic, legislative, socio-political, technical and organizational nature. Lack of
economic and legislative incentives, differing interpretation of regulations, lack of initiative, vision and competences, uncertainties associated with immaturity of the technology, and lack of time were some of the specific identified barriers. The engineers’ perceptions and typical heuristics of the water sector could also prevent some alternative, feasible solutions being considered fairly and impartially.

The current paradigm in the wastewater sector is characterized by linear models based on collection-treatment-disposal of mixed and diluted wastewater. The new challenges affecting our socio-technical wastewater systems, in terms of water (re)use, recovery of nutrients and energy, and removal of micropollutants call for a new sanitation paradigm that exploits resources provided by the wastewater (water, nutrients, and energy) rather than dealing with the symptoms of poor treatment (eutrophication and contamination). Despite the advances in alternative decentralized systems, centralization heuristics still appear to be widely applied, although some pilot and full-scale examples show that viable alternatives are emerging.

6. Acknowledgements

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8. Author contributions

Brenda Vidal: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Visualization; Writing - original draft; Writing - review & editing. Annelie Hedström: Conceptualization; Funding acquisition; Methodology; Project administration; Supervision; Writing - review & editing. Liz Sharp: Methodology; Writing - review & editing. Inga Herrmann: Conceptualization; Funding acquisition; Methodology; Project administration; Supervision; Writing - review & editing.

9. References


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societal benfits]. Bromma.(In Swedish)


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https://doi.org/10.1016/J.SCS.2016.09.013
Supplementary material

1.1. Interview guidelines organized in themes and support questions. The original questions were in Swedish

➢ Opening questions
  1. Professional background
  2. Describe your work, role and responsibilities
  3. How familiar/involved are you with the case study/project?

➢ The case-study/project: description of the technical solutions
  4. Which type of wastewater system (including type of wastewater treatment process) was present before? If applicable
  5. Which type of wastewater system (including type of wastewater treatment process) has been implemented and is present now?
  6. Size of the systems (PE), population. Is it a newly built or already existing area?

➢ The choice of the technical system
  7. What was the reason why the old system had to be changed?
  8. What was the selection of the new system based on? What were the main drivers or reasons for your organization to implement the solution we are discussing?
  9. Did you use decision-making aid tools such as multi-criteria analysis, or cost/benefit analysis prior the selection of the system, to compare several options and get a better overview? How were they used? Were external consultants involved?
  10. Who was involved in the discussion and selection of the new system?
  11. Were other solutions considered at some point throughout the process but not selected at last? Which ones, and why were they not selected?
  12. Were there some relevant aspects such as e.g., environmental impact, multi-functional co-benefits, possibility to implement less conventional systems, sustainability aspects, others, etc. that you think were generally not considered when choosing the technical solution?
  13. What were considerable challenges encountered during the selection process? Prioritize them.
  14. What do you think is the general perception in your municipality/water utility about the implementation of centralized or decentralized solutions?
  15. Do you think there is a common practice, or conventional or standard solution in the sector?
  16. Do you know anything about resource-recovery oriented sanitation systems? -Which resources are considered most important or receive most focus on in your municipality? Nutrients (P, N), energy (heat, biogas), other.
  17. Do you know if/how other municipalities or countries have implemented any type of resource-recovery oriented sanitation systems? Could you give examples?
  18. What do you think is needed to implement a different system than the chosen one e.g., a decentralized plant instead of building a connecting pipe, a system with focus on resource recovery, or a system with lower climate emissions than the one chosen?
  19. If you were making the same decision now, would you do it differently?

➢ Sustainability aspect
  20. What would be a sustainable wastewater treatment system for you?
  21. How does your municipality/water utility contribute to the sustainability of the wastewater sector?
22. Do you think that different sustainability aspects (environmental impact, emissions, carbon footprint, use of resources, nutrients recycling, distances, asset management, affordability, etc.) were considered when selecting the new system, or the drivers were skewed/an aspect dominated the discussion and final choice?

23. Is there space or opportunities to innovate and experiment with alternative sanitation systems in your organization or you see/experience resistance at the organizational or planning level when considering/proposing alternative sanitation systems?

24. Do you think there is a need to have a new “thinking” in the wastewater sector?

25. What do you think is needed to bring new approaches or “thinking” into the sector?

26. What are the main barriers you see for implementing systems with focus on e.g. resource recovery, or more resource efficient (lower climate emissions)?

27. For newly built projects: was the choice influenced by the SDGs (UNs sustainable development goals) in some way?

28. How can we help, from the research and education sector, to support the municipalities to think and choose more sustainably/so they can take more sustainable decisions?

End of the interview.
1.1. Survey about drivers. The original questions were in Swedish.

In the table below is the questionnaire guideline with the list of drivers provided to the interviewees by email after the interview. They were asked to rank them on a scale of 0-3 based on how those drivers had been considered when deciding to implement the selected system. The definition of the drivers was slightly modified when needed to fit the case studies.

Where 0 is: didn’t influence or was not considered in discussions.

Where 1 is: It was present or it was considered but did not affect the choice in either way.

Where 2 is: Considered as important and contributed to the choice implemented

Where 3 is: A main driver for the choice to implement

<table>
<thead>
<tr>
<th>The decision to implement the specific solution discussed earlier during the interview was dependent on:</th>
<th>Your grade (0-3)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>System related aspects:</strong></td>
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<tr>
<td>The main WWTP does/does not have capacity</td>
<td></td>
</tr>
<tr>
<td>The nearest main pipe network does/does not have capacity</td>
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<tr>
<td>The distance to an existing municipal system is short/long (please, specify the distance in the comments section)</td>
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<tr>
<td>Drinking water supply is centralized/being installed in the area</td>
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<tr>
<td><strong>Municipal vision:</strong></td>
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<tr>
<td>Municipal development plan with focus on sustainable/innovative/resource oriented ideas that influence the urban water systems (Please use the box below to explain it further)</td>
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<tr>
<td>Local environmental goals/green profile (Please use the box below to explain it further)</td>
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<tr>
<td>Contribution to the UN goals for sustainable development</td>
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<tr>
<td><strong>Economic aspects:</strong></td>
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<tr>
<td>Lower operational cost (if higher, please specify)</td>
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<tr>
<td>Lower investment cost (if higher, please specify)</td>
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<tr>
<td><strong>Environmental aspects:</strong></td>
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<tr>
<td>Environmentally sensitive local receiving waters</td>
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<tr>
<td>Local water scarcity</td>
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<tr>
<td>Nutrients recovery possibilities</td>
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<tr>
<td>Energy recovery possibilities</td>
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<tr>
<td>Health risks – e.g bathing water quality</td>
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<tr>
<td>Health risks – e.g. overflow points</td>
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<tr>
<td><strong>Organizational aspects:</strong></td>
<td></td>
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<tr>
<td>It is a standard known procedure, perceived as &quot;safe&quot; and with low risk</td>
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<tr>
<td>Collaboration with a research institute/university</td>
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<tr>
<td>Presence of a key leader/driving force</td>
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<tr>
<td>Result from an evaluation/analysis/investigation comparing different options (Please use the box below to explain it further)</td>
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<tr>
<td>It is considered “easier”/less complicated to operate</td>
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<tr>
<td>Working environment aspects (Please use the box below to explain it further)</td>
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<tr>
<td><strong>Social aspects:</strong></td>
<td></td>
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<tr>
<td>Sociopolitical/public pressure (e.g. media coverage) (Please use the box below to explain it further)</td>
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</tr>
<tr>
<td><strong>Other (for any category)?</strong></td>
<td></td>
</tr>
</tbody>
</table>
Would you like to further explain or clarify the above selected choices? Please write here:
Söderholm, K., Vidal, B., Hedström, A., Herrmann, I. (2022)

Flexible and resource-recovery sanitation solutions: What hindered their implementation? A 40-year Swedish perspective

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Flexible and Resource-Recovery Sanitation Solutions: What Hindered Their Implementation? A 40-Year Swedish Perspective

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Flexible and Resource-Recovery Sanitation Solutions: What Hindered Their Implementation? A 40-Year Swedish Perspective

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ABSTRACT
Although Sweden pioneered in the development of resource-recovery sanitation solutions, and there has existed a political awareness of such solutions since the 1990s, their implementation has been slow. We adopt a historical (40-year) perspective and use the main journal of the Swedish sanitation sector as source material to go into depth why this has been the case. Central explanations emerge in terms of previously strong governmental control and continuously tightened environmental requirements that ceaselessly have expanded and strengthened the large-scale centralized sanitation system. In parallel, the sector has continuously been reminded of the shortcomings of alternative (and smaller) solutions and of the tension between recovery and treatment/risk management. The study highlights the possibility of achieving long-term and profound impacts from policy mixes, as well as the strong influence of the sum of challenges and choices over a long time, on today’s perspectives and propensity for change.

KEYWORDS
Wastewater treatment; Sweden; history; urine diversion; resource-recovery sanitation

Introduction and Problem Definition
Increasing pressures of ageing infrastructure, demographic changes, heightened environmental awareness and climate change have raised concerns about the “lack of adaptation” of most of society’s large-scale technological systems. The conventional centralized sanitation system is generally considered inflexible due to high capital costs and technological and institutional lock-ins. The inherent inflexibility of the centralized sanitation system is further explained by the difficulties to optimize the recovery of resources from a system originally designed with a different purpose, i.e., to improve urban hygiene and to control water pollution (McConville et al., 2016; Hoffmann et al., 2020). In Sweden, public sanitation is widespread with close to 90 percent of the population connected to public sanitation systems. In urban areas, the connection rate is almost 100 percent, and only four
out of a total of 290 municipalities have an access-rate to public sanitation that is less than 50 percent of the population (Swedish Government, 2018.34). Swedish sanitation is further heavily centralized.

Since the 1990s, alternative sanitation solutions that are more diverse in terms of, e.g., source separation (e.g., urine diversion [UD]), decentralization, and hybridization (e.g., combined local grey water treatment and off-site black water treatment), have been identified as important paths in addressing pressing global challenges like rapid urbanization, eutrophication, and climate change (Hoffmann et al., 2020; Capodaglio et al., 2017; Sitzenfrei and Rauch, 2014; Sharma et al., 2013; Brands, 2014; Sharma et al., 2010; Larsen, 2011). Still, there is a lot left to be desired in terms of the pace at which these solutions are implemented. A lot of research has been produced that tries to understand the background, and methods to remedy the inertia, where most have dealt with alternative sanitation solutions taking place at the micro level and less with the macro-level dynamics that shape how sanitation is managed (for an overview, see Hoffmann et al., 2020). Important macro-level dynamics in this context concern the interactions between institutions, actors, and technology at the national and/or regional level, such as how policy mixes could support the diffusion of alternative solutions. Such interactions and effects are usually processes taking place over several years, and nothing that can easily be studied without adopting an historical perspective. Similarly, a large-scale technological system, such as the centralized sanitation system, is the sum of several, long-standing incremental changes, decisions, and efforts. It is, therefore, difficult to understand in depth—such as concerning its propensity for change—without a longer historical perspective.

In this article, we will use a historical (40-year) perspective to try to go into depth both on the reasons behind the inertia in establishing alternative sanitation solutions in Sweden, and on the long-standing dominance of the centralized sanitation system. Sweden is an interesting case in this context because it was an early pioneer in UD, and the political awareness of alternative sanitation solutions have been significant. Still the adoption rates have been slow, and the centralization of the Swedish sanitation system is continuously expanding. As source material for our study, we have used the main sector journal of the Swedish water and wastewater sector from 1974 to 2015, i.e., VAV-nytt [Water and Wastewater Treatment News] (1974–2001, four issues/year), later changed to Svenskt vatten [Swedish Water] (2002 and onwards, four issues/year). The publisher was/is the sector organization VAV [the Swedish Water and Wastewater Treatment Plant Association] founded in 1962, whose members include all the municipal water utilities to which 90 percent of the Swedish population is connected. Based on this extensive source material, four central themes have emerged which explain the difficulties in establishing alternative sanitation solutions and for the lengthy “lock-in” of the centralized sanitation solution. These include: (a) previous strong national intervention; (b) long-term challenges related to the centralized solution; (c) challenging existence for small plants and on-site systems due to tightened environmental requirements and neglect; and (d) the great environmental mobilization of the late twentieth century.

**Method and Analytical Framework**

The source material has been analyzed using a classical historical method based on the perception of the source as a material for producing knowledge that is not previously
known, and where the critical examination and analysis of the source material are central to determining what kind of knowledge can be gained from it, and what value this knowledge has. VAV-nytt/Svenskt vatten is a distinct sector journal, which we consider a highly valuable source material in this context. Hence, the readers as well as the authors of the journal articles (including presentations/comments/discussions) are to the largest part representatives of the sanitation sector in terms of practitioners, officials, and politicians from the local to national level, mainly from the public sector, in addition to a smaller proportion of researchers and private enterprises in terms of suppliers. In other words, stakeholders are represented from different levels and in terms of regulators, developers, and implementers. This is reflected in the character of the journal which, hence, covers both technical, economic, organizational, political/regulative aspects of the management of sanitation. Taken together with the fact that the journal has had a permanent and regular publication over the studied period (1974–2015, forming a total of 160 un-digitalized journal issues), we consider this a valuable source material.

Through the analysis of the journal issues, our ambition was to seek to identify, with as open mind as possible, themes with explanatory value for the difficulties in establishing alternative sanitation solutions and for the lengthy “lock-in” of the centralized sanitation solution. The analysis was carried out by an historian of technology who prior to the analysis had in-depth knowledge of the early (late nineteenth- and early twentieth-century) implementation of the centralized sanitation system in Sweden; however, not particularly concerning the expansion of the centralized sanitation system versus alternative sanitation solutions over the last 50 years. Hence, the basic analysis was conducted by a researcher with essential socio-technical and historical understanding of the centralized sanitation system but without a pre-understanding of “what should” explain its inertia and dominance in recent decades.

In the choice of the sector journal as main source material, we are aware of a potential built-in bias for the central sanitation solution in many texts since it has been/is dominant. However, we experience that this source material, largely formulated by and directed to the sector (in a wide sense) itself where the inertia of the centralized sanitation solution (and the parallel opt-out of alternative sanitation solutions) took place, offers a broad (if not complete) insight into the most central explanatory themes over the period studied. This narrative has been supplemented with information from some public reports and other literature (and a few interviews) to offer a more complete picture of certain processes and development paths. Especially the most comprehensive theme of the study—long-term challenges related to the centralized solution—has been supplemented with public reports and other literature for this reason.

Our analysis is inspired by the socio-technical transition literature (Geels, 2002) in an attempt to sort, evaluate, and contextualize the reasons for difficulties in establishing alternative sanitation solutions and for the long-term dominance of the centralized sanitation system, i.e., why a socio-technical transition in line with the often-expressed political agenda has not occurred. The multi-level perspective (MLP) framework attempts to deal with the complexity and resistance of large systems to change. It posits analytical and heuristic levels on which processes interact and align to result in socio-technical system transformations; niches (micro-level), regimes (meso-level), and landscape (macro-level) (Geels, 2005).
In niches, innovations of a more radical nature can avoid the often-skewed competition between existing and novel technologies and have the possibility to be developed in interaction with a limited number of users. Such niches protect against established technologies and create possibilities for learning where incremental innovations, e.g., learning in production that lowers costs, can over time make the new technology competitive with respect to the established socio-technical “regimes” (Kemp et al., 1998). Geels (2002) describes a socio-technical regime as consisting of existing production and consumption systems with strong “lock-in” in terms of common values and mutual dependence between important social groups, such as end-users, suppliers, and the political and educational systems. Any changes therefore tend to happen through a gradual configuration and reconfiguration based on what is happening within the regime, in different competing niches and on what is sometimes called the “landscape level” (Geels, 2014). At the landscape level, comprehensive ecological, cultural, geopolitical, and macroeconomic changes may take place which affect all regimes, and these can refer to emergencies, natural disasters, or technological trends (e.g., digitalization). Taken together, it is the interaction between what is happening at the landscape level, e.g., increased awareness of climate change and what causes it, and in various underlying niches, that creates prerequisites for a socio-technical transition. Promising experiments and radical innovations in niches are often dependent on changes at the landscape level to disseminate. Correspondingly, if there are no responses in niches to the changes that occur at the landscape level, no rapid change can take place.

While the MLP framework provides guidance about the gradual changes that need to happen among niches, the regime, and the landscape levels, it provides insufficient guidance about how this interaction actually occurs in practice and what “intervention points” there are for policy instruments (if there is a political will) to stimulate a certain type of development in a certain direction. Still, for public policy to facilitate necessary transitions towards a more sustainable society and achieve challenging climate policy goals, there is great need for a deeper understanding of the dynamics of transition processes, including the preceding time-consuming, complex interactions.

This article is an attempt to contribute to an expanded understanding of these dynamics through a long-term (40-year overview) perspective of a cross section through niches, regimes, and landscape when it comes to the “lock-in” of the centralized sanitation system in Sweden in parallel to politically driven visions for sustainable development and alternative sanitation solutions. Before we go into more detail on the four central themes with explanatory value in these matters, we offer a brief overview of alternative sanitation solutions in Sweden and other western countries. At the end of the article, the explanatory values of the various themes are summarized and discussed through the lens of the socio-technical transition literature.

**Alternative Sanitation Solutions in Sweden and Other Western Countries—An Overview**

Alternative sanitation solutions have in part been developed and implemented in Sweden since the 1990s, in parallel with the upcoming recycling and sustainability discourses. Hence, in the mid-1990s, several high-profile UD pilot projects (such as in eco-villages) were launched in Sweden. In fact, Sweden is the birthplace of the modern UD toilet, and
Swedish experience has served as an important source of inspiration for the increasing global support of UD (Fam and Mitchell, 2013). The main drivers for UD implementation in Sweden, as in many other industrialized countries, are typically related to the mitigation of problems with eutrophication in the receiving waters and the meeting of new legislation and environmental goals. In low-income areas, including rural, peri-urban, and urban settings around the world, main drivers are often associated with the lower costs of UD systems, alongside the possibility of obtaining a high quality, fast-acting fertilizer and improved sanitation conditions (Kvarnström et al., 2006; Lüthi and Panesar, 2013; Simha and Ganesapillai, 2017).

Since the late 1990s, however, the diffusion of UD has declined considerably in Sweden. McConville et al. (2016) explain the overall marginalization of source separation in Sweden today by the lack of knowledge development along with a weak interchange between knowledge development and entrepreneurial activity. Fam and Mitchell (2013) to a greater extent depart from a macro perspective and as a negative force in this context, point at the dominance of top-down initiatives in the organizing of UD systems in Sweden from the late 1990s—due to increasing political awareness and national funding for sustainability initiatives—in contrast to the typically bottom-up social organizing of UD systems in Sweden in the years before.

In the late 1990s, UD was to some degree politically supported by municipalities that were sometimes funded by the so-called Local Investment Program (LIP), a national investment program during the period 1998 to 2002 of SEK 6.2 billion for the transition to sustainable development. Still, only about 10 percent of the LIP funding went to the water- and wastewater field, and then mainly to measures in the centralized sanitation system, such as investments in drinking water, storm water, wastewater treatment plants, and sewage networks in urban environments. All in all, LIP financed UD-WC and -dry toilet systems in 10 Swedish municipalities, alongside so-called filter units and “package plants” in a few municipalities. The lack of community involvement in these projects later revealed challenges for end-users in organizing and managing the new technology, particularly when it came to measures in individual wastewater facilities for a group of individual property owners, which were less successful than measures relating to single-family households (Swedish EPA, 2005). In the evaluation of the LIP program in 2005, it was found that because of the program, in several municipalities there was now good basis for larger-scale investments in alternative sanitation solution facilities. Still, it was of critical importance with central decisions in the short term that promoted the development, such as in the form of financial instruments, supporting regulations, R&D initiatives, and technology development (Swedish EPA, 2005). Such national support, however, never really did become a reality. Overall, the politically driven vision for sustainable development behind the initiatives of LIP, nutrient recycling, UD, etc., of the late 1990s, shifted to issues related to climate change in the 2000s.

In other European countries, such as Germany, Switzerland, and the Netherlands, concerns over the conventional centralized systems’ loss of valuable nutrients, inflexibility, and sometimes resource (water and energy) inefficiency have grown since the 1990s. This has in turn given way to a new paradigm with a focus on re-using the resources found in wastewater and implementing alternative sanitation solutions. The main drivers behind this development also have to do with new requirements on emerging pollutants removal, or dealing with the uncertainties that climate change
introduces in the water systems (Swart and Palsma, 2013; Larsen et al., 2009; Londong, 2013). In these countries, alternative sanitation solutions did not, however, take off, due to the extended acceptance of already known systems and technologies, and the lack of knowledge and uncertainties surrounding the alternative solutions (Londong, 2013; Swart and Palsma, 2013). Alternative sanitation solutions, such as black water and urine-source separation have mainly developed through research and pilot projects, and showcases by universities and water companies (Londong, 2013; Boller, 2013; Swart and Palsma, 2013; Sievers et al., 2016; Morandi and Steinmetz, 2019).

The Netherlands experienced a rather successful role of the Dutch Foundation for Applied Water Research (STOWA) in the initial phases of the early development of alternative sanitation solutions a few years into the twenty-first century. This Foundation broadened the research field and included new stakeholders. STOWA further chaired the creation of a “Coordinating Body” where institutes and water boards were represented: more than half of the water boards in the Netherlands became involved in over 40 research and pilot projects, including vacuum and urine diverting toilets and biogas production from black water. Although STOWA was important, the driving forces were the local parties such as housing associations, water boards, and companies. Drivers are still needed to promote further advance of the sector in the Netherlands, such as a marketplace for nutrients from wastewater fractions (Swart and Palsma, 2013; STOWA, 2006).

Australia experienced a different initial development and implementation of resource-recovery sanitation solutions. The main drivers in the Australian context were both several pollution episodes in the 1980s in Sydney and Melbourne associated with wastewater effluents causing severe episodes of algal blooms, and later water scarcity caused by two decades of long-term drought. These pollution episodes created broader awareness of the impacts of insufficient wastewater treatment. Hence, driven by concerns about nutrients discharge, Melbourne Water decided to retrofit a major treatment plant to produce recycled water for reuse in agriculture. Extended drought conditions that started to affect the state of Victoria in the late 1990s and the consequent water crisis became a further driver for the use of recycled water. Still, the existing dominance of centralized systems meant that the water utilities had less knowledge and experience about new resource-recovery solutions despite them often being the preferred option discussed in workshops (such as between state governments, water utilities, and stakeholders) on community-based water recycling (Fam et al., 2014; Gardner and Sharma, 2013). The implementation of alternative sanitation solutions still today encounters constraints and are often met with skepticism at first by the established traditional mindset of the authorities due to the lack of acceptance and knowledge, as well as the uncertainties surrounding the new solutions (Chapman, 2019).

Below, we introduce and discuss the four central themes in a Swedish 40-year perspective, as these offer explanatory value for the difficulties in establishing alternative sanitation solutions and for the lengthy “lock-in” of the centralized sanitation system.

**Previous Strong National Intervention**

The centralized and public sanitation system was continuously expanded during the entire period investigated in this study (as well as before this and still today).
However, in the 1970s, the expansion was particularly intensive as a result of strong national intervention. On the one hand, the national intervention included increased governmental responsibility for infrastructure development in general (as well as for the environment in the case of the wastewater system), and on the other hand, it included imposing increased responsibility on the part of the municipalities for arranging water services (including sanitation).

Under the motto “big is beautiful,” from the early nineteenth century to the mid-1970s, the Swedish national government took major responsibility for infrastructure development from prioritizing large-scale projects with a strong influence from urban planners and decision makers. It concerned strong governmental control by force and extensive government financial support, standardization, and large-scale, centralized solutions. Apart from the sanitation system, there was also a massive expansion of electrified railways, road infrastructure, and a major public housing initiative (Söderholm and Wihlborg, 2015). The large sanitation systems provided a completely new line of work for planners at all levels of government ministries, such as the Swedish Environmental Protection Agency (EPA), county administrative boards, and municipalities while creating demand for more infrastructure consultants and engineers.

Major public infrastructure projects further required the mobilization of capital and know-how, where The Municipal Reform of 1971—with a large number of small administrative units merging into larger municipalities—can be seen as one way to create a sufficiently large foundation for community planning and major investments (Erlingsson et al., 2010). Hence, in parallel to the great infrastructural development, there was a general trend towards more decentralized governmental decision-making in Sweden ever since the 1950s (Lafferty and Meadowcroft, 2000), with infrastructure becoming a central issue for municipal governance, and with corps of experts and officials at their disposal (Kaijser et al., 1988). The responsibility of the municipalities for arranging water services (including sanitation) has been gradually increased within this context. The criterion has been to arrange water services “in a larger context” (often assessed at 20–30 properties) if needed “due to sanitary inconvenience” (Swedish Government 1955:314, Law on public water and wastewater treatment plants), or “regarding health protection” (Swedish Government 1970:244, Law on public water and wastewater treatment plants). The county administrative boards experienced a tangible upturn in connection with the municipal mergers, from their mission to stimulate and, if needed, force coordinated solutions among municipalities, not least when it comes to centralized sanitation. Hence, in this way, the Swedish technical and political elite had a joint project (Drangert and Löwgren, 2005).

Alongside the increased responsibility imposed on municipalities for arranging water services (including sanitation), extensive government subsidies were granted. Especially throughout the 1970s, these constituted an important component in Swedish municipal wastewater treatment planning (Drangert and Löwgren, 2005). Hence, after the Swedish EPA was established in 1967, and the Environmental Protection Act went into effect in 1969, the governmental subsidy system was aimed at stimulating high-grade biological/chemical treatment (biological and chemical phosphorus treatment: Swedish Government 1968:308). The Swedish national government was a forerunner in stimulating such far-reaching treatment. Most wastewater treatment plants around the world (including in Europe) still lack such treatment. Moreover, the emission levels that the
Swedish wastewater treatment plants must comply with are today often considerably tougher compared to what the EU directive stipulates.

Normally, in the 1960s, a 50 percent investment subsidy was granted by the Swedish national government for high-grade biological/chemical treatment and for pumping stations and sewer pipes. During the budget years 1971/72, 1972/73, and 1973/74, it was possible to get subsidies up to 75 percent of the total investment costs. Directing the subsidies to sewer pipes, pumping stations and large-scale, high-grade biological/chemical treatment, stimulated centralized sanitation solutions, i.e., joint solutions for nearby communities as well as for connecting more remotely located facilities, such as outdoor swimming pools and campgrounds to a community’s wastewater treatment plant (VAV-nytt 1975: 1, 3). Altogether, the national government invested around SEK 1.5 billion (equivalent to about SEK 8 billion in today’s monetary value) in the expansion of large-scale municipal wastewater treatment plants, whereupon the proportion of Swedes connected to public sanitation systems increased dramatically (Swedish EPA, 2018: 3).

Publicly owned sanitation facilities have continued to increase even after the 1970s. The possibility to privatize water and sanitation services was indeed dealt with in a timely manner in the early 2000s in connection to the preparatory work of the replacing Law on public water services (Swedish Government 2006:412). The investigation, however, found that access to water and sanitation services was a basic need/prerequisite for a satisfactory standard of living and, consequently, water services constituted a natural monopoly not suitable for privatization and commercialization. The “natural” municipal responsibility was placed in paragraph 6 of the law (Swedish Government 2006:412) and was further extended by including the environment as one criterion. Paragraph 51 in turn states that it is the responsibility of the county administrative board to supervise the fulfillment of the municipal responsibility to arrange and maintain water services where needed.

Overall, the Law (Swedish Government 2006:412) offers the opportunity to expand public sanitation, but it also implies that the municipality can be forced to expand public water services regardless of costs. Since public sanitation to a large extent is developed in urban areas and cities, issues about the municipal responsibility to arrange water services are mainly raised in “conversion areas” (here meaning areas with old summer houses at commuting distance to larger cities converging to year-round living), in densifications outside existing areas of operation, and in connection with exploitation. There are still a total of about 950,000 properties in Sweden that are not connected to public sanitation, out of which about 450,000 are recreational properties, often with inefficient private on-site sanitation systems (Swedish Government 2018:34).

**Long-Term Challenges Related to the Centralized Solution**

The extensive expansion of centralized large-scale sanitation during the 1970s gradually caused problems, both related to the geographic expansion of the network, and the constantly increasing amount of wastewater to be processed. Since the municipalities to a high degree focused on joint wastewater treatment solutions for “nearby” communities, the expansion largely concerned laying sewer-pipes of record lengths in a short time followed by wastewater pumping stations. Thus, there were many new technical problems to be solved. This is reflected in the sector journal VAV-nytt all through the 1970s; the
The journal was practically filled with technical instructions and information about ongoing investigations, operational studies, standardization processes, etc. for the long sewers, the pumping stations, and wastewater treatment plants (e.g., VAV-nytt 1974: 1, 5; VAV-nytt 1977: 1, 3). Among other things, in 1978, the sector organization VAV appointed a working group for the purpose of compiling instructions for the design of wastewater pumping stations. This working group concluded that the operational interruption frequency was 2.1 operational interruptions per pumping station per year, where the causes of these were insufficient scope of preventive maintenance, operational procedures, operating conditions, and external circumstances (VAV-nytt 1982: 2, 7).

Problems with the pumping stations continued during the 1980s, in part simply because they were increasing in numbers, but also because the equipment involved had become increasingly complicated and the operating personnel’s competence was often not high enough. For this reason, in the mid-1980s, the sector organization VAV, in cooperation with the Swedish Construction Research Board, developed a three-day training program for these personnel (VAV-nytt 1987: 1, 9). Prior to this, VAV had also developed a continuing education course for sewer pipe-installers (VAV-nytt 1982: 4, 5). The background was a field study of 112 two-year-old stretches of sewer pipes made of PVC and concrete conducted by a private consult in 1979. The study showed that already after two years of operation, the sewer pipes showed major defects where deficient installation (sewer pipe-laying) was given as the main reason (VAV-nytt 1979: 3, 3). By the mid-1980s, the education course for sewer pipe-installers had been conducted about 30 times with a total of 750 participants from over 60 municipalities. Henceforth, the course would be given also in the training of sewer pipe-installers at upper secondary schools (VAV-nytt 1984: 3, 10).

The problems with the ever-larger sewer networks also included the fact that hydrogen sulfide, which is very harmful to health, was formed in the wastewater when it stayed in the sewer pipes for a long time (and could leak out). For instance, in the municipality of Linköping there were many problems with this because the sewer network there branched widely in all directions. Thus, ever since the mid-1970s, the sanitation management of Linköping had been working with measures to prevent the formation of hydrogen sulfide (VAV-nytt 1979: 1, 6f; 1980: 2, 5).

Increased Regulation of Wastewater

The treatment requirements for wastewater have increased continuously over the studied period, with peaks in the mid-1970s and early 1980s. The mid-1970s saw the crest of the heavy expansion of 1965–1975, with the government subsidies for high-grade biological/chemical treatment. However, the Swedish authorities continuously increased the environmental requirements on municipal wastewater treatment, and during permit processes the authorities often insisted that the biological wastewater treatment installations should be supplemented with chemical treatment (VAV-nytt 1978: 1; 3; VAV-nytt 1980: 1, 5).

Especially in the early 1980s, the Swedish EPA pressed the issue of chemical treatment and in several cases of permit reviews regarding wastewater treatment plants appealed the decision of the Concession Board for Environmental Protection, to the Supreme Court, such as regarding the wastewater treatment plants of the towns of Skellefteå, Halmstad,
Härnösand, and Arboga. The Supreme Court, in part with reference to the now withdrawn government subsidies,$^2$ either rejected the EPA’s request or postponed (in the case of Halmstad) the final decision for five years (VAV-nytt 1980: 3, 3; 1981: 4, 2). The sector organization VAV was hesitant about the need for chemical treatment, and in 1980, a research collaboration between the Swedish EPA, VAV, and SWEP (Wastewater Works Evaluation Project) was initiated to study how different types of operations influence results and the economy at municipal wastewater treatment plants with biological-chemical treatment (VAV-nytt 1980: 4, 7; 1981: 2, 8). Furthermore, the annual “VAV Day” in 1981 had the heading “Do We Treat Wastewater Too Much?” Because of the highly topical subject, the conference had many participants. The Swedish EPA’s general director Valfrid Pålsson attended in person. He noted that the normal requirement for the municipal wastewater treatment plants would probably remain biological and chemical treatment, with the possibility for deviation from the normal requirement if the municipality could show that, e.g., chemical treatment would have extremely modest effects in relation to the costs (VAV-nytt 1981: 2, 2). In 1982, VAV noted that now the government considered that in addition to full biological treatment, phosphorus reduction would become a general requirement (VAV-nytt 1982: 2, 4). And in 1984, the Supreme Court finally ruled in line with the EPA on chemical treatment in the case of the Kungsbacka Wastewater Treatment Plant. The sector organization VAV determined that the decision could be interpreted such that now also requirements for chemical treatment would be set on all municipal wastewater treatment plants along the west coast of Sweden (VAV-nytt 1984: 4, 4).

**Problems with Sludge**

Alongside the increasing treatment requirements and the continuously growing number of households connected to the sewage systems, the volume of sludge increased rapidly. The main application for the use/disposal of sludge were on farmland or as fill for landscaping (landfill covers, topsoil along highways, etc.) (VAV-nytt 1988: 1, 7). One part of the vision from the very start of the great expansion of municipal wastewater treatment in the 1960s and 1970s, was to use the sewage sludge as fertilizer and soil improvement on farmland. This was a low-cost solution for sludge disposal that also brought benefits to farmers. And yet, this practice was not free of controversy, as there was hesitation from both the public and some farmers about using human waste as fertilizer.

The different views on the value and utility of man’s own by-products have persisted throughout history and across the world. Whereas East Asian civilizations historically have been generally positive towards agricultural application of the by-product, European countries have possessed more of an ambivalent attitude, leaning more towards “dispose” than “recycle”. Although the nineteenth century witnessed an increase in the practice of utilizing the by-product as fertilizer in several countries in Western Europe, this changed when water-based sanitation systems started to be introduced from the second half of the nineteenth century and onwards. This created a disconnection between humans and human waste and reduced the opportunity to use it as fertilizer. Raw excreta volumes from outhouses decreased, and instead new problems started to arise related to the discharge of untreated wastewater into waterways (Olsson et al., 2018).
Despite the hesitance from a growing number about using sewage sludge on farmland, it was not until the 1970s that public guidelines were launched in Sweden regarding how to hygienize the sewage sludge in order to reduce the risks related to pathogens. The 1970s and early 1980s saw an ever more intensified discussion about the possible risks related to other contaminants in the sludge, especially heavy metals and different forms of organic micropollutants, which in time led to increased focus on the need to reduce emissions of heavy metals up-streams (Olsson et al, 2018). During the heavy expansion during the period 1965-1975, with the government subsidies for high-grade biological/chemical treatment, there had never been any discussions on reducing emissions up-stream (Drangert and Löwgren, 2005).

The sector organization VAV worked on the intention to use sewage sludge on farmland. For instance, in the mid-1970s, VAV participated in experiments at the technical agriculture experimental station in Uppsala for the purpose of developing spreading equipment for wastewater sludge on fields because existing manure spreaders had proven to have poor capacity and were too weakly constructed (VAV-nytt 1975: 3, 3). Dewatering of the sludge was another important development field, of which VAV further organized a conference in the mid-1970s (VAV-nytt 1975: 3, 3). This was indeed highly relevant in case the sludge was to be incinerated, an option discussed at the time. Still, this was considered problematic as it would increase the risk of air pollution and new residual products (ash) that must be deposited (VAV-nytt 1988: 1, 8).

By the late 1980s, the “sludge on farmland” situation had intensified considerably due to a now loud public debate, especially from the mid-1980s when the Swedish “environmental guru” Björn Gillberg publicly criticized the major wastewater treatment plant in Gothenburg regarding its emissions of heavy metals. This was followed by a Greenpeace action with the same message (contaminants in sludge from the Gothenburg plant), something which in turn forced the Swedish Farmers’ Association to recommend their members not to apply sludge to fields (VAV-nytt 1989: 3, 10). Hence, this debate strongly affected Swedish food consumers. VAV-nytt refers to a front-page article in the largest Swedish newspaper Dagens Nyheter on November 13, 1987, stating that the agricultural use of sludge could become “an environmental bomb as devastating as DDT”. This newspaper article is referred to as “a biased account in the campaign” being conducted to stop agricultural use of sludge (VAV-nytt 1987: 3, 1).

In parallel to the public sludge-debate in the late 1980s, the Swedish EPA prepared new general advice for the handling of sludge from wastewater treatment plants, and in connection to this concluded that there was no significant environmental risk connected with using sewage sludge on farmland (VAV-nytt 1987: 1, 6; 2, 4f). This was followed by an important statement by the sector organization VAV, the Municipalities Association, and the Farmers’ Association concerning sludge deliveries to farmers in April 1987. The opinion of VAV was that reuse of sludge “must still be correct” as it is nonetheless “the result of effective environmental protection” (VAV-nytt 1988: 1, 8).

However, the consensus was short-lived, and already in February 1989, the chairman of ARLA Foods (an association of dairy farmers) declared that the organization would stop the use of sludge on farmland as of the spring of 1990: “Unfortunately we have ended up in a state of opinion where our environmental profile is not the sharpest. One example is the sludge issue. It can become a new symbolic issue and for that reason it is important to act quickly and decisively” (VAV-nytt 1989: 1, 3). The sector
organization VAV responded repeatedly during the late 1980s by pointing out that (1) no effects on health or the production capacity of farmland could be detected from spreading sludge; (2) sludge from major treatment plants in general had considerably lower levels of hazardous substances than septic tanks (for a small number of families), and further; (3) Swedish sludge as a rule had considerably lower levels of undesirable substances than that in other countries (VAV-nytt 1989: 2, 12; 1989: 2, 1ff; 1989: 2, 17; 1989: 3, 9).

National authorities, in turn, changed position on the issue several times over the 1990s. Hence, in the 1990 government budget proposition, the Minister of the Environment, Birgitta Dahl, stated that additional restrictions concerning sludge could be expected (VAV-nytt 1990: 1, 5). And in 1996, the then-environmental minister, Anna Lindh, declared that: “We must have life cycle … i.e., the sludge will [and can] be used … and all the experts say that sludge works.” She found it “too bad” that major actors such as ARLA kept the sludge from going out onto the fields (VAV-nytt 1996: 4, 20ff).

The uncertain sludge situation had major effects on the sector organization VAV as well as on individual municipalities and plants. The Gothenburg wastewater treatment plant, to cite one example, reduced its disposal of sludge to agricultural fields from about 50 percent in the mid-1980s, to about 20 percent in the mid-1990s (VAV-nytt 1995: 2, 18). This was solved in part by instead storing sludge in a cavern originally intended for oil storage (VAV-nytt 1989: 2, 17). A lot of development work was carried out by sewage managers and wastewater treatment plants, e.g., alternative technology solutions for storage of sludge were investigated (e.g., VAV-nytt 1994: 2, 16). At the same time, quality assurance was considered possible so that the sewage sludge could take place on farmland, and the sector in line with this itself developed stricter guidelines through a certification system which started in 2002 as a development project and was developed into a complete certification system called Revaq in 2008. Revaq, which is run by the sector organization VAV and a steering group in which the Swedish Farmers’ Association and the Swedish Food Federation participate, today ensures the quality of about 40 wastewater treatment plants’ work with upstream measures for sludge quality improvement and recycling of nutrients to enable sludge to be spread on farmland (there are just over 2,000 municipal water and sewerage plants in Sweden). Although the ambition behind Revaq was to establish a long-term plan for dealing with sewage sludge, it has not resolved the issue yet. Not only is the rate of Revaq-connected plants small, Swedish grain mills will in general not accept grains that have been fertilized with sewage sludge, even if it is Revaq-certified, for fear of consumer backlash.

Furthermore, since the launch of Revaq, concerns have been growing about [“new”] contaminants previously not in focus, particularly pharmaceuticals and microplastics. The consequence is that, despite the “upstream work” being highly effective (notably in terms of reducing heavy metal contents in the sludge), the future of sewage sludge application on farmland is looking bleaker. Notwithstanding, and although other countries in Europe have begun to abandon the practice, in Sweden, national policy remains committed to farmland application of sewage. As for the possibility of landfills the sludge, it has been partly banned in Sweden from 2005 (landfilling of sewage sludge is now only allowed if it has been processed through composting). Hence, there is an urgent need to find new methods of dealing with the sewage sludge (Swedish Government, 2020: 3).
When it comes to smaller Swedish treatment plants, intended for a maximum of 2,000 PE, which we will focus more on in the next section, the sludge is often transported to larger treatment plants because it is considered the most economically advantageous option. At longer transport distances, such as in northern Sweden, the sludge can be taken care of on site at the smaller treatment plants. Methods for local disposal consist of, e.g., composting, or long-term storage in so-called sludge lagoons. The sludge can thereafter be used as construction soil. Of the smaller plants, only a very limited number of plants spread sludge on farmland at the beginning of the 2000s, and the situation should not be different now. In fact, sludge from smaller plants is generally not suitable for spreading on farmland as it is often of poorer quality in terms of nutrient content and it is not as desirable as sludge from larger plants. This sometimes causes problems for the larger plants in receiving sludge from the smaller plants (Swedish Government, 2020: 3).

**Challenging Existence for Small Plants and On-Site Systems due to Tightened Environmental Requirements and Neglect**

While the centralized sanitation solution has been strongly stimulated by the prevailing large-scale thinking, the smallest wastewater treatment plants in the smallest communities did not suffer only from fitting poorly into the template, but had a harder time meeting the new emissions regulations (VAV-nytt 1978: 4, 6). Steep requirements for treatment were solved more easily in the larger plants—it was easier and cheaper to control and improve technology, as well as to expand. High-grade biological/chemical treatment in small installations was extremely expensive (Marklund 2018). In addition, the flow and pollution variations were normally “significantly” greater for smaller plants (this refers to the sizes 10–500 PE or 100–2000 PE) than for larger plants (VAV-nytt 1977: 1, 13; 1988: 2, 5), meaning there was great need for frequent sampling, especially for plants providing service to fewer than 200 persons (VAV-nytt 1975: 3, 3). The large variations further meant the small plants were difficult to design. Also, they lacked attention from the scientific wastewater treatment field (VAV-nytt 1988: 2, 5). The limited research that the sector organization VAV could engage in concerned their primary area of activity, i.e., the centralized, large-scale solutions. Although the Swedish Construction Research Board partly engaged in developing solutions other than the centralized solution up until the late 1980s, the research in the sector was thereafter generally weakened in Sweden in favor of research at the universities and technical colleges (Marklund, 1994).

As the smaller wastewater treatment plants of the 1970s and 1980s often were managed and owned jointly by individual property owners who lacked operational and maintenance competence, they were often constructed in a far too complicated manner, and in great need for careful monitoring and care to function satisfactorily. It was not easy for the property owners to choose the right model of package plant in the 1970s as there were numerous plants on the market due to the tightened environmental requirements. Altogether, the problems associated with the smaller plants in the form of maintenance difficulties, challenges in meeting environmental requirements, and a general lack of research to remedy the situation, meant that a system with wastewater pumping stations and pressure mains often was perceived as a more flexible solution. Even if the capital costs were often higher for the connecting pipelines than for
a small package plant, the lower operating costs for the connecting pipelines were considered an offset to this (VAV-nytt 1977: 3, 6; 1978: 4, 6; 1979: 1, 6f; 1980 2, 2f; Marklund, 2018). An overall central motivation for the expansion of the centralized sanitation system in the 1970s, was to relieve small local receiving waters from insufficient and mal-functioning private on-site sanitation systems, and to handle increased treatment requirements more easily (VAV-nytt 1978: 4, 6).

The problems with insufficient on-site sanitation systems persisted during the entire investigation period. In the mid-1990s, about half a million single-family houses for permanent residence and an equal number of summer cottages were considered to have private wastewater treatment installations, usually one’s own at one’s own property, and typically in some form of septic tank (Swedish EPA, 1994) (VAV-nytt 1994: 2, 17). In the late 1990s, the sector organization VAV noted an awakened interest among policy actors and the sector for on-site sanitation systems (VAV-nytt 1999: 2, 43ff), but saw a somewhat split role of the organization in the respect that VAV in ethical terms “obviously” had responsibility for these installations. On the other hand, on-site sanitation systems were not included in the responsibility of the municipal wastewater utilities that fund VAV (VAV-nytt 1998: 6, 12; 1999:2, 43ff). Still, some in the sector wanted VAV to assume a greater responsibility for on-site sanitation installations —“the forgotten wastewater treatment sector”—that accounted for a large percentage of the phosphorous release to water and posed a risk to human health. The proponents reasoned the engagement of VAV with the argument that its members possessed the necessary know-how. Hence, it was a problem that no one took responsibility for the technical development of on-site sanitation installations (VAV-nytt 1999: 1, 9; VAV-nytt 1999: 2, 44).

In response to the awakened interest, VAV initiated the “Small Wastewater Solutions” project in 1998, which in part resulted in the establishment of an ombudsman for the smaller installations (VAV-nytt 1999: 1, 37). This also induced some collaboration between the Municipalities Association, the Swedish EPA, and VAV, where the issue of responsibility for small-scale wastewater solutions was at the forefront (VAV-nytt 1999: 2, 43ff). An important explanation for the growing interest of both VAV and public authorities for the on-site sanitation systems, was that the requirements for better water and wastewater treatment had continuously increased in conversion areas since the 1970s, due to generally increased shower, wash, and WC standards. Often, attempts had been made to solve this through local infiltration of wastewater, which in turn entailed increased risks of contamination of the groundwater (VAV-nytt 2000: 4, 80; SV, previously VAV-nytt, 2007: 1 25).

There was detailed reporting in the early 2000s about projects where the sector participated. The projects investigated, tested, and evaluated various forms of small wastewater systems, such as in the Stockholm archipelago for the purpose of reducing transports and nutrient load on the sea while at the same time supplying more nutrients to agricultural land (VAV-nytt 2000: 3, 38f). In another project, 14 on-site sanitation systems (eight different types) were evaluated at the lake Bornsjön. Results showed that, in principle, it was always better with larger sanitation systems where personnel could monitor and control the process better (VAV-nytt 2001: 4, 40ff). A few years later, in 2008, phosphorous was considered a greater problem and about 20 percent of the Swedish phosphorous that ended up in the already heavily polluted inland sea, the
Baltic Sea,\(^5\) was now considered caused by on-site sanitation systems (Svenskt Vatten 2008: 6, 8f). That same year, 2000 of the roughly 20,000 on-site systems within Värmdö municipality were inventoried, which showed that as many as two-thirds of the property owners were not managing their installations properly. If the situation was similar nationwide, that meant that up to 600,000 of the country’s one million on-site sanitation systems had deficient treatment at the time (Svenskt Vatten 2008: 2, 25f).

When the Swedish Agency for Marine and Water Management was established in 2011, it took over the role from the Swedish EPA of monitoring and guiding authority for wastewater systems <200 PE. Still, the difficulties for property owners to get advice on suitable wastewater treatment from either authorities and/or contractors were repeatedly pointed out it in the sector journal during the following years. It was generally considered both simpler and cheaper for property owners to await inspection than to act on their own initiative as the legislation was interpreted differently by different actors (Svenskt Vatten, previously VAV-nytt, 2013: 2, 28). This was attributed in part to the municipalities’ lack of overall guidelines, and to the fact that there was no authority with responsibility for the role between municipal water services and on-site sanitation systems. Hence, municipalities had to find strategies themselves for reaching out with information and motivating property owners to take remedial measures (Svenskt Vatten 2012: 5, 6ff; Svenskt Vatten 2012: 5, 28; Svenskt Vatten 2013: 2, 28f).

In 2013 and 2014, the phosphorous load from the on-site sanitation systems for about one million people was judged to be as great as the load from all the municipal wastewater treatment plants which serviced about eight million people (Svenskt Vatten 2013: 2, 28). The Agency for Marine and Water Management was further criticized in the sector journal for not cooperating enough with the Swedish EPA, and for proposing requirements that were not compatible with what the Swedish EPA had recently proposed. Also, the Agency was criticized for “going after” all wastewater systems for up to 200 persons, i.e., not only the very worst on-site sanitation systems but also the municipal ones, even though their contribution to eutrophication was/is very small compared to that of those on-site. According to the sector organization VAV, it was an extensive control apparatus and bureaucracy in combination with shortage of resources that “for decades” had put the brakes on the work with on-site sanitation systems, and now the agency’s proposals would increase the problems (Svenskt Vatten 2013: 2, 28f; 2014:2, 28). In the 2010s, the sector journal further typically provided information on municipalities with water and sanitation issues in the outer areas, which to a great degree concern identifying so-called 6 § areas, i.e., areas where the municipalities were obliged to provide services according to the Water Service Act Law (Swedish Government 2006:412). Sometimes the county administrative boards enjoin municipalities to connect groups of properties to the general wastewater system (Marklund, 2018; Svenskt Vatten 2014: 4, 54ff; 2015:5, 6f). Hence, the centralized sanitation solution continues to grow.

**The Great Environmental Mobilization of the Late Twentieth Century**

Combined, the extensive sludge- and emissions-related challenges connected with the major wastewater treatment expansion from the 1960s and onwards, meant the sector was relatively well equipped to meet the approaching environmental mobilization of
the late 1980s and early 1990s in terms of the Brundtland commission, the sustainability discourse, Agenda 21, the Rio Declaration, and the increased focus on life-cycle assessment (LCA). At that point (early 1990s), some in the public debate seriously question the centralized sanitation solution in favor of alternative, smaller scale and “environmentally sound” technologies. In VAV-nytt, the issue was deemed “hot political stuff,” both locally and in the Swedish national parliament. Several municipalities made early decisions on far-reaching changes in-line with the new discourse, such as the towns/cities of Tanum, which prohibited water toilets, Västervik, where water toilets were prohibited outside urban areas, and Oxelösund, Halmstad, and Malmö, where existing treatment systems were supplemented with wetlands and ponds. The issue was brought up in the leadership of Sweden’s largest political party, the Social Democrats, who wanted to introduce UD toilets for the purpose of improving the environment as well as creating 90,000 new jobs (this was included in the party program but was never realized) (VAV-nytt 1994: 2, 27ff).

The criticism of the centralized sanitation solution in the early 1990s was that it could entail hygienic risks in the receiving waters, that it did not recycle nutrients in wastewater, and that it was not “eco-friendly” and resource efficient. Still, the sector was quick to turn the debate to its advantage by promoting wastewater as a resource containing both nutrients and energy that ought to be exploited (VAV-nytt 1992: 3, 38ff). Hence, in a lengthy article in VAV-nytt about the various eras of wastewater treatment technology—the public health era from the 1850s to the 1930s, the environmental protection era from the 1930s and on—it was determined that we are now in the so-called life-cycle era. The life-cycle concept, it was concluded, was not new to wastewater treatment technology. Hence, since the start of the modern wastewater technology history in the mid-1800s, the driving force with roots in the agrarian society was about returning the nutrients in wastewater to food production, to agriculture. Still, reuse had been forgotten during the environmental protection era when wastewater instead came to be seen only as a disposal problem (VAV-nytt 1994: 2, 11).

In line with the societal environmental mobilization, by the mid-1990s, the sector started to highlight wastewater treatment plants as ‘leading environmental companies.’ Sweden now had the world’s most advanced wastewater treatment system, and the sector was well equipped when faced with stricter requirements to secure water supply and reduce environmental impact. The hygienic problems that existed in, e.g., southern Europe were unknown in Sweden. “Even so, wastewater treatment plants are often considered a source of pollution instead of the environmental protection installation it actually is” (VAV-nytt 1994: 2, 13ff). The sector saw Agenda 21 as a “reawakening for the wastewater treatment plants’ environmental work,” and an opportunity for the wastewater treatment operation to clarify its environmental work and share with the municipalities and others the know-how that existed in the sector. The sector organization VAV started a research project with the purpose of giving its members guidance on how Agenda 21 could be used to bring wastewater treatment issues into the municipal decision-making process sooner, and to put forward their competence in these matters (VAV-nytt 1994: 2, 21ff).

Although the alternative sanitation solutions in the mid-1990s were being emphasized by certain groups in society as more environmentally sound and preferable to the conventional system, several articles in VAV-nytt reflect that the wastewater treatment
sector did not yet consider this a realistic alternative. In contrast to the conventional system that was built to protect people from sanitary irregularities, the alternative systems were not yet tested for longer time-periods and in several “eco-villages,” hygiene had even become a problem because of the alternative technology. For one thing, more people were exposed to toilet waste with dry systems than with conventional systems, one in every family when dry systems were used versus one in ten thousand with conventional systems (VAV-nytt 1994: 2, 27ff; 1995: 2, 37). In 1995, VAV-nytt reported on Åkesta eco-village in Västerås where containers for the biological toilets were in shafts under the houses. From these, fecal matter was carried in buckets up a steel stairway and passed living spaces before coming out of the houses. There had been such major problems with flies that they resorted to insecticide and then buried the fecal matter as waste in the forest. When it comes to the diverted urine, groundwater infiltrated the pipes for urine transport between the houses and the urine tank, so the farmer had to empty the tank 10–15 times more often than if only urine had been in the tank (VAV-nytt 1995: 2, 37). Alternative solutions in the form of dry and close-to-nature solutions could work well on a small scale with the right planning and care, the sector organization VAV concluded, but there was an inherent weakness in that the households themselves would have to take more responsibility and do more work. And was it even more ecological? It could just as well be the conventional system. Hence, the water quality in local receiving water bodies in the outer-town areas had improved steadily for decades due to the conventional solution (VAV-nytt 1995: 2, 33).

In particular, the issue of UD was criticized in VAV-nytt in the mid-1990s. The gist of the criticism was that there were too many questions related to the big infrastructural requirements that three waste streams (urine, fecal matter, and graywater) would impose on the buildings. How will fecal matter collection and treatment, and urine collection, transport, and storage (awaiting spreading on farmland) come about (VAV-nytt 1995: 2, 33)? There were further only two variants of porcelain toilets with water flushing for separation of urine and fecal matter on the Swedish market (VAV-nytt 1995: 2, 37). On a longer term, it was not ruled out that UD would compete with the conventional system in sparsely populated areas, the sector organization VAV believed. Still, it was highly uncertain whether UD systems entailed higher or lower energy use than a conventional system. Research was also lacking on several decisive points, such as how the urine was to be stored to prevent the release of ammonia and bacteria to the atmosphere (VAV-nytt 1996: 2, 50ff; 1995: 2, 37). Environmental minister Anna Lindh stated in an interview in VAV-nytt that there was nothing that got wastewater treatment people as distraught as when UD comes up. Still, she believed it would be necessary in the future and therefore wanted to test it on a larger scale (VAV-nytt 1996: 4, 20ff).

The UD issue was still topical in the sector journal in the late 1990s. In 1999, readers were informed of a report from SLU [Swedish University of Agricultural Sciences] presenting UD as “very interesting in the countryside if there are farmers who are interested in the urine.” Still, it could not yet be recommended in larger communities (VAV-nytt 1999: 1, 40). In 2003, Håkan Jönsson (researcher in “LCA technology” at SLU) wrote in an opinion piece that Sweden ought to invest in UD technology because it provided business opportunities “now” (around 2000) when the interest in source separating systems was rapidly increasing internationally. Jönsson referred to a German study investigating the use of UD in a new, larger housing area, and a China study of an area where
about 100,000 source-separating toilets had been installed in recent years. According to Jönsson, Swedish solutions could contribute in solving the world’s wastewater problems as Sweden was “at the forefront” in source-separating wastewater (SV, previously VAV-nytt, 2003: 1, 40ff).

Problems persisted into the 2000s in finding farmers who were interested in the nutrients from wastewater, and the handling and transport were still difficult and expensive (VAV-nytt 2003: 2, 40f; Svenskt Vatten 2004: 2, 34). Even if urine separation was in theory an advantageous solution for nutrients recovery it was hard to make it work in practice in the mid-1990s as well as 10 years later. There was no alternative to the conventional wastewater treatment technology where stringent requirements on hygiene, environmental protection, durability, economy, and safety in function were met. VAV considered it best to keep working with the conventional centralized system (VAV-nytt 1995: 2, 20ff; 2003: 3, 28ff).

Source separation is mentioned also in later years in the sector journal, if not nearly as intensively as in the late 1990s. In 2007, e.g., there was a report of an experiment with waste disposal on the initiative of an Environmental Office in Gothenburg, to find a new alternative route for the return of plant nutrients (Svenskt Vatten 2007: 3, 29ff). In 2012, the readers of the sector journal were, in turn, told that Örebro municipality emphasized source-separation as an alternative to the conventional system in conversion areas (Svenskt Vatten 2012: 5, 6ff), and in 2015, that the Swedish Agency for Marine and Water Management argued that a transformation to more source-separating technology ought to happen in the future. In that year, the sector journal further reported on experiences with source-separating technology in the Netherlands (Noorderhoek) and Germany (Hamburg Wasser), including biogas-producing reactors that process wastewater (blackwater) from vacuum toilets (Svenskt Vatten, April 2015, International Special, 23ff).

Discussion and Conclusions

From delving into the Swedish sanitation sector journal, VAV-nytt/Svenskt Vatten, we have gained a deeper understanding of the interactions and processes that have preceded and reinforced the centralized sanitation solution in Sweden ever since the 1970s. Not least, we have seen a strong example of policy intervention for (if not transition per se) a major and long-term upgrading/development of the centralized sanitation system. In the 1960s and 1970s, the Swedish centralized sanitation system, due to strong national intervention, underwent large-scale development steps in terms of improved biological and chemical treatment steps. An interplay of different policy areas was involved, including municipal and environmental reforms, large-scale infrastructural visions, government control, and extensive state subsidies. This included important resources to solve technical and environmental challenges that inevitably arose in parallel with the expansion. Overall, this is important guidance on how policy actions can catalyze resources to spur progress and stimulate development in a certain direction, thus recognizing intervention points that ultimately lead to a broader sustainability transformation. When the negative environmental effects of the centralized sanitation systems were realized in the 1950s and 1960s, Sweden already had an efficient machinery for rapid and large-scale infrastructural expansion that could be redirected
towards the large-scale improvement of the centralized sanitation system in terms of biological and chemical treatment steps. Here, it is important to point out that the major national policy instruments used then, are still available for use today.

Transition has not taken place when it comes to alternative sanitation solutions, and it is in the recapitulation of interactions in practice (and the possible absence of these) between niches, the regime, and the landscape level in the Swedish case of wastewater treatment over a period of 40 years that we can understand the lack of such a transition on a deeper level. Hence, although there was a possible “intervention point” for policy in the direction of alternative sanitation solutions in the late 1980s and 1990s, in terms of a strong sustainability discourse with important undertones of recycling, and a parallel public debate which seriously questioned the centralized sanitation system, there was already (and still is) a tension in the regime leading to an inevitable lock-in towards the centralized sanitation solution. This tension is between nutrients and resource recovery on the one hand, and containment, treatment, and risk management of the wastewater streams on the other, outermost crystallized in the challenging sludge issue but with obvious consequences for all resource-recovery contexts in the wastewater sector. The tension is further connected to strong perceptions of where human feces belong and do not. In this context, even if the alternative sanitation solutions would have worked well hygienically and technically in the early 1990s (i.e., in time for the possible intervention point), due to the tension there was a problem in finding farmers interested in the recovered resources as it could negatively affect their ability to sell their farm products. Hence, the alternative sanitation solutions lacked obvious advantages over the centralized solution, even in terms of resource-recovery.

The dominance of the centralized sanitation solution in terms of treatment and risk management was continuously reinforced over the studied period, due to the long-term overcoming of many health and environmental challenges. This was continuously highlighted in the sector journal, along with the recycle-heritage of the conventional system, and how it would still be if only the agricultural sector drew its bit. In parallel, the shortcomings of the alternative (and smaller) solutions were constantly highlighted over the period. Important challenges to be addressed for alternative sanitation solutions to have a greater impact in Sweden as well as internationally, include creating a marketplace for nutrients from wastewater fractions and linking new sanitation solutions (e.g., greywater or urine separation) with conventional wastewater infrastructures (Morandi and Steinmetz, 2019). Policy intervention is thus appropriate in both the market and technology development fields. Policy would be needed to create a “niche” for the new solutions as it is really only then these can be developed and improved. In other words, it is not always possible to wait for new technology to improve, it needs to be partially protected (expanded) to get a chance to improve, i.e., not only through research but also from learning-by-doing, and learning-by-using (see, e.g., Kemp et al., 1998; Wilson, 2012).

In Sweden, policy has not intervened adequately to achieve this. For instance, LIP was a very broad (technology-neutral) investment in sustainability, whereby the actual investment in alternative sanitation solutions was limited in both scope and time, not least in comparison with what the STOWA projects in the Netherlands seem to have achieved at about the same time (STOWA, 2006). Also, even though the evaluation of LIP in 2005 suggested a follow-up, in reality the political forces that pushed for alternative sanitation solutions...
solutions some 30 years ago in the name of recycling are weaker today and have partly
devolved into the climate issue, and with increased focus on energy aspects alongside
the circular economy (Swedish Government 2020:3).

Overall, to argue in this context that it is mainly about the lack of acceptance and
knowledge of the alternative solutions, or technical lock-in, which is often done in the
literature, is to simplify, at least in the Swedish case. Hence, with the tension between
recovery and treatment/risk management in the regime still highly relevant—alongside
the remaining needs for promising experiments and radical innovations on alternative
sanitation solutions—the challenges related to achieve the transition towards alternative
sanitation solutions are indeed more complex and would benefit from a more mission-
oriented policy (Maccucato, 2018) on a par with that which existed, especially during the
1970s and 1980s, in the upgrading/development of the Swedish centralized sanitation
system.

Notes

1. The investments in high-grade biological/chemical treatment are reflected in the continued
   expansion and centralization of the wastewater treatment system. Hence, in 1974, 540,000
   new individuals were connected to biological-chemical treatment at the same time as the
   number of treatment plants decreased by 65 (VAV-nytt 1975: 1, 3). In 1975, 25 high-
   grade biological/chemical treatment plants were constructed and about 40 treatment
   plants were shut down (VAV-nytt 1977: 2, 7).
2. The government subsidies for sewer systems and wastewater treatment plants ceased
   through the so-called savings bill (1978/79:95).
3. An employer and industry organization for companies that produce food and drink in
   Sweden.
4. Statens råd för byggnadsforskning, usually called Byggforskningsrådet (BFR), was a former
   Swedish agency that existed from 1960 until 2000. BFR was a research board that conveyed
   funding for research within the fields of construction and urban planning. As of January 1,
   2001, BFR’s mission was taken over by the Swedish research council Formas.
5. The Baltic Sea is severely affected by eutrophication (much due to emissions of phosphorus)
   and is one of the world’s most polluted seas. It borders Denmark and Sweden to the west,
   Finland, Russia, Estonia, Latvia, and Lithuania to the east, and Poland and Germany to the
   south. With its 2,400 kilometers, Sweden’s coast is one of Europe’s longest. Since the end of
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