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Estimating environmental and societal impacts from scaling up urine concentration technologies

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ABSTRACT

There is a growing trend for nutrient recovery from wastewater as part of the transition to a circular economy. Most nutrients in household wastewater originate from urine and one way to facilitate reuse of these nutrients is to concentrate the urine into fertiliser products. Urine concentration technologies are still in the development phase and not implemented at scale. The aim of this study was to provide guidance to technology developers and policymakers by assessing the environmental and societal impacts of urine concentration technologies. In particular, it includes practical aspects such as worker safety, space availability and local fertiliser needs that have not been included in previous studies. Future scenarios on implementing three different urine concentration technologies (alkaline dehydration, nitrification-distillation, ion-exchange with struvite precipitation) in a planned residential area in Malmö, Sweden, were developed. The technologies were evaluated using multi-criteria assessment (MCA), with environment, technical, economic and health sustainability criteria derived from the Sustainable Development Goals (SDGs). It was found that all urine concentration technologies performed well against many of the sustainability criteria examined and can contribute to achieving SDGs, especially regarding nitrogen recovery. Specific areas for further development were identified for each technology. An impact assessment on scaling up demonstrated that nitrogen emissions to surface water were significantly reduced when more than 60% of urine in Malmö city was subjected to urine concentration. Nitrogen and phosphorus recovered from recycling only 15–30% of urine in Malmö could supply 50% of Malmö municipality's fertiliser demand.

1. Introduction

Growing populations and expanding cities are generating more wastewater, increasing the amount of pollutants, such as the eutrophying nutrients nitrogen (N) and phosphorus (P), arriving at wastewater treatment plants (WWTPs) (Larsen et al., 2013). Even with effective treatment, increasing urban population will lead to a net increase in eutrophying nutrients in waterways, with negative effects on aquatic life (UN Sustainable Development Goal (SDG) 14) (van Puijenbroek et al., 2015). While N and P cause environmental problems in waterways, they are also commonly required as fertilisers in agriculture. Fertiliser use is currently linear, resulting in accumulation of nutrients around large cities and leading to soil nutrient stripping and decreased soil fertility (Harder et al., 2019). The resulting imbalances in biogeochemical flows of N and P now transgress the planetary boundaries of a safe operating space for human activity (Steffen et al., 2015). For these reasons, many experts recommend extending the use of human excreta as fertiliser, to enable a more closed-loop society in line with SDG12 (Responsible consumption and production) (Drangert et al., 2018; Guest et al., 2009; Harder et al., 2020; Larsen et al., 2009).

Separate treatment of urine would enable its reuse as fertiliser, because most nutrients found in wastewater originate from urine (Fumasoli et al., 2016), e.g. approximately 80–90% of N and 50–80% of P in human excreta are found in urine (Vinnerås, 2001). The potential for resource recovery can be an important driving force for introduction of urine separation (Larsen et al., 2021a). However, urine comprises 95% water and <1% N (Vinneräs et al., 2006), i.e. compared with chemical fertilisers the nutrient concentration in urine is low. According to Vinnerås et al. (2006), an adult produces at least 10 L urine per week, meaning that every 1000 people in a city produce over 10 m^3 of urine per week. Managing (e.g. storage and transportation) large volumes of urine separately from other wastewater poses logistical challenges and

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concerns about energy use. By removing water or extracting the nutrients from urine, the volume can be greatly reduced, facilitating transport and nutrient reuse (Etter et al., 2015; Senecal, 2020). Larsen et al. (2021b) charted the development of technologies for urine treatment 2006–2019 and observed significant development, with some technologies entering the next development phase of industrial optimisation.

Several literature reviews have documented the state of knowledge regarding technologies for urine treatment and provided qualitative comparisons of technology performance (Larsen et al., 2021b; Martin et al., 2020). Life-cycle assessments of municipal wastewater systems with urine diversion show that urine diversion can improve environmental performance, particularly greenhouse gas emissions, relative to conventional systems (Besson et al., 2021; Hilton et al., 2021; Lam et al., 2020). Other studies have shown that economic costs of implementing urine diversion systems are similar to conventional systems (Ishii and Boyer, 2015). Promising urine treatment systems entering the next development phase include alkaline dehydration, nitrification-distillation and ion-exchange with a struvite precipitation pre-step. These three technologies have a technology readiness level (TRL) of 6, meaning that prototypes have been tested in relevant environments (Etter et al., 2015; Fumasoli et al., 2016; Simha et al., 2020c; Tarpeh et al., 2018a). All these systems have the ability to recover >80% N and P from urine. Fertilisers produced from these technologies have high fertiliser efficiencies for both N and P, similar to mineral fertilisers (Martin et al., 2021). While previous studies have focused on environmental impacts and costs, multi-dimensional sustainability aspects of full-scale implementation of such technologies has not been assessed. Since development is reaching the operational stage, it is important to identify areas for optimisation, to guide policy and decision-making on use of these technologies.

The aim of this study was to assess the environmental and societal impacts of large-scale implementation of urine concentration technologies, in order to provide guidance to technology developers and policymakers. It was assumed that these urine concentration technologies were implemented in a new development area in Malmö, while the rest of the city remained connected to the existing wastewater system. Specific objectives were to assess (i) the advantages and disadvantages of different urine concentration technologies and (ii) the level of implementation of urine concentration technology required for net benefits in Malmö (population ~350,000). First, we present the investigated urine concentration technologies and the study context scenarios. The methods section describes how we use a multi-criteria assessment (MCA), with criteria derived from sustainability assessment literature and the SDGs and their targets, to investigate practical aspects such as worker safety, space availability and local fertiliser needs that have not been included in previous studies. The results and discussion section presents an overall performance matrix and evaluation of each technology against specific criteria. Finally, we use future scenarios to evaluate the effects of scaling different urine concentration technologies from a Swedish perspective. Implications for potential SDG contributions, development and implementation strategies for urine concentration technologies at scale, as well as areas that require further research are presented in the concluding section.

2. Urine concentration technologies and study context

2.1. Alkaline dehydration

A method for drying fresh urine in an alkaline medium has been developed at the Swedish Agricultural University, first described in Senecal and Vinnerås (2017). Urine is collected in urine-diverting toilets or urinals and can be treated at household level (Senecal and Vinnerås, 2017), or semi-centralised if a limebox is installed in the toilet (Senecal, 2021). Urine drying in an alkaline medium (pH > 10) prevents hydrolysis of urea and thereby N losses (Simha et al., 2020b). A dehumidifier is used for dehydration, resulting in a dry powder.

2.2. Nitrification-distillation

A method for nitrification-distillation of urine has been developed at EAWAG, as described in Etter and Udert (2015). Urine is collected in urine-diverting toilets or urinals and conducted to a storage tank, from which it is pumped to a nitrification column. Nitrification is used to stabilise N compounds in the urine, to prevent loss of N and odours. Distillation by a vacuum distiller is used to evaporate the water from the urine (Etter and Udert, 2015). The result is a liquid concentrate.

2.3. Ion-exchange and struvite precipitation

The ion-exchange method, with struvite ($MgNH_4PO_4 \cdot 6\ H_2O$) precipitation, recovers selected nutrients (N by ion-exchange, mainly P by struvite precipitation) (Etter and Udert, 2015; Tarpeh et al., 2017). Urine is diverted and collected separately and stored in sealed containers to keep the urine composition constant by preventing N losses through ammonia (NH_3) volatilisation (Tarpeh et al., 2018a). During storage, urea is converted by hydrolysis to ammonium (NH_4^+), which adsorbs to the negatively charged ion-exchange resin (Tarpeh et al., 2017). Based on Tarpeh et al. (2018a), Dowex Mac 3 resin with adsorption density of around 4 mmol N/g was assumed to be used and the resin was assumed to be regenerated with sulphuric acid. Struvite is precipitated from the urine by adding a magnesium source and is separated from the liquid by filtering (Antonini, 2013; Etter and Udert, 2015). The dried product is a powder (McConville et al., 2020).

2.4. Scenarios

The future scenario for a new urban development (Sege Park) in Malmö investigated whether urine concentration technologies can help achieve selected SDGs. The ambition is for Sege Park to become a testbed of urine diversion, in at least one building, to recover nutrients and reduce emissions (Malmö Stad, 2015). However, the net impact of implementation of urine diversion in just one building would be negligible. Therefore, in one future scenario, urine concentration technology was assumed to be applied in all 700 new homes in Sege Park (2100 residents). It was assumed that each person produced 550 kg (~550 L) urine per year (Vinnerås et al., 2006) and that two-thirds of the urine was treated with urine concertation systems to account for losses such as urine excreted at other locations than at Sege Park and losses when collected by the urine diverting toilet (von Münch and Winker, 2011). All greywater was assumed to be produced at home. Based on the assumed composition of household wastewater produced per person and year (Table S1.1 in Supplementary Material (SM)), the urine fraction comprised around 1% of the total volume. Assuming concentrations of 7300 mg N/L and 660 mg P/L, based on suggested design values in Vinnerås et al. (2006), the urine produced in Sege Park contained 75% of N and 45% of P in total household wastewater from the area.

The baseline scenario was a conventional system where household wastewater was assumed to be transported in the conventional wastewater network. System expansion was used to calculate the energy demand associated with commercial N and P fertiliser production, which was assumed to be replaced by N and P fertiliser produced by urine concentration. The urine concentration technologies investigated were: alkaline dehydration, nitrification-distillation and ion-exchange with struvite pre-precipitation. For all urine-concentrating technologies, it was assumed that the urine was captured by urine-diverting toilets and transported to a semi-central treatment plant within the residential area (Fig. 1). The blackwater (urine excluded) was assumed to be transported to Sjölunda WWTP in Malmö (capacity 550,000 pe) (VA SYD, 2019), using the conventional wastewater network. The alkaline dehydration system included a *limebox* in the toilet, which added magnesium oxide (MgO) to the urine to prevent urea hydrolysis (Simha et al., 2021a). For the ion-exchange system, the residual waste after stripping the urine (Tarpeh et al., 2017) was assumed to be transported to the WWTP. All N

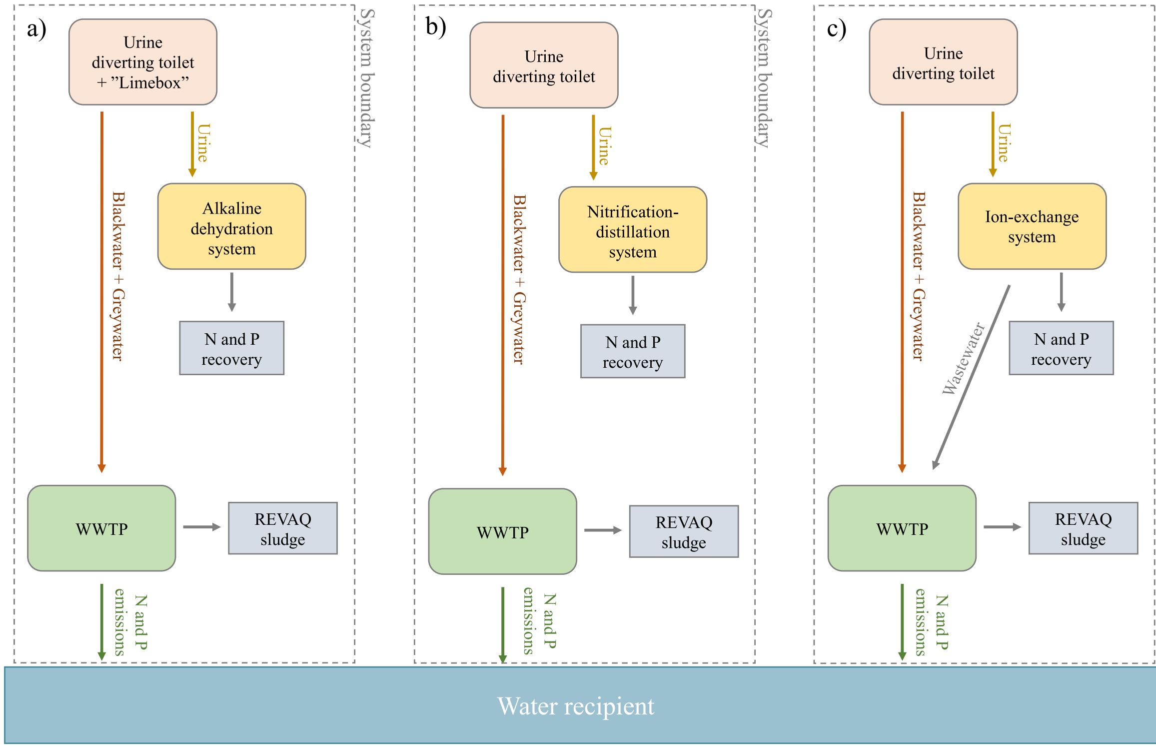

Fig. 1. Schematic illustration of the different urine concentration systems: a) alkaline dehydration; b) nitrification-distillation and c) ion-exchange.

and P products recovered from the urine concentration technologies and REVAQ-certified sludge from Sjölunda WWTP were assumed to be applied to land (VA SYD, 2019).

3. Methods

Multi-criteria assessment was used to evaluate the sustainability of different systems for urine concentration, using criteria derived from selected SDGs. Sustainability is a concept that includes complex interactions between nature and society. Assessing sustainability thus requires the use of multiple perspectives, including use of qualitative and quantitative data. MCA is a transparent process to support decision-making by comparing different options based on multiple criteria. It is thus capable of handling the inherent complexity and broad scope of sustainability, including diverse types of data inputs (Lindfors, 2021). Practically however, the method cannot cover all aspects of sustainability and is more suitable for comparing different systems performing the same task, rather than providing information about the total anthropogenic impact (Hellström et al., 2000). Thus, this study was not holistic, but rather examined selected key aspects for technology development (see 3.1). Fig. 2 provides an overview of the study method and shows how the MCA criteria, their respective indicators and the indicator performance scales were determined from the SDGs and literature. The urine concentration technologies were then assessed based on their performance in order to identify implications for the SDGs, as well as development and implementation strategies. To determine the wider implications of implementing urine diversion and concentration in Malmö, analysis of the impacts of scaling up was performed for two criteria: N and P emissions from WWTP to surface water and total fertiliser production.

3.1. *Selection of criteria*

There are 17 SDGs. SDG6 directly targets sanitation and wastewater treatment, but sanitation is closely linked to several other SDGs. According to the Sustainable Sanitation Alliance, sustainable sanitation should protect human health and the environment and be economically viable, socially acceptable and technically and institutionally appropriate (SuSanA, 2017). We used this definition, as well as literature on sustainability assessments (e.g., Cossio et al., 2020) to define four assessment categories in the MCA: i) environment; ii) technical function; iii) economic and iv) health (Table 1). In order to formulate criteria and indicators for each category that we considered relevant for this study's context, we reviewed literature on sustainability assessments of wastewater treatment systems in general (Cossio et al., 2020; Schütze et al., 2019), and in a similar context (Hellström et al., 2000; Johannesdottir et al., 2021; Vidal et al., 2019), as well as reviewing specific SDGs and their targets. Note that the aim of the MCA is not to assess the total sustainability impacts of the system, but rather to provide guidance to technology developers and policymakers. Thus, impacts associated with e.g. manufacturing of chemicals used is not included, while the cost of using chemicals is. The social and institutional categories were excluded due to uncertain data relating to the novelty of the technologies.

3.2. *Assessment of indicators*

Each indicator was assessed separately using a performance scale with specific limits developed for each criterion based on relevant literature and/or data providing perspectives on the indicator, such as threshold values. The performance of each technology for each indicator was graded on a scale of 1–5, with stoplight colour coding for visualisation. Full details of the assessment methodology and indicator grading scores are presented in Sections S1-S4 in SM.

3.2.1. *Nutrient emissions*

Data on wastewater volume and nutrient concentrations (mg/L) in influent and effluent from Sjölunda WWTP, and the corresponding N and P removal rates, were taken from the latest available environmental report (VA SYD, 2019). Emissions of N and P when the Sege Park fraction was added to the WWTP were then calculated for the conventional system and for the urine concentration systems. Calculation of volume and nutrient concentrations in household wastewater from Sege Park

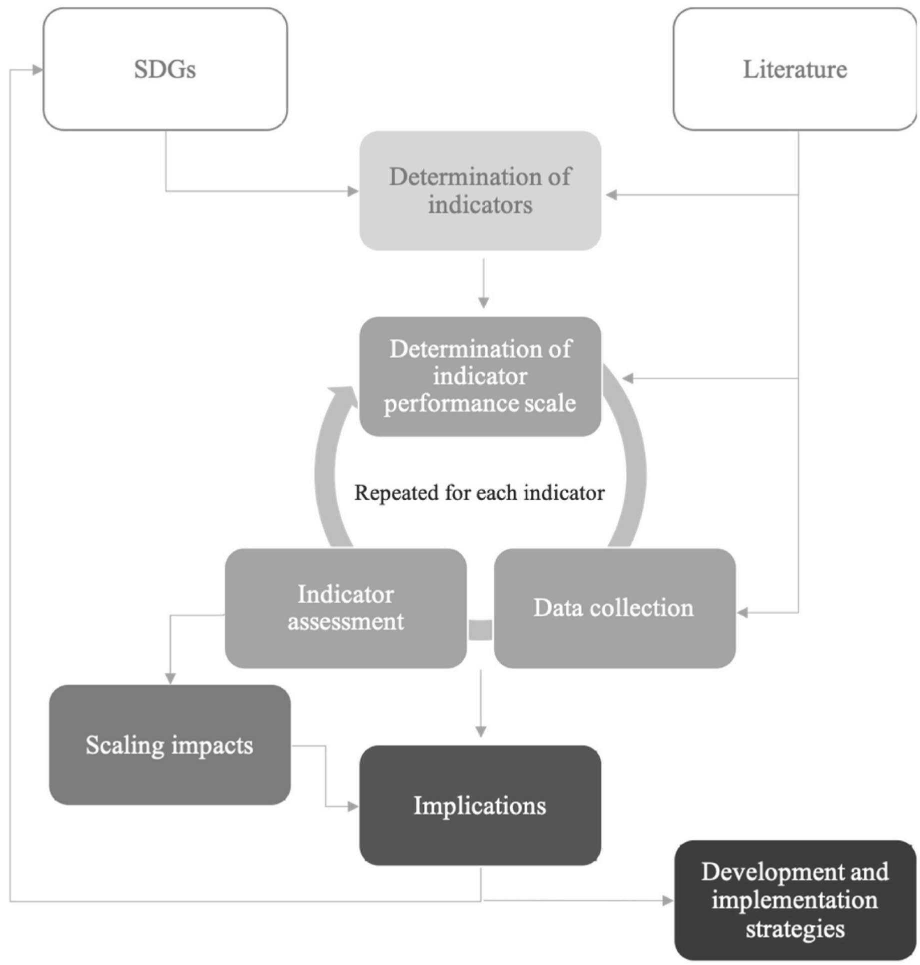

Fig. 2. Flow diagram providing an overview of the method of the study i.e., how criteria and indicators were defined, investigated and assessed to provide implications on SDG contributions as well as development and implementation strategies.

was based on assumed characteristics of the wastewater (Tables S1.1 and S1.3 in SM). The performance in MCA was graded based on N and P concentrations in WWTP effluent for the systems studied (S1 in SM).

3.2.2. *Nutrient recovery*

Nutrient recovery for the conventional system was calculated based on N and P concentrations in the REVAQ-sludge. Sjölunda WWTP has 94% P removal efficiency (VA SYD, 2019) and it was assumed that all precipitated P ended up in the sludge. Regarding N recovery, about 20–30% of N in the wastewater ends up in the sludge but only 50% of this remains in the sludge after dewatering (Jönsson, 2019). From this information, N recovery for the conventional system was assumed to be 15%. For the urine concentration systems, percentage recovery from faeces and greywater in the WWTP was assumed to be the same as for the conventional system. Nutrient recovery from urine concentration technologies was calculated using potential maximal recovery values, to enable fairer comparison between the different technologies, since they were at different stages of development and optimisation. In recovery calculations, potential N loss in pipe transport and urine storage tanks was not considered in this study due to uncertainties related to the magnitude of the potential N loss and how the potential loss differed between the different systems. Results from this study are thus indicative of recovery potential and not absolute values. Performance grading was determined based on percentage N and P recovery from all household wastewater produced in Sege Park, and excluded other important nutrients such as potassium and micronutrients (S1 in SM).

3.2.3. *Energy*

Energy demand was calculated based on the energy input required to treat the urine produced in Sege Park with the different urine concentration technologies. Energy consumption connected to manufacturing of chemicals/materials used in treatment was not taken into consideration. Energy consumption for the conventional system was calculated using data on energy demand for N removal in the WWTP and for production of equal amounts of N and P (as commercial fertilisers) to those recovered from the urine concentration systems. The energy demand for the technologies was compared against annual energy use in a residential apartment (assumed 3600 kWh/yr (Vattenfall, 2020)), in order to determine the performance (S1 in SM).

3.2.4. *Space*

The nitrification-distillation technology requires 5 m^2 per reactor and a total of 10 m^2 per unit (Etter and Udert, 2015). The alkaline dehydration technology was assumed to require about 3 m^2 per reactor, based on a field study by Simha et al. (2020c). The total area required was then assumed to be double the actual physical space of the reactor, i.e. 6 m^2 per unit. For the ion-exchange system, no data were found concerning the space required for this set-up. Therefore, an estimate was made based on the number of components in the system and their approximate size and urine storage needs. A similar space requirement as for the nitrification-distillation system was assumed, 10 m^2 per unit. The number of treatment units required for each system was determined based on treatment capacity and the criteria were graded considering half the suggested parking space per household in Sege Park (6.25 m^2)

Table 1
Criteria derived from the SDGs and specific targets investigated in multi-criteria assessment, and the indicators used.

Category	Criteria	SDG	Analysis	Indicator
Environment	Eutrophying emissions	14 – *14.1 ...prevent and significantly reduce marine pollution of all kinds...*	Quantitative	Concentration of N & P in WWTP emissions (mg/L)
	Nutrient recovery	6 – *6.3 ...substantially increasing recycling and safe reuse globally*	Quantitative	% N & P-recovery from total household wastewater produced in Sege Park
		9 – *9.4 ...upgrade infrastructure and retrofit industries to make them sustainable, with increased resource-use efficiency...*		
		11 – *11.3 ...integrated and sustainable human settlement planning and management...*		
		12 – *12.5 ...substantially reduce waste generation through prevention, reduction, recycling and reuse*		
	Energy demand	9 – *9.4 ...upgrade infrastructure and retrofit industries to make them sustainable, with increased resource-use efficiency...*	Quantitative	% of annual apartment energy demand for urine concentration
Technical	Space efficiency	11 – *11.3 ...integrated and sustainable human settlement planning and management...*	Quantitative	Space required in residential area (m2/household)
Economic	Cost	6 – *6.2 ...adequate and equitable sanitation...*	Quantitative	Annual cost per household (SEK/yr)
Health	Working environment	3 – *Ensure healthy lives and promote well-being for all at all ages*	Semi-quantitative	Additional risk with urine concentration
		8 – *8.8... promote safe and secure working environments for all workers...*		

(S2 in SM).

3.2.5. *Cost*

For the alkaline dehydration technology, the annual cost was estimated based on components in the system set-up in the field study performed by Simha et al. (2020c). The nitrification-distillation set-up was assumed to be that described by Etter and Udert (2015) where the costs for the different components are presented. The cost for the ion-exchange set-up was estimated based on components and costs presented in previous studies (Antonini, 2013; Etter et al., 2015; Ishii and Boyer, 2015; Kavvada et al., 2017; Landry and Boyer, 2016; Tarpeh et al., 2018a). Costs for each system included the capital costs, operation & management (O&M) costs and the profit from urine N and P fertiliser sales, and were compared to the usage fee in Malmö and the average usage fee in Sweden (S3 in SM).

3.2.6. *Workers' health*

All systems were assumed to be safe to use for the population in Sege Park, because the only difference to the conventional system at household level for the different systems investigated was the toilet (urine-diversion flush toilet instead of conventional flush). For the alkaline dehydration technology, the fertilisers made from recovered nutrients were assumed to be free from pathogens after four days of storage at 20 °C (Senecal et al., 2018). Use of the nutrient concentrate from nitrification-distillation was also assumed to be safe, because the technology has complete pathogen inactivation due to the distillation (Etter and Udert, 2015). For the ion-exchange system, it was assumed that treatment followed recommended Swedish guidelines so that the urine was stored for 30 days prior to treatment, making it safe to use on food and fodder crops that are to be processed (Schönning and Stenström, 2004). Since pathogen removal was assumed, the health assessment focused on health risks associated with operating the different systems. Thus, the MCA focused on additional hazards for workers' health arising from the urine concentration systems considering the chemicals used, system maintenance, potential for process failure and pathogen exposure from partially treated urine (Etter and Udert, 2015; McConville et al., 2020; Senecal, 2020; Tarpeh et al., 2018a). Information about hazardous events associated with the baseline WWTP was taken from a summary by IVL (2020) and applied to all systems, since all future scenarios used the WWTP for blackwater treatment. Health risks were assessed using a semi-quantitative risk assessment in the manual *Sanitation Safety Planning* by WHO (2016) (S4 in SM).

3.3. *Impacts of scaling up*

To investigate the level of implementation of urine concentration technology required to make a difference in Malmö, impacts of scaling up on N and P emissions to surface waters and fertiliser production were examined. Nitrogen emissions resulting from connecting different proportions of the population to a urine-diverting sanitation system were calculated using data on concentrations of nutrients in the influent and total volume of wastewater treated at Sjölunda WWTP (VA SYD, 2019). Based on the characteristics of the wastewater (Table S1.1), urine diversion would result in 79% N and 49% P removal from wastewater for alkaline dehydration and nitrification-distillation systems, and 78% N and 46% P removal for ion-exchange with struvite precipitation systems. Nitrogen removal efficiency at the WWTP was assumed to remain 70% for N and 94% for P.

To calculate the proportion of urine produced in Malmö that would need to be captured as fertiliser in order to meet local demand for fertiliser, the area of available agricultural land was multiplied by mean fertiliser dose per ha. Malmö municipality has 4588 ha agricultural land (Malmö Stad 2020a) and mean fertiliser dose per ha was assumed to be that in the local region of Skåne: 8 kg P and 122 kg N per ha (SCB, 2020). According to Martin et al. (2021), urine-based N and P fertilisers have similar efficiencies to mineral fertilisers, so we assumed that the recovered N and P had the same fertilising capacity as commercial N and P fertiliser. The proportion of urine produced in Malmö that would need to undergo urine concentration in order to cover the fertiliser requirement in Malmö was calculated based on N and P concentrations in urine reported by Vinnerå;s et al. (2006). The population of Malmö was taken as 347,949 (Malmö Stad, 2020b). The calculations were performed for four different recovery rates.

3.4. Limitations

Important sustainability criteria such as social acceptance and risk associated with reuse were not considered, as these have been covered in other studies. Social surveys of consumers and farmers have found acceptance for the concept of urine reuse (Ishii and Boyer, 2016; Simha et al. 2017, 2021b; Segrè Cohen et al., 2020). There are potential risks with micropollutants (e.g. pharmaceutical residues) in urine-derived products, necessitating an additional treatment step to eliminate these (Larsen et al., 2021b). Such additional treatment steps were beyond the scope of this study. There are also uncertainties in the results due to the innovative nature of the technologies studied. Additional challenges and opportunities will likely emerge as the technologies go to scale, but were outside the scope of this study. However, results from this study can assist in future optimisation of these urine concentration technologies.

4. Results and discussion

According to the performance matrix (Table 2), all three urine concentration technologies investigated performed very well for several criteria. The most significant improvement from implementing urine concentration was in nutrient recovery, particularly for N (Table 2). At the scale studied (e.g. technology serving only a proportion of the population), the future scenarios did not perform differently from the baseline with regard to eutrophying emissions or P recovery. However, energy demand, space efficiency, cost and worker health were more varied. These results in particular can provide guidance for scaling-up and optimising these emerging technologies.

4.1. Environment

4.1.1. Eutrophying emissions

Use of the urine concentration technologies gave a negligible reduction in eutrophying emissions to the recipient at the scale assessed. Concentration of nutrients in WWTP effluent remained at 12 mg N/L and 0.3 mg P/L (Table 2). The proportion of wastewater from Sege Park was too small (0.25% of all wastewater treated at the WWTP) to achieve a marked reduction in nutrient loading.

However, urine diversion had the potential to remove 75% of N and over 40% of P in wastewater from Sege Park before it reached the WWTP, indicating that implementation of urine concentration technologies is likely to result in lower concentrations of eutrophying emissions from WWTPs (for other impacts of scaling-up, see section 4.5.1). The 75% N removal efficiency would also meet the requirement for >70% annual nitrogen removal efficiency (Swedish EPA, 2016), even before the nitrogen removal treatment step at the WWTP.

4.1.2. Nutrient recovery

Nutrient recovery values presented in Table 2 are percentage recovery from total household wastewater produced in Sege Park. Compared with the total population of Malmö the percentage recycling is almost negligible, as the population in Sege Park is <1% of the total population. Phosphorus recovery was similar for the conventional system (94%) and the different technologies. For alkaline dehydration and nitrification-distillation, P recovery from the urine was 100% (Etter et al., 2015; Senecal, 2020). Assumed P recovery from struvite precipitation was 93% (Etter et al., 2015). For faeces and greywater treated at the WWTP, percentage P recovery was assumed to be the same as for the conventional system. The resulting P recovery from total household wastewater produced in Sege Park was 97% for alkaline dehydration and nitrification-distillation, and 96% for the ion-exchange system. In contrast, there was a large difference in N recovery between the conventional system (15%) and the urine concentration technologies. Nitrogen recovery from urine was assumed to be 98% for alkaline dehydration (Simha et al., 2020b) and 99% for nitrification-distillation and ion-exchange (Etter et al., 2015; Tarpeh et al., 2018b). Nitrogen recovery from faeces and greywater from the urine concentration systems was assumed to be the same as for the conventional system. Resulting N recovery for the urine concentration systems from total household wastewater produced in Sege Park was 78% for alkaline dehydration and 79% for the nitrification-distillation and ion-exchange systems.

Recovery efficiency values used in these calculations were the maximum recovery and applying the systems on a larger scale could result in lower N recovery due to N losses during transport, storage and urine concentration. For instance, the alkaline dehydration technology

Table 2
Performance matrix for the multi-criteria assessment, where CS is the conventional system, AD is alkaline dehydration, ND is nitrification-distillation and IE is ion-exchange with struvite precipitation. Performance was graded on a five-point scale with stoplight colour coding for visualisation (red = very poor performance, dark green = very good performance).

	Environment					Technical	Economic	Health
	Eutrophying emissions		Nutrient recovery		Energy demand	Space efficiency	Cost	Working environment
SDG	14		6, 9, 11, 12		9	11	6	3, 8
Indicator	Concentration of N in WWTP emissions (mg/L)	Concentration of P in WWTP emissions (mg/L)	%N recovery from total household wastewater produced in Sege Park	%P recovery from total household wastewater produced in Sege Park	% of annual apartment energy demand for urine concentration	Space required in residential area (m^2/household)	Additional annual cost per household (USD/yr)	Additional risks for workers (Number additional risk score ≥H)
Conventional system	12	0.3	15	94	4	-	-	-
Alkaline dehydration	12	0.3	78	97	31	0.6	320	1
Nitrification-distillation	12	0.3	79	97	5	1	840	3
Ion-exchange & Struvite precipitation	12	0.3	79	96	1	0.3	120	2

is designed to treat fresh urine and is therefore dependent on a functioning limebox. Malfunction would result in N losses (Simha et al., 2021a), so optimisation of MgO dosage and strategies to avoid losses on scaling-up are required. Nevertheless, nutrient recovery could drop to 65% and areas with urine concentration systems could still recover 50% of the N in household wastewater. Phosphorus recovery is not as sensitive to operating conditions as N recovery since, unlike N, P is not volatile.

4.1.3. Energy demand

Energy demand for the nitrification-distillation and ion-exchange systems was assumed to be 0.15 kWh/L_{urine} (Etter et al., 2015) and 0.021 kWh/L_{urine} (Antonini, 2013; Kavvada et al., 2017), respectively. Their energy consumption was graded very good, as they consumed ≤5% of annual energy demand of an apartment (Table 2). In comparison, the current WWTP consumes 4% of annual energy demand of an apartment. Energy demand was significantly higher for the alkaline dehydration system (1 kWh/L_{urine}) (J. Senecal, pers. comm. 2021), representing 31% of annual energy demand for an apartment.

Energy is a critical challenge for the alkaline dehydration system, with consumption needing to be reduced to around 25% of current levels to make it competitive with the other technologies. This system requires considerable amount of energy for heating and fans for moving air across the drying beds (Senecal, 2020). Simha et al. (2020c) suggest that the energy demand for drying urine in alkaline dehydration could be reduced from 5.8 to 21.7 to 0.22 kWh/L_{urine}, by recovering 85% of the energy required for evaporation using a heat pump. Energy consumption could also be reduced by using sensors to regulate the heating units to operate only as needed, and/or recycling heat in the exhaust air using a heat exchanger (Senecal, 2020). Excess heat from alkaline urine drying could potentially also be used for heating houses. In addition, other studies indicate that 90% urine diversion could result in 22% energy reduction at the WWTP (Badeti et al., 2021); a benefit not modelled in our study.

4.2. Technical

Regarding space efficiency, all systems required ≤1 m^2 per household and, when related to half the space requirement for parking per household (6.25 m^2), were graded as good to very good (Table 2). Based on treatment capacity, the alkaline dehydration and nitrification-distillation systems required 70 treatment units for the Sege Park area. Assumed treatment capacity of the ion-exchange and struvite precipitation system was over three times as large, and only 21 treatment units were required. Alkaline dehydration required 60% of the space per unit required by the two other urine concentration systems.

Note that none of the systems has been optimised and optimisation could affect the appearance and amount of treatment units, and their space requirement. The system with the largest footprint, nitrification-distillation, required 1 m^2 per household, primarily due to the nitrification rate (assumed 450 mg N/$L_{reactor}$/day). Additionally, the distiller has a 1-2 orders of magnitude higher capacity than nitrification columns (Etter and Udert, 2015). A higher nitrification rate and using more nitrification columns per distiller would result in fewer units and less space required. It is possible that these technologies, nitrification-distillation in particular, can reduce their space requirement with future optimisation. For decision makers, it is important to consider local conditions and space availability within a residential area when planning installation of urine concentration technologies, e.g. whether a building has available space in the basement or a separate building is required.

4.3. Economic

Capital costs were found to be highest for nitrification-distillation (4.7 million USD), while O&M costs were highest for alkaline dehydration (0.17 million USD/yr). The profit from sold fertiliser was the same for all systems (7000 USD/yr) (S3.2 in SM). Estimated O&M costs were based on electricity demand, key consumables (e.g. alkaline medium and ion-exchange resin) and service cost (considering the number of units). However, the service requirement may vary between urine concentration systems, affecting O&M costs. In Sege Park, the highest cost item for O&M of alkaline dehydration was servicing (110,000 USD/yr, similar to nitrification-distillation and more than double the cost of ion-exchange), followed by electricity costs (54,000 USD/yr, six times higher than nitrification-distillation). Additional costs per household were 320 USD/yr for alkaline dehydration, 840 USD/yr for nitrification-distillation and 120 USD/yr for the ion-exchange system (Table 2).

While costs were lower for ion-exchange than for the other urine technologies, it was still expensive compared with the conventional system. Difficulties in creating a product with high N concentration while using regenerant efficiently could affect the costs. According to Tarpeh et al. (2018b), further development is needed to enable reuse of the regenerant multiple times and/or add an additional step with reverse osmosis. An additional treatment step is also needed for the residual stream of N and P in stripped urine, and should be investigated. The most expensive component for the nitrification-distillation system was the distiller (annualised cost 4000 USD/yr/unit), which was 1-2 orders of magnitude greater than for any other component in all systems. However, the distiller has 1-2 orders of magnitude higher capacity than the nitrification columns (Etter and Udert, 2015) and could have been used for several more nitrification columns than assumed in this study, which would significantly reduce the costs. For the alkaline dehydration and nitrification-distillation systems, the treatment capacity (L urine treated per day) was less than one-third of the assumed capacity for the ion-exchange system and thus over three times as many treatment units were required. This may largely explain the price difference. It is important to bear in mind that the cost calculations were rough estimates and that the systems are at different levels of development. The costs for alkaline dehydration and ion-exchange were calculated based on cost estimates for all components, while those for nitrification-distillation are thoroughly described in the literature (Etter and Udert, 2015). Cost estimates for all technologies were based on production of one unit, but when applied on a larger scale the price per unit is likely to decrease.

4.4. Health

The hazardous events identified for the conventional system were mostly related to work in the WWTP. The risk level was defined as high for almost all hazardous events studied: exposure to aerosols, exposure to noise from machines, heavy lifting and poor working positions, exposure to chemicals, risk of falling, methane accumulation and falling into basin (Table S4.3 in SM). Potential hazardous events in the WWTP also applied for the urine concentration technologies. The conventional system was set as a baseline and was not assigned a performance grading. The alkaline dehydration technology had one additional hazardous event to the baseline, i.e. exposure to chemicals when changing the alkaline substrate, which received a high risk score in the semi-quantitative risk assessment (Table 3). It was thus graded as good. Nitrification-distillation had three additional hazardous events that received a high risk score: exposure to accumulated ammonia vapours in urine storage, exposure to accumulated nitrite and explosion caused by ammonia nitrite. The ion-exchange technology had two additional risks: exposure to accumulated ammonia vapours in urine storage and exposure to sulphuric acid during regeneration. Both nitrification-distillation and ion-exchange systems were graded as neither good nor bad.

The potential risks associated with ammonium nitrite vapours, which may pose a risk of explosion if the distiller runs dry due to thermal instability of ammonium nitrate, and nitrite accumulation following urine overloading of the system, were the main reasons for the lower

Table 3
Additional risks (above baseline) identified with the three urine concentration systems investigated. Risk score (R) determined based on a relationship between severity (S) and likelihood (L) and the risk level (RL).

Additional high (H) risks with urine concentration systems							
Hazard identification				Risk assessment			
#	Hazardous event	Hazard	Exposure route	L	S	R	RL
Alkaline dehydration system							
1	Changing alkaline substrate	Exposure to chemicals	Skin contact	3	8	24	H
Nitrification-distillation system							
2	Exposure to accumulated ammonia vapours if urine store needs to be opened	Toxic vapours	Inhalation	2	8	16	H
3	Exposure to accumulated nitrite	Toxic vapours, strong acid	Inhalation	2	8	16	H
4	Explosion caused by ammonium nitrite	Body injury, death	Explosion	1	16	16	H
Ion-exchange and struvite precipitation system							
5	Exposure to accumulated ammonia vapours if urine store needs to be opened	Toxic vapours	Inhalation	2	8	16	H
6	Exposure to sulphuric acid during regeneration	Strong acid	Inhalation, skin contact	3	8	24	H

performance in terms of risks. Process control to ensure that the distiller does not run dry and a stable throughflow of urine reduce the risk of system malfunction (Etter and Udert, 2015). Control of risk factors should be part of system optimisation for all urine concentration systems. We also recommend quantitative risk assessment, to provide additional in-depth analysis of the risks and possibly identify further risks.

4.5. *Impacts of scaling*

4.5.1. *Eutrophying N emissions*

Since the reduction in eutrophying emissions to recipient waters due to implementation of urine concentration technologies at Sege Park was negligible, the impact of wider application of urine diversion and concentration in Malmö was investigated. Wider implementation of urine diversion could reduce N emissions in effluent from the WWTP to ≤10 mg/L (good performance) without increasing the N removal rate of the WWTP. This could be achieved if 30% of all wastewater produced in Malmö came from urine diversion systems with separate treatment (Fig. 3a). In order to meet expected stricter limits on N emissions to surface water of 5–6 mg N/L (Åmand et al., 2016), over 60% urine diversion would be needed.

Other studies have examined the effect of urine diversion on N concentrations in effluent from a WWTP with biological N removal. Wilsenach and Van Loosdrecht (2003) found significantly reduced N emissions when up to 50% of the population served used urine diversion. Badeti et al. (2021) modelled significant N removal in effluent with up to 90% urine diversion, while more than 90% urine diversion reduced the capacity for biological nutrient removal.

Reduction of P emissions to surface water was not as significant as the reduction of N emissions as urine concentration only contribute with a marginal increase in P removal (from 94% to 96–97% removal). However, implementation of urine concentration technologies has the potential to reduce P emissions to surface water as well (Fig. 3b). For example, if 70% of the wastewater produced in Malmö came from urine diversion systems the P emissions to the surface water could potentially reduce from 0.3 mg/L to 0.2 mg/L (very good performance).

The quality of WWTP sludge may also be affected by expansion of urine concentration technology, because less P will end up in the sludge. This would affect the cadmium-phosphorus (Cd/P) ratio in the sludge. Sludge from Sjölunda currently has a Cd/P ratio of 25 mg Cd/kg P, although the recommendation is to reduce it to a maximum of 17 mg Cd/kg P before 2025 (Svenskt Vatten, 2020). This is because Cd may be toxic to organisms at moderate levels and Cd levels in Swedish soils are naturally high (Svenskt Vatten, 2019). In 2019, only 38% of REVAQ-certified WWTPs had a ratio <20 mg Cd/kg P (Svenskt Vatten, 2020). Increased urine diversion would require greater efforts to prevent sources of Cd, such as stormwater runoff and process chemicals, entering wastewater influent. If the sludge becomes unusable, this would affect P recycling and also carbon addition to the soil, which is beneficial for soil structure.

4.5.2. *Fertiliser demand*

For a urine concentration system with 99% N recovery rate, little more than 40% of urine in Malmö would need to be concentrated to meet the N fertiliser demand for all agricultural land in Malmö municipality (Fig. 4a). For a system with 70% N recovery rate, >50% of urine would need to be concentrated. To meet the P fertiliser demand in Malmö municipality, about 30% of urine produced in the city would need to be concentrated at 100% recovery rate, or 40% at 70% P recovery rate (Fig. 4b).

Uncertainties concerning urine N and P fertiliser efficiency could affect the results. Thus, field trials are needed on the fertiliser products from different urine concentration technologies, performed under the same conditions. It is also important to consider that Malmö is not self-sufficient, but imports much of the food consumed in the city. While a large proportion of the fertiliser demand in Malmö municipality can be met, this is not representative of the fertiliser demand for crops actually consumed in Malmö. According to Harder et al. (2021), nutrients accumulate around urban centres and may thereby meet a large part of the fertiliser demand in larger cities. However, this means that many rural areas farther from urban centres, which may produce large volumes of food, may be depleted of nutrients.

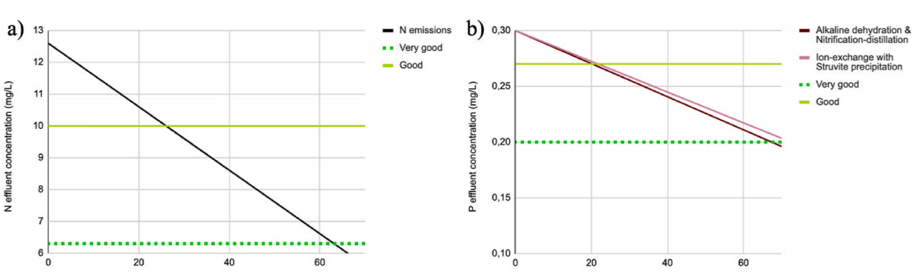

Fig. 3. a) Nitrogen concentration in WWTP effluent with different proportions of wastewater from urine concentration systems. The results are presented as a single line because the differences between the technologies are not visible at this scale. b) Phosphorus concentration in WWTP effluent with different proportions of wastewater from urine concentration systems for the alkaline dehydration and nitrification distillation systems (dark red) and the ion exchange and struvite precipitation system (purple). The lines Good (<10 mg/L for N and <0.27 mg/L for P) and Very good (<6.3 mg/L for N and <0.2 mg/L for P) define limit values for performance levels in the multi-criteria assessment.

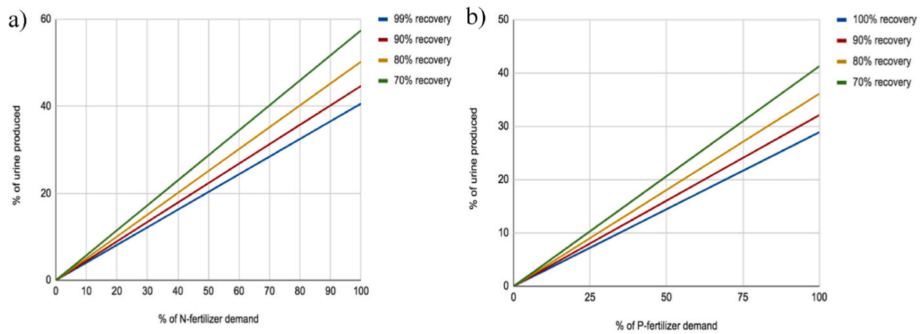

Fig. 4. Percentage urine produced in Malmö that would need to be treated with urine concentration technologies at four different a) N recovery rates and b) P recovery rates, in order to meet different percentages of N and P fertiliser demand in Malmö municipality.

5. Implications

This study showed that implementing urine diversion can contribute to achieving five environmental related SDGs (6, 9, 11, 12 and 14). This study is thus in-line with previous studies. Environmental benefits of urine diversion are consistently reported in life cycle assessments (Lam et al., 2020; Larsen et al., 2021a). Several other studies have shown that urine diversion can significantly reduce greenhouse gas emissions and energy consumption at the WWTP (Badeti et al., 2021; Hilton et al., 2021), thus it will also have positive climate impacts (SDG13).

This study found that issues related to costs, space efficiency (although generally good performance) and working conditions are areas for improvement in urine diversion systems (SDG 3,6,8). Other studies have also found that costs for urine diversion systems can be more expensive than conventional systems (Landry and Boyer, 2016). Kavvada et al. (2017) found that space rental was a key driver of costs of the system they modelled in San Francisco, thus improving space efficiency is a cost as well as an urban planning issue. However, Ishii & Boyer (2015), found costs for urine diversion systems to be equal to conventional systems on a monetary basis. Further system optimisation will likely lower costs and improve safe efficiency. This study has shown that attention should be focused on working conditions too.

Urine diversion systems still face technical bottlenecks such as energy use, costs and risks (shown in this study) and socio-institutional bottlenecks such as lack of norms, standards and installation know-how (Larsen et al., 2021a). Urine treatment systems are close to industrial optimisation (Larsen et al., 2021b), but full technical optimisation will require large-scale implementation, together with efforts to increase acceptance and legitimation of the concept. Dissemination and verification of our preliminary results will be key to further development.

The results in this study can guide development and implementation strategies for urine concentration technologies at scale. The main conclusions were:

- Urine diversion and concentration technologies can significantly reduce nitrogen emissions to surface waters and local demand for external fertiliser inputs, even at implementation levels of 20–30% of a city.
- All systems should include strategies for reducing costs, as well as potential N losses when scaled, in order to maximise N recovery and in turn fertiliser value.
- Alkaline urine dehydration requires optimisation of energy demand, to reduce the energy consumption and costs.
- Nitrification-distillation requires optimisation of the nitrification rate and matching it to the distillation capacity, which can reduce space requirements and costs. Attention should also be given to risk factors for workers.
- Ion-exchange with struvite precipitation can be improved with respect to costs and risk for workers, in particular regarding use of sulphuric acid in regeneration of the ion-exchanger.

This study demonstrated positive potential impacts of urine concentration systems, but occasionally at the expense of e.g. increased costs and/or energy demand. More in-depth analysis is required of social acceptance, costs and risks associated with the treatment process and use of the fertiliser product. The results in this study could be used with expanded system boundaries to better understand trade-offs and incentives for implementing urine concentration, supporting transition to circular resource use.

CRediT authorship contribution statement

Matilda Gunnarsson: Conceptualization, Methodology, Investigation, Formal analysis, Writing – original draft. **Cecilia Lalander:** Supervision, Writing – review & editing. **Jennifer R. McConville:** Conceptualization, Methodology, Supervision, Writing – review & editing.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2022.135194.

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