Environmental Science
Water Research & Technology

PAPER

View Article Online
View Journal | View Issue

Cite this: *Environ. Sci.: Water Res. Technol.*, 2019, **5**, 1599

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

Brenda Vidal, *[a] Annelie Hedström, [a] Sylvie Barraud, [b] Erik Kärrman [c] and Inga Herrmann [a]

Received 20th May 2019, Accepted 26th July 2019

DOI: 10.1039/c9ew00425d

rsc.li/es-water

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.

Water impact

Integrating the main sustainability dimensions and accounting for tradeoffs between criteria increase the complexity and challenge decision processes when choosing sanitation systems. This study provides researchers and decision-makers with a comprehensive approach to reflect upon sustainability criteria, allocation of weights and scenarios of action that may be encountered when seeking sustainable solutions in the water sector.

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.[1] Estimations from the Swedish EPA suggest that approximately 20% of the facilities do not comply with national standards,[2] and similar figures (an estimated 10% to 20% of existing systems) have been reported in the United States.[3]

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.[4] This is done to assure integrity and multidimensionality. However, the definition and estimation of indicators defining sustainability are challenging,[5] 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

[a] Department of Civil, Environmental and Natural Resources Engineering, Luleå University of Technology, SE 971 87 Luleå, Sweden. E-mail: brenda.vidal@ltu.se
[b] Department of Civil Engineering and Urban Planning, National Institute of Applied Sciences of Lyon, 34 Avenue des Arts Bâtiment J.-C.-A. Coulomb, 69621 Villeurbanne CEDEX, France
[c] Division of the Built Environment, Research Institutes of Sweden, 111 21 Stockholm, Sweden

† Electronic supplementary information (ESI) available. See DOI: 10.1039/c9ew00425d

decision-making processes, particularly in order to deal consistently with large amounts of complex information.[6] All MCA approaches require an exercise of judgment and nearly all decisions imply a weighting system of some sort.[6] 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)[7] but often present results from specific case studies *e.g.* Lennartsson *et al.* (2009)[8] or include a limited number of alternatives *e.g.* Bradley *et al.* (2002)[9] only evaluates two types of OSS. Indicators of different attributes such as those derived from life cycle assessments, cost analysis, mass balances or qualitative 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.[4,10,11] Furthermore, life cycle approaches,[12–15] environmental systems analyses[16,17] and sustainability assessments[18,19] have been applied. Due to the large number of small-scale and on-site sanitation technologies that currently exist,[20] and despite the criteria and indicators already suggested in the scientific literature,[8,10,11] 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,[21] and assessing different scenarios. The methodological approach presented here contributes to the understanding of indicators selection and trade-offs in their performance, and intends to provide general considerations when evaluating how sustainable a system is. The results of this study are therefore of interest to analysts, but also to legislators, regulating authorities, producers and operators of on-site sanitation systems who will ultimately be involved in decision-making processes.

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).[22]

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[4,8,10,15,18] 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[1,8,23,24] 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-sorption capacity of the material is exhausted.[25,26] The retention capacity of the soil is finite and varies with soil mineralogy, organic content, pH, redox potential and cation exchange capacity,[27] 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.[28] 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 1 Summary of the sustainability criteria and indicators. VH = very high; H = high; M = medium; L = low; VL = very low

Criteria category	Indicator	Unit	Qualitative/quantitative	Aim	Evaluation method
Environmental	Nutrient removal (tot-N and P)	%	Quantitative	High removal	Mass balance calculations
	Potential for nutrient recycling (N, P)	%	Quantitative	High potential	Mass balance calculations
	Global warming potential (GWP)	Kg CO_2-eq. per year	Quantitative	Low potential	Life cycle analysis
	Cumulative energy demand (CED)	MJ per year	Quantitative	Low demand	Life cycle analysis
	Energy recovery	H-M-L	Qualitative	High recovery	Qualitative evaluation
Economic	Capital cost	€ per year	Quantitative	Low cost	Cost analysis
	Operation & maintenance cost	€ per year	Quantitative	Low cost	Cost analysis
Socio-cultural	Social acceptance	VH-H-M-L-VL	Qualitative	High acceptance	Qualitative evaluation
Technical	Robustness	H-M-L	Qualitative	High robustness	Qualitative evaluation
Health	Risk of pathogen discharge	VH-H-M-L-VL	Qualitative	Low risk	Qualitative evaluation

based systems and package plants, the tot-N removal depends on nitrification and (limited) denitrification processes that occur on-site.[27] 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[29]) 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,[30] 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 CO_2-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[31] 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).[32]

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.[16,33] The present value of the components and services were taken from the main distributors' websites (*e.g.* Avloppscenter[34]) 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,[1] users are familiar with them and they are considered convenient due to their simplicity.[8] 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[8,35] 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;[23] 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.[36] 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.[23]

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 ESI.†

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.[37] 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®[38]) and low-flush urine-diverting toilets with 0.3 for small flush and 2.5 L for big flush (EcoFlush®[39]) 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 2 Summary of sanitation alternatives

Type of system	No.	Description
Soil-based	A1	Wastewater collected in a three-chamber septic tank and pumped to a sub-surface sand filter with distribution pipes. Effluent collected at the bottom of the filter with drainage pipes
	A2	As A1, but the wastewater continues infiltrating and percolating through the underlying soil instead of being collected under the sand filter
	A3	As A1, but with additional polishing step for phosphorus (P) removal (alkaline P-filter) using the filter media Polonite®
	A4	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
	A5	As A4, but the wastewater continues infiltrating and percolating through the underlying soil as A2
Source separation	S1	Ultra-low-vacuum toilet for blackwater (BW) collection, BW stored in a holding tank and transported to a central outside treatment facility (using urea (1%) for hygienization) once a year. Separate collection of graywater (GW) in a septic tank and subsequent sand filter
	S2	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)
Package plants	P1	A single unit buried underground with three sedimentation tanks and two bioreactors with aeration. Effluent filters through alkaline P-filter with Polonite® material
	P2	A single unit buried underground operating in a 2-phase semi-continuous regime with activated sludge process, with equalization tank, aeration tank and chemical dosing

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® 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 ($a\ S\ b$) (i) "*if there were enough arguments to decide that a 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*" [measured by discordance indices $D_{\text{i}}\ (a,b)$].[49]

To calculate the concordance index $C(a,b)$, the alternatives were evaluated against each indicator by pairwise multiplying the partial concordance values $c_{\text{i}}\ (a,b)$ 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 $c_{\text{i}}\ (a,b)$. 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 $D_{\text{i}}\ (a,b)$, 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 $D_{\text{i}}\ (a,b) = 0$ 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

Table 3 Summary of relevant assumptions

	Septic tank volume (m^3)	Phosphorus removal mainly by	Electricity use[a] (kW h y^{-1})	Biogas production	Sand/gravel used for construction (m^3)
A1	2.2	Sand	7.5 (pump)	Yes	39
A2	2.2	Sand	7.5 (pump)	Yes	16
A3	2.2	P-filter	7.5 (pump)	Yes	39
A4	4	Coagulant sedimentation	7.5 (pump) +1 (dosing equipment)	Yes	39
A5	4	Coagulant sedimentation	7.5 (pump) +1 (dosing equipment)	Yes	16
S1	1.2 (GW) and 6 (BW)	BW separation	7.5 (pump)	Minor (only sludge from GW)	39
S2	2.2 (GW + feces) and 3 (urine)	Urine diversion	7.5 (pump)	Minor (only sludge from GW)	39
P1	≈2.5	P-filter	450 (whole plant)	Yes	0
P2	2.5	Coagulant sedimentation	550 (whole plant)	Yes	Negligible

[a] Electricity use refers to the consumption during operation of the facilities.

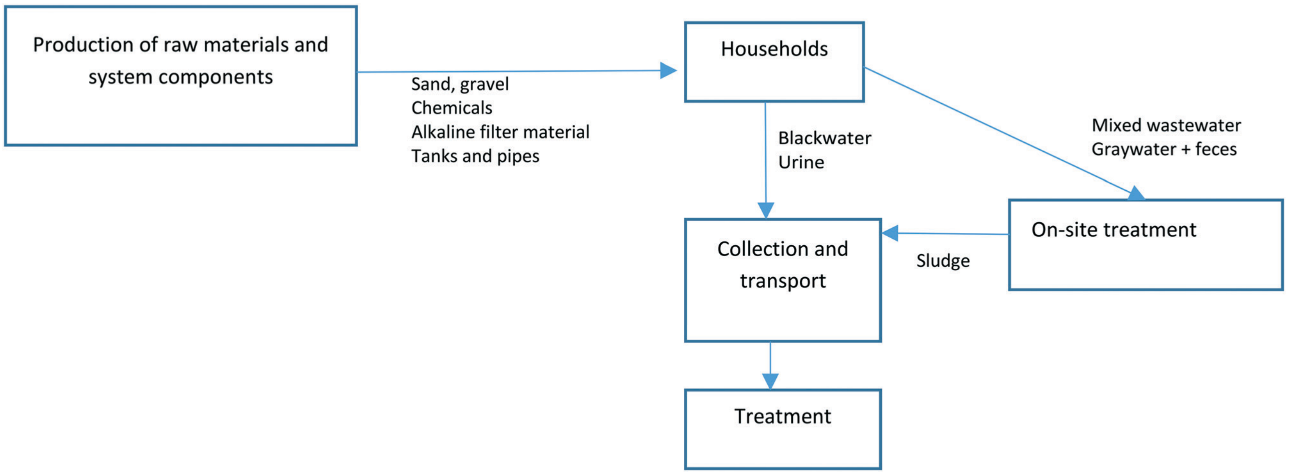

Fig. 1 General overview of the system boundaries for the indicators GWP and CED.

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.[50] Further detailed descriptions of the methodology can be found in the scientific literature.[49,51,52]

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.[45] To weigh the importance of the selected indicators, the panel method was applied.[22] 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).

$$W_i = \frac{1}{n}\sum_{i=1}^{n}\frac{w_i^* \times 100}{W^*} \qquad (1)$$

where:
W_i = weight of indicator i,
w_i^* = 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[53] 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,[53] 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 ESI.†

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,[28,36] 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 ton[62] and the cost reported for urea sanitization of BW,[63] 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.[32] 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 removal[26,54] but are instead intended to degrade carbonaceous material.

Table 4 Performance matrix of the different alternatives with respect to each indicator

Criteria	Indicators	Unit	Sand filter A1	Drain field A2	Sand filter + P-filter A3	Chem. P removal + sand filter A4	Chem. P removal + drain field A5	GW, BW S1	Urine diversion S2	Package plant P1	Package plant P2
Environmental	Tot-N removal	% tot-N	30	35	30	30	35	95	85	40	40
	P removal	% P	40	40	90	90	90	90	80	90	90
	Potential for N recycling	% N	2	2	2	2	2	90	70	2	2
	Potential for P recycling	% P	5	5	8	29	29	81	53	9	29
	Global warming potential	Kg CO_2-eq. per hh[a] per year	54	35	88	67	53	80	73	104	95
	Cumulative energy demand	MJ per hh[a] per year	2806	2403	3393	3985	3663	4471	3952	7627	8562
	Energy recovery	H-M-L[b]	Medium	Medium	Medium	High	High	Low	Medium	Medium	High
Economic	Capital cost	€ per year	468	465	761	564	605	686	581	627	650
	Operat. & maint. cost	€ per year	166	166	377	380	380	287	253	536	481
Socio-cultural	Social acceptance	VH-H-M-L-VL[b]	Very high	Very high	High	Medium	Medium	High	Low	High	High
Technical	Robustness	H-M-L[b]	High	High	Medium	Medium	Medium	Medium	Low	Low	Low
Health	Risk of pathogens discharge	VH-H-M-L-VL[b]	Medium	High	Low	Low	Medium	Very low	Medium	Low	Low

[a] hh = household. [b] VH = very high; H = high; M = medium; L = low; VL = very low.

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 CO_2 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 CO_2 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 CO_2 eq. kg^{-1} of sludge considering anaerobic digestion and dewatering processes[64] 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)[38] than the urine diversion toilet (€456)[39] 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 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 management 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 convenience and low complexity as reported in the literature.[8] 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 personnel 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.[66,67]

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 the filter material.[27,68] The soil-based alternatives with polishing steps (A3–A5) had medium robustness because of the increased number of risks when adding extra components and consumables, *e.g.* P-filter, P-removal chemicals.[66] The GW and BW separation (S1) also had medium robustness 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 alternative 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 pipe[69] or ventilation 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 systems[27] 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 barriers 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.[71]

Table 5 Indifference and preference thresholds as defined for the analyzed indicators and approach of definition of threshold values

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

[a] (Eurostat, 2016).[72]

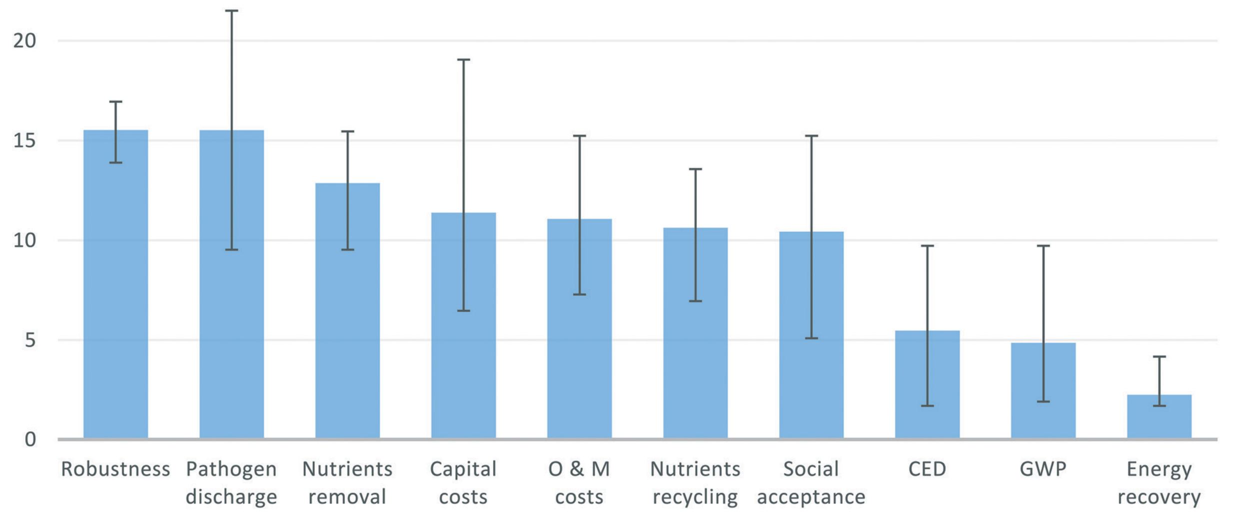

Fig. 2 Normalized weights for each indicator based on weights given by the six members of the reference group. The error bars indicate the maximum and minimum normalized weights given for each indicator. Sum of weights across indicators = 100.

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

Table 6 Ranking of alternatives for the different scenarios. *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

	Description	Ranking of alternatives
Scenario 0	With original weights from reference group	S1 > A1, A4 > A3 > A2 > A5 > S2 > P2 > P1
Scenario 1	Lowest importance to nutrients-related indicators (*e.g.* northern Sweden)	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

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 N_2O 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) outranked 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 recycle 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 outranked 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

Modified parameter	Ranking	Comments
None	S1 > A1, A4 > A3 > A2 > A5 > S2 > P2 > P1	Baseline (scenario 0) ranking
1. Decrease in the lifetime of Polonite® from 3 to 2 years	S1 > A1, A4 > A3 > A2 > A5 > S2, P2 > P1	Increase in O&M cost, GWP and CED in A3 (28%, 16% and 7% respectively) and in P1 (19%, 7% and 3% respectively)
2. Include the cost for BW and urine treatment	S1 > A1, A4 > A3 > A2 > A5 > S2 > P2 > P1	Increase O&M cost in S1 and S2, when assuming a municipal fee of €50 and €20 hh^{-1} y^{-1} for BW and urine management respectively
3. Change the weight of the importance of P (100%) and N (0%) removal	S1 > A1, A4 > A5 > A2 > A3 > S2 > P2 > P1	There are no requirements for N removal for wastewater systems for less than 10 000 PE
4. Decrease in the reuse of BW to 0%	S1 > A1 > A4 > A3 > A2 > A5 > S2 > P1 > P2	BW is collected and treated in a WWTP together with sludge instead of urea; lower potential to recycle nutrients and lower GWP
5. Increase robustness of S2, P1 and P2 from low to medium	S1 > A1 > A4 > S2 > A2 > A3, P2, P1 > A5	Considering they are managed properly and less failures occur
6. Change indif. and pref. thresholds for indicator *Robustness*	S1 > A1 > A4 > S2 > A2 > A3 > P2, P1 > A5	To reflect the uncertainty in the evaluation of robustness, the indif. and pref. thresholds were increased to 1 and 2 respectively

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 m^3 instead of the conventional 2.2 m^3 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.[74] Wet composting of BW, on the other hand, would generally require some energy input for stirring and aeration[12] and to increase the temperature,[75] 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.[65] 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 m^3) 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,[23] 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[76] and generally do not comply with the existing environmental guidelines in terms of P reduction as already concluded in previous studies.[54,55] 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,[54] but also the lack of optimal design and construction in terms of hydraulics and clogging risks.[68] 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.[24,27]

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 unnoticeably, 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[13,66] and monitoring reports.[36] 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.[27] 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.[4] Additionally, package plants can be installed in areas where bedrock, soils or fluctuating groundwater tables limit the implementation of soil-based systems.[27] 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).[16] 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.[16]

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[30] 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.[16] Metals can also accumulate in Polonite® filters, although most of their content probably deposits in the septic tank[66] and the low concentrations accumulated in the filter material would likely not restrict their use as fertilizers.[77] 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 CO_2-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 analysists assume that *the criteria are not all independent.*[78] Often, the most suitable criteria for a judgement of alternatives are interconnected and present multiple interactions between them.[78,79] 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.[22] 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).[6] 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.[80] 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 multi-criteria decision making.[81]

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,[12,17,82] 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.[17,74]

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.[22] 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 when 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

This study was conducted with the financial support from the Swedish Research Council Formas (project number 942-2015-758) and the J Gustaf Richert Foundation. The members of the reference group are thankfully acknowledged for their contribution. The authors would like to thank Jonathan Alm from the Development Centre for Water (Utvecklingscentrum för vatten) in Norrtälje for the fruitful discussions, as well as all the distributors and constructors who provided useful information.

References

1 M. Olshammar, M. Ek, L. Rosenquist, H. Ejhed, A. Sidvall and S. Svanström, *På uppdrag av Havs-och vattenmyndigheten Uppdatering av kunskapsläget och statistik för små avloppsanläggningar [The state of knowledge and statistics for small sewage plants. Swedish Agency for Marine and Water Management]. Report 166*, Norrköping, (in Swedish), 2015.
2 EPA Swedish, *Wastewater treatment in Sweden- Swedish EPA Publications*, Stockholm, 2014.
3 J. Wilson, L. Boutilier, R. Jamieson, P. Havard and C. Lake, Effects of hydraulic loading rate and filter length on the performance of lateral flow sand filters for on-site wastewater treatment, *J. Hydrol. Eng.*, 2011, **16**, 639–649.
4 A. J. Balkema, H. A. Preisig, R. Otterpohl and F. J. Lambert, Indicators for the sustainability assessment of wastewater treatment systems, *Urban Water*, 2002, **4**, 153–161.
5 M. Finkbeiner, E. M. Schau, A. Lehmann and M. Traverso, Towards life cycle sustainability assessment, *Sustainability*, 2010, **2**, 3309–3322.
6 J. S. Dodgson, M. Spackman, A. Pearman and L. D. Phillips, *Multi-Criteria Analysis: A Manual*, London, 2009, vol. 11.
7 P. P. Kalbar, S. Karmakar and S. R. Asolekar, Selection of an appropriate wastewater treatment technology: A scenario-based multiple-attribute decision-making approach, *J. Environ. Manage.*, 2012, **113**, 158–169.
8 M. Lennartsson, E. Kvarnström, T. Lundberg, J. Buenfil and R. Sawyer, *Comparing sanitation systems using sustainability criteria - EcoSanRes Series, 2009–1*, Stockholm, 2009.
9 B. R. Bradley, G. T. Daigger, R. Rubin and G. Tchobanoglous, Evaluation of onsite wastewater treatment technologies using sustainable development criteria, *Clean Technol. Environ. Policy*, 2002, **4**, 87–99.
10 D. Hellström, U. Jeppsson and E. Kärrman, A framework for systems analysis of sustainable urban water management, *Environ. Impact Assess. Rev.*, 2000, **20**, 311–321.
11 G. Tjandraatmadja, A. K. Sharma, T. Grant and F. Pamminger, A decision support methodology for integrated urban water management in remote settlements, *Water Resour. Manag.*, 2013, **27**, 433–449.
12 P. Tidåker, C. Sjöberg and H. Jönsson, Local recycling of plant nutrients from small-scale wastewater systems to farmland—A Swedish scenario study, *Resour., Conserv. Recycl.*, 2007, **49**, 388–405.
13 S. Lehtoranta, R. Vilpas and T. Mattila, Comparison of carbon footprints and eutrophication impacts of rural on-site wastewater treatment plants in Finland, *J. Cleaner Prod.*, 2014, **65**, 439–446.
14 X. Xue, T. Hawkins, M. Schoen, J. Garland and N. Ashbolt, Comparing the life cycle energy consumption, global warming and eutrophication potentials of several water and waste service options, *Water*, 2016, **8**, 154.
15 M. E. Schoen, X. Xue, A. Wood, T. R. Hawkins, J. Garland and N. J. Ashbolt, Cost, energy, global warming, eutrophication and local human health impacts of community water and sanitation service options, *Water Res.*, 2017, **109**, 186–195.
16 P. Weiss, D. Eveborn, E. Kärrman and J. P. Gustafsson, Environmental systems analysis of four on-site wastewater treatment options, *Resour., Conserv. Recycl.*, 2008, **52**, 1153–1161.
17 J. Spångberg, P. Tidåker and H. Jönsson, Environmental impact of recycling nutrients in human excreta to agriculture compared with enhanced wastewater treatment, *Sci. Total Environ.*, 2014, **493**, 209–219.
18 M. Molinos-Senante, T. Gómez, M. Garrido-Baserba, R. Caballero and R. Sala-Garrido, Assessing the sustainability of small wastewater treatment systems: A composite indicator approach, *Sci. Total Environ.*, 2014, **497–498**, 607–617.
19 N. Diaz-Elsayed, X. Xu, M. Balaguer-Barbosa and Q. Zhang, An evaluation of the sustainability of onsite wastewater treatment systems for nutrient management, *Water Res.*, 2017, **121**, 186–196.

20 W. Green and G. Ho, Small scale sanitation technologies, *Water Sci. Technol.*, 2005, **51**, 29–38.

21 B. Roy, *ELECTRE III: Un algorithme de classement fondé sur une représentation floue des préférences en présence de critères multiples [A ranking algorithm based on a fuzzy representation of preferences in the presence of multiple criteria]. Cahiers du Centre d'Ét*, (in French), 1978.

22 H. V. Rowley, G. M. Peters, S. Lundie and S. J. Moore, Aggregating sustainability indicators: Beyond the weighted sum, *J. Environ. Manage.*, 2012, **111**, 24–33.

23 O. Palm, L. Malmén and H. Jönsson, *Robusta, uthålliga små avloppssytem. En kunskapssammanställning [Robust and durable small wastewater systems. A knowledge compilation]. Report 5224*, Stockholm, (in Swedish with English summary), 2002.

24 B. Vidal, A. Hedström and I. Herrmann, Phosphorus reduction in filters for on-site wastewater treatment, *J. Water Process Eng.*, 2018, **22**, 210–217.

25 C. Vohla, M. Kõiv, H. J. Bavor, F. Chazarenc and Ü. Mander, Filter materials for phosphorus removal from wastewater in treatment wetlands—A review, *Ecol. Eng.*, 2011, **37**, 70–89.

26 A. Sinclair, R. Jamieson, R. J. Gordon, A. Madani and W. Hart, Modeling phosphorus treatment capacities of on-site wastewater lateral flow sand filters, *J. Environ. Eng.*, 2014, **140**(2), 04013002.

27 USEPA, *Onsite Wastewater Treatment Systems Manual (EPA/625/R-00/008)*, 2002, 367.

28 D. Hellström, L. Jonsson and M. Sjöström, *Bra Små Avlopp Slutrapport - Utvärdering av 15 enskilda avloppsanläggningar [Good small sewage final report - Evaluation of 15 sanitation sewage facilities]*, Stockholm, (in Swedish), 2003.

29 B. Vinnerås and H. Jönsson, The performance and potential of faecal separation and urine diversion to recycle plant nutrients in household wastewater, *Bioresour. Technol.*, 2002, **84**, 275–282.

30 V. Cucarella, T. Zaleski, R. Mazurek and G. Renman, Fertilizer potential of calcium-rich substrates used for phosphorus removal from wastewater, *Polish J. Environ. Stud.*, 2007, **16**, 817–822.

31 *ISO 14040, 2006. Environmental Management - Life Cycle Assessment - Principles and Framework: International Standard 14040*, International Standards Organisation, Geneva, 2006.

32 H. Jönsson, A. Baky, U. Jeppsson, D. Hellström and E. Kärrman, *Composition of Urine, Feaces, Greywater and Biowaste for Utilisation in the URWARE Model. Report 2005:6*, Urban Water, Chalmers University of Technology, Gothenburg, Sweden, 2005, vol. 6.

33 P. Ridderstolpe, *Markbaserad rening. En förstudie för bedömning av kunskapsläge och utvecklingsbehov [Soil-based treatment. A preliminary study for assessing the state of knowledge and development needs]. Report 2009:77*, Västra Götalands county, (in Swedish), 2009.

34 *Avloppscenter*, https://www.avloppscenter.se/en/.

35 D. Hellström and L. Jonsson, Evaluation of small on-site wastewater treatment systems, *Management of Environmental Quality: An International Journal*, 2006, **17**, 728–739.

36 M. Hübinette, *Tillsyn på minireningsverk inklusive mätning av funktion [Inspection of package plants including measurement of functioning]. Report 2009:07*, Västra Götalands county, (in Swedish), 2009.

37 EPA Swedish, *Förord till faktablad om enskilda avlopp: Tilloppsledningar slamavskiljare fördelningsbrunnar [Preface to the fact sheet on on-site wastewater treatment: distribution pipes, septic tanks, distribution wells]*, (in Swedish), 2003.

38 Wostman, *Manual EcoVac vacuum toilet system*, 2017, 4–17.

39 Wostman, *Manual EcoFlush low flush toilet system*, 2017, 3–9.

40 WHO, *WHO Guidelines for the safe use of wastewater, excreta and greywater - Volume 4*, WHO, 2006, 182.

41 A. Nordin, Ammonia Sanitisation of Human Excreta Treatment Technology for Production of Fertiliser, *Doctoral Thesis*, Swedish University of Agricultural Sciences, 2010.

42 H. Kjerstadius, A. Bernstad Saraiva and J. Spångberg, *Can source separation increase sustainability of sanitation management? Report Nr.5*, VA-teknik Södra, Lund, 2016.

43 O. Palm, E. Elmefors, P. Moraeus, P. Nilsson, L. Persson, P. Ridderstolpe and D. Eveborn, *Läget inom markbaserad avlopsvattenrening - Samlad kunskap king reningstekniker för små och enskilda avlopp [State of the art in soil-based wastewater treatment - Review of treatment techniques for small-scale wastewater treatment], Swedish EPA, Report 6484, 1–35*, Stockholm, (in Swedish with English summary), 2012.

44 J.-L. Bertrand-Krajewski, S. Barraud and J.-P. Bardin, Uncertainties, performance indicators and decision aid applied to stormwater facilities, *Urban Water*, 2002, **4**, 163–179.

45 J. Figueira, V. Mousseau and B. Roy, *Multiple Criteria Decision Analysis: State of the Art Surveys*, Springer, Boston, 2005.

46 M. Velasquez and P. T. Hester, An analysis of multi - criteria decision making methods, *Int. J. Oper. Res.*, 2013, **10**, 56–66.

47 F. Tscheikner-Gratl, P. Egger, W. Rauch and M. Kleidorfer, Comparison of multi-criteria decision support methods for integrated rehabilitation prioritization, *Water*, 2017, **9**, 68.

48 M. Cinelli, S. R. Coles and K. Kirwan, Analysis of the potentials of multi criteria decision analysis methods to conduct sustainability assessment, *Ecol. Indic.*, 2014, **46**, 138–148.

49 P. Vincke, *Multicriteria Decision - Aid*, John Wiley & Sons, Chichester, 1992.

50 J. Almeida Dias, J. R. Figueira and B. Roy, *The software ELECTRE III-IV. Methodology and user manual (Version 3.x)*, University Paris - Dauphine, LAMSADE, Paris, 2006.

51 B. Roy, *The Outranking Approach and the Foundations of Electre Methods*, Springer Berlin Heidelberg, Berlin, Heidelberg, 1990.

52 J. Figueira and B. Roy, Determining the weights of criteria in the ELECTRE type methods with a revised Simos' procedure, *Eur. J. Oper. Res.*, 2002, **139**, 317–326.

53 EPA Swedish, *ISSN 1403–8234, Naturvårdsverkets författningssamling. Naturvårdsverkets allmänna råd om små avloppsanordningar för hushållsspillvatten [General advice about small-scale household wastewater treatment facilities]*, Sweden, (in Swedish), 2006.

54 D. Eveborn, D. Kong and J. P. Gustafsson, Wastewater treatment by soil infiltration: Long-term phosphorus removal, *J. Contam. Hydrol.*, 2012, **140**, 24–33.

55 E. Kärrman, Å. Erlandsson, D. Hellström, B. Björlenius and P. Tidåker, Centralised or decentralised sanitation in Swedish summerhouse areas in transition to permanent living?, *Water Sci. Technol.*, 2007, **56**, 157–164.
56 J. Simos, *Évaluer l'impact sur l'environnement. Une approche originale par l'analyse multicritère et la négotiation [Environmental impact assessment. An original approach for multi-criteria analysis and negociation]*, Presses Polytechniques et Universitaires Romandes, Laussane, (in French), 1990.
57 AB Ecofiltration Nordic, *Polonite® - The sustainable treatment for phosphorus removal and recovery*.
58 C. Pratt, S. A. Parsons, A. Soares and B. D. Martin, Biologically and chemically mediated adsorption and precipitation of phosphorus from wastewater, *Curr. Opin. Biotechnol.*, 2012, **23**, 890–896.
59 European Council, *Council directive concerning urban waste water treatment 91/271/EEC*, 1991, 13.
60 European Court of Auditors, *Combating eutrophication in the Baltic Sea: further and more effective action needed. Special report number 3*, Luxembourg, 2016.
61 A. Richert, H. Jönsson, P. Tidåker and E. Petersens, *Avloppsfraktioner från enskilda kretslopp [Sewage fractions from individual wastewater systems]. Report 038*, Uppsala, (in Swedish with English summary), 2010.
62 A. Nordin, D. Göttert and B. Vinnerås, Decentralised black water treatment by combined auto-thermal aerobic digestion and ammonia – A pilot study optimising treatment capacity, *J. Environ. Manage.*, 2018, **207**, 313–318.
63 Y. Andersson, *Kretsloppsanpassning av små avlopp [Adaptation of small sewage systems to closing the loops]. Report 2011:33*, Västra Götalands county, (in Swedish), 2011.
64 A. Hospido, T. Moreira, M. Martín, M. Rigola and G. Feijoo, Environmental evaluation of different treatment processes for sludge from urban wastewater treatments: anaerobic digestion versus thermal processes, *Int. J. Life Cycle Assess.*, 2005, **10**, 336–345.
65 J. R. McConville, E. Kvarnström, H. Jönsson, E. Kärrman and M. Johansson, Source separation: Challenges & opportunities for transition in the Swedish wastewater sector, *Resour., Conserv. Recycl.*, 2017, **120**, 144–156.
66 D. Hellström and L. Jonsson, Evaluation of small wastewater treatment systems, *Water Sci. Technol.*, 2003, **48**, 61–68.
67 T. A. Larsen, A. C. Alder, R. I. L. Eggen, M. Maurer and J. Lienert, Source separation: will we see a paradigm shift in wastewater handling?, *Environ. Sci. Technol.*, 2009, **43**, 6121–6125.
68 L. Rolland, P. Molle, A. Liénard, F. Bouteldja and A. Grasmick, Influence of the physical and mechanical characteristics of sands on the hydraulic and biological behaviors of sand filters, *Desalination*, 2009, **248**, 998–1007.
69 K. M. Udert, T. A. Larsen and W. Gujer, Biologically induced precipitation in urine-collecting systems, *Water Sci. Technol.: Water Supply*, 2003, **3**, 71–78.
70 T. A. Stenström, Hygiene, a major challenge for source separation, Source Sep. Decentralization Wastewater Manag., in *Source Separation and Decentralization for Wastewater Management*, ed. T. A. Larsen, K. M. Udert and J. Lienert, IWA Publishing, London, 2013, pp. 151–161.
71 M. Rogers and M. Bruen, Choosing realistic values of indifference, preference and veto thresholds for use with environmental criteria within ELECTRE, *Eur. J. Oper. Res.*, 1998, **107**, 542–551.
72 Eurostat, *Energy, transport and environment indicators*, Publications Office of the European Union, Luxembourg, 2016.
73 J. Zheng, C. Egger and J. Lienert, A scenario-based MCDA framework for wastewater infrastructure planning under uncertainty, *J. Environ. Manage.*, 2016, **183**, 895–908.
74 H. Kjerstadius, S. Haghighatafshar and Å. Davidsson, Potential for nutrient recovery and biogas production from blackwater, food waste and greywater in urban source control systems, *Environ. Technol.*, 2015, **36**, 1707–1720.
75 L. Malmén, O. Palm and E. Norin, A collection and treatment system for organic waste and wastewater in a sensitive rural area, *Water Sci. Technol.*, 2003, **48**, 77–83.
76 I. Herrmann, B. Vidal and A. Hedström, Discharge of indicator bacteria from on-site wastewater treatment systems, *Desalin. Water Treat.*, 2017, **91**, 365–373.
77 A. Renman, G. Renman, J. P. Gustafsson and L. Hylander, Metal removal by bed filter materials used in domestic wastewater treatment, *J. Hazard. Mater.*, 2009, **166**, 734–739.
78 M. Molinos-Senante, T. Gómez, R. Caballero, F. Hernández-Sancho and R. Sala-Garrido, Assessment of wastewater treatment alternatives for small communities: An analytic network process approach, *Sci. Total Environ.*, 2015, **532**, 676–687.
79 X. Flores-Alsina, A. Gallego, G. Feijoo and I. Rodriguez-Roda, Multiple-objective evaluation of wastewater treatment plant control alternatives, *J. Environ. Manage.*, 2010, **91**, 1193–1201.
80 I. Gölcük and A. Baykasoɿlu, An analysis of DEMATEL approaches for criteria interaction handling within ANP, *Expert Syst. Appl.*, 2016, **46**, 346–366.
81 A. Fetanat and E. Khorasaninejad, A novel hybrid MCDM approach for offshore wind farm site selection: A case study of Iran, *Ocean Coast. Manag.*, 2015, **109**, 17–28.
82 M. E. Kvarnström, C. A. Morel and T. Krogstad, Plant-availability of phosphorus in filter substrates derived from small-scale wastewater treatment systems, *Ecol. Eng.*, 2004, **22**, 1–15.