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Evaluating source separation wastewater systems using traditional life cycle assessment and the planetary boundaries approach

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ABSTRACT

Life cycle assessment (LCA) is a commonly used method for assessing environmental impacts of systems, but cannot produce absolute values, i.e. a comparison with existing calculated values, which represents limits of what can be emitted into the environment. Therefore, absolute environmental sustainability assessments have been developed to assess impacts against the planetary boundaries (PBs) of the safe operating space for humanity. Since PB-LCAs are novel, it is useful to analyze both results from this method and conventional LCAs, something which has not been done before. This study applied both methods to two full-scale sanitation systems in the city of Helsingborg, Sweden. The current conventional system for handling wastewater with active sludge and food waste to biogas production was compared with the novel project H+ source separation system with three pipes (food waste, grey and black water) with increased resource recovery through anaerobic digestion, ammonia stripping, struvite precipitation and pelletization. The Planetary Boundaries LCA (PB-LCA) results showed that both systems exceeded eight of the assigned shares of PBs, including climate change and biogeochemical flows of nutrients. Traditional LCA (ReCiPe impact assessment) showed net savings for the H+ system in a few categories and considerable reductions in several impacts, e.g., global warming potential (GWP), stratospheric ozone depletion, terrestrial acidification, and water consumption. In PB-LCA the H+ system gave additional impacts in both assessments for a few categories, mostly due to high consumption of chemicals in the ammonium stripping process used for nutrient recovery. In conclusion, the combined assessments highlight hotspots for process optimization in the H+ system. From a methodological standpoint, PB-LCA still needs improvements to better reflect avoided burdens and results from traditional LCA should be fully transparent and analyzed carefully. The assessment methods complement each other and can be combined to better represent environmental performances of systems.

1. Introduction

Planetary boundaries to prevent anthropogenic pressures from destroying the Earth System have been established, by identifying the thresholds within which humanity can safely operate (Rockström et al., 2009). This safe operating space (SOS) indicates the critical limits of environmental impacts (Ryberg et al., 2021), beyond which catastrophic environmental changes can occur. The biogeochemical flow boundaries for nitrogen (N) and phosphorus (P) have already been exceeded (Steffen et al., 2015), and urgent action is needed to bring these back within the SOS.

Among the United Nations (UN) Sustainable Development Goals (SDGs) established to reduce environmental pressures and achieve global sustainability (United Nations, 2015), SDG 6 aims “to ensure availability and sustainable management of water and sanitation for all”. Access to sanitation and sanitary and healthy living conditions is considered a basic human right. Water is essential for human life and activities, so wastewater must be viewed as a valuable resource and treated to recover nutrients and water (Diaz-Elsayed et al., 2019). In fact, efficient use, reuse, and recycling of wastewater resources would

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help achieve multiple SDGs (Andersson et al., 2020).

Conventional sewer-based wastewater management appear to be unable to meet the SDG targets (Larsen et al., 2021). In particular, conventional systems are failing to provide sustainable options to close resource loops and reduce nutrient emissions to surface waters (SDG 14.1) (Andersson et al., 2020; Larsen et al., 2021). Water, energy, and nutrients (e.g., N. P) can be recovered from wastewater via different types of technologies. Source separation of wastewater, through urine diversion or composting toilets, has been shown to improve resource recovery and reduce environmental impacts (Besson et al., 2021; Landry and Boyer, 2016; Spångberg et al., 2014). However, these novel systems can bring new environmental impacts that occur throughout their entire life cycle i.e., from generation to final discharge (Dixon et al., 2003). Therefore, life cycle assessment (LCA) is needed to assess the environmental performance of emerging wastewater treatment systems.

Most previous LCA studies of wastewater treatment systems have compared different technological scenarios (Gallego et al., 2008; Garfí et al., 2017; Kalbar et al., 2016; Piao et al., 2016). Different configurations for source separation have also been studied, e.g., by Lima et al. (2022), Besson et al. (2021), Landry and Boyer (2016), and Spångberg et al. (2014). Studies comparing conventional systems with source separation have identified advantages of source-separating decentralized/small-scale systems, particularly lower impacts from avoided fertilizer production (Benetto et al., 2009; Lundin et al., 2000).

In system assessments, estimating environmental impacts in terms of the planetary boundaries (PBs) and the share of safe operating space (SoSOS) that each individual and/or societal function (service) occupies can help guide sustainable development. The SoSOS can be used as reference when assessing whether a human activity is absolutely sustainable or not (Bjørn et al., 2020b). Absolute environmental sustainability assessment (AESA) evaluates whether a product or service can be considered sustainable in an absolute sense, by calculating distance to known environmental limits (Bjørn et al., 2019, 2020a). In planetary boundaries-life cycle assessment (PB-LCA), the PBs are used as the environmental limits. The other main difference between PB-LCA and traditional LCA is that avoided emissions are not considered, as PB-LCA shows only emissions from a system, while by-products (recovered resources) are represented by larger SoSOS.

Only a few studies on wastewater treatment have been performed using PB-LCA. In one such study, Ryberg et al. (2021) used PB-LCA to evaluate whether a Danish utility company that supplies wastewater treatment and water could be considered absolutely sustainable. They found that although the company provides essential services, it exceeded the SoSOS in 10 of 18 impact categories assessed. However, that study did not apply traditional LCA and to the best of our knowledge, no previous study has analyzed a system using both LCA and PB-LCA; which could be valuable when investigating the sustainability of novel source separation wastewater systems, as the choice of impact assessment method affects the results.

This study evaluated the environmental impacts of wastewater treatment systems, for a novel treatment system in the Swedish city of Helsingborg by using both LCA and PB-LCA as impact assessment methods. In a new residential development close to the harbor, Helsingborg has implemented a novel source separation system (H+ system) consisting of pipes that separate wastewater into three streams: food waste from garbage disposal units, blackwater from vacuum toilets, and greywater from sinks, laundry rooms, and shower drains. These three streams are conveyed to a wastewater treatment plant, which treats them separately in order to recover clean water for reuse, substrate for biogas production, and plant nutrients (P and N), and to remove organic micropollutants (Kjerstadius et al., 2015, 2017).

Hence, the specific objectives of the present study were to: i) assess the environmental impacts of the H+ system, and ii) identify advantages and disadvantages of conventional LCA and PB-LCA. The results can inform decision-makers in their choice of technology and choice of assessment method.

2. Material and methods

The comprehensive environmental assessments performed in Helsingborg used the common practices of both forms of LCA, i.e., the ISO standards 14,040 and 14,044 for the traditional LCA and for the base of the PB-LCA in addition to the steps used by Ryberg et al. (2021). A summary of the methods and data used is provided below, while more details can be found in the Supplementary Material (SM).

Input data were gathered from real-life operations in Helsingborg and from the technology suppliers (for a full list, see SM).

2.1. Case study

Helsingborg, a small city on the south coast of Sweden, has a population of 112,496 inhabitants and occupies an area of 38.41 km^2. In 2013, an urban renewal project was launched in the harbor area to modernize the city and link the district with the city center. The project area, known as Oceanhamnen, was designed to accommodate source separation wastewater systems in new residential buildings containing apartments and offices for up to 2500 person-equivalents (p.e.). The buildings contain vacuum toilets for blackwater and kitchen grinders for food waste, and collection of these three streams, plus greywater, is by gravity. The wastewater streams from the Oceanhamnen district are treated separately in a local small-scale WWTP, with a total flow of 171 m^3/day and the main characteristics can be found in Table 1.

2.1.1. Scenarios

The new system implemented in Helsingborg was assessed in comparison with the conventional system currently implemented in the remainder of the municipality, which is described in Kjerstadius et al. (2017) and shown in Fig. 1. In the conventional system, households sort food waste into paper bags that are collected by truck and transported to a treatment plant, where the waste undergoes mechanical homogenization and digestion to biogas. The liquid digestate is used as fertilizer in agriculture, the biogas is upgraded to fuel, and the residues from the pre-treatment are incinerated for electricity and heat recovery. All domestic wastewater (BW, GW) is collected in a joint sewer network and directed to the local WWTP, where it undergoes primary and secondary treatment, with anaerobic digestion reactors and biogas generation. The biogas is upgraded to biofuel and used in city buses and the sludge is used as soil amendment and biofertilizer. Treated water goes through a sand filter for polishing, before discharge into the ocean.

In the H+ system, the source-separated streams are transported separately to the WWTP (Fig. 2). FW goes through pasteurization before reaching the anaerobic digester, generating biogas and sludge. The digestate goes through struvite precipitation and ammonium stripping. Sludge goes through dewatering and then pelletizing together with recovered struvite and ammonium sulfate, resulting in a pelleted biofertilizer that is used to replace mineral fertilizer in agriculture. The

Table 1
Characteristics of the wastewater streams considered.

Parameters	Food Waste[a]	Blackwater	Greywater
Flow	2.90 m^3/day	11.0 m^3/day	157 m^3/day
COD	24,344 mg/L	10,341 mg/L	480 mg/L
P-total	74 mg/L	199 mg/L	5 mg/L
N-total	427 mg/L	1781 mg/L	12 mg/L
TS	18,566 mg/L	10,413 mg/L	545 mg/L
TSS	–	6945 mg/L	135 mg/L
VS	15,789 mg/L	7663 mg/L	–
NH_4-N	–	1510 mg/L	12 mg/L

Note: Chemical Oxygen Demand (COD), Phosphorus (P), Nitrogen (N), Total Solids (TS), Total Suspended Solids (TSS), Volatile Solids (VS), Ammonium Nitrogen (NH4-N).

[a] Food waste stream was just considered as a liquid fraction in the new system. Source: Calculated based on Jönsson et al. (2005).

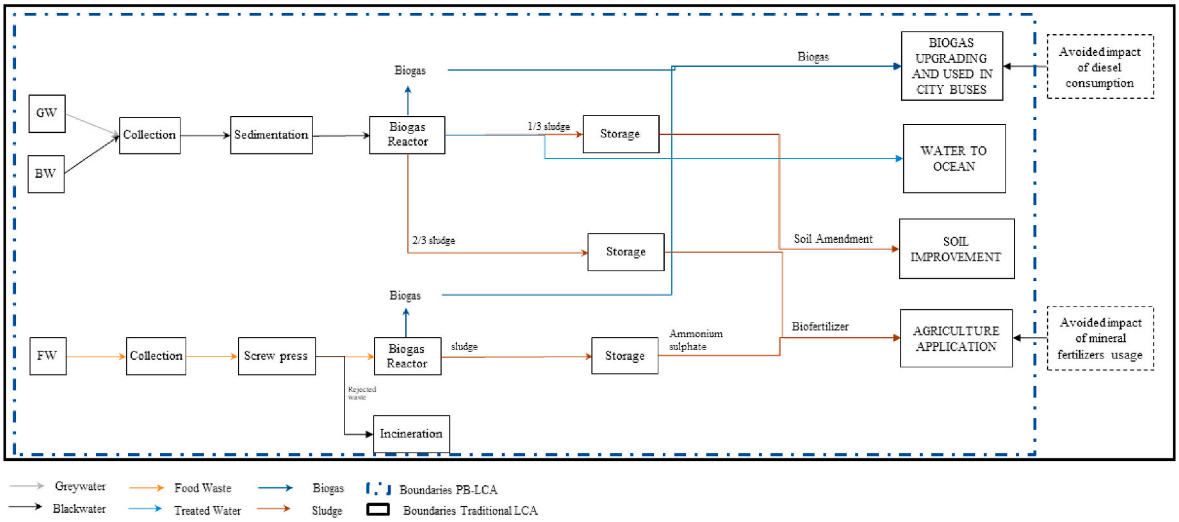

Fig. 1. Scenario conventional treatment. Note that avoided impacts (dashed box) are only considered in the traditional LCA.

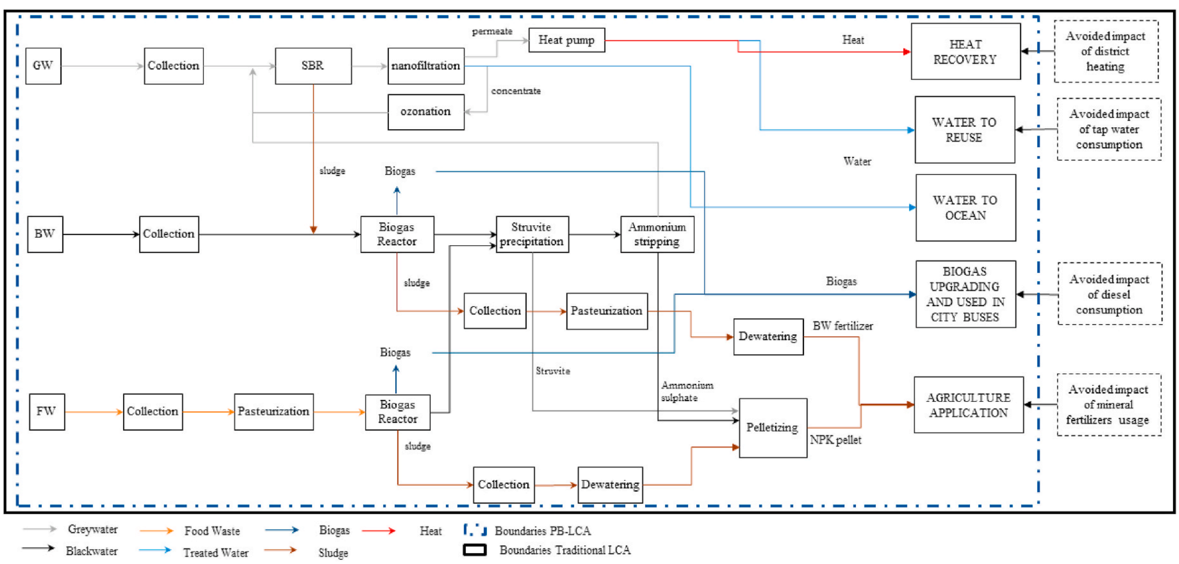

Fig. 2. Scenario H+. Note that avoided impacts (dashed box) are only considered in the traditional LCA.

biogas is also upgraded for use in city buses. The BW is treated the same way as FW, but the sludge undergoes pasteurization after the biogas reactor. Lastly, GW is treated in a sequencing batch reactor (SBR), followed by disinfection processes of nanofiltration and ozonation (addition of ozone to kill microorganisms). It is important to note that only the concentrate goes through ozonation and then it is sent back to the SBR, while the permeate goes through a heat pump where heat and water are recovered and reused. The sludge from the SBR process joins the BW stream before the anaerobic reactor.

Even though the treatments provide potable water quality, due to the regulations in Sweden and the lack of specific permits, both treated BW and GW are discharged 20% in the ocean and 80% is reused for irrigation.

2.2. Life cycle assessments

Both assessments were performed following the ISO standards (ISO, 2006a; ISO, 2006b) for LCAs.

2.2.1. System boundaries

The system boundaries are represented by Figs. 1 and 2 as described in the captions. The reason why the assessments have different boundaries lies in the fact that we performed two different and separate assessments with the same case study. The goal is to point out the weaknesses and strengths of both assessments in the way they are currently performed.

2.2.2. Goal and scope

The scope covered construction and operation of both systems, including background and foreground processes. It is important to stress that system expansion and substitution are not applied in PB-LCA, so these were only calculated and considered in the traditional LCA. Therefore, the environmental gains from recovered products in PB-LCA are included when calculating SoSOS, i.e., the more functions a system has (e.g., useful by-products), the larger its SoSOS.

The functional unit (FU) was defined as management of all domestic food waste and wastewater generated by one person equivalent per year, including collection, treatment, and disposal.

For estimation of wastewater generation, we assumed 2.4 p. e. for

320 apartments in Oceanhamnen and 0.5 p. e. for 1600 workers, totaling 1568 p. e. for the first two phases of construction. The inventory was calculated for a lifespan of 50 years.

2.2.3. Inventory

A detailed description of the inventory for both systems can be found in the SM, as well as the description of the systems, technology flows, and the Ecoinvent processes used.

2.2.4. Life cycle impact assessment

We used the ReCiPe® 2016 method (Huijbregts et al., 2017) for traditional LCA (Midpoint, World – Hierachist version), the PB-LCA method, and Simapro® for modeling. Characterization factors were taken from Ryberg et al. (2018) for PB-LCA and from the Simapro database for the ReCiPe® method, as were the normalization factors.

In the PB-LCA, the impact categories are defined as the nine PBs, but "introduction of novel entities" has not yet been quantified, so it was not considered in the assessment. In addition to the eight pre-defined categories, there are several regional considerations to be accounted for, as suggested by Bjørn et al. 2020a, b, since most local impacts are not global. The eight pre-defined PBs and their units are shown in Table 1 and the additional regional categories (n = 4) are further addressed in section 2.3.3.

As for the traditional LCA, we considered all impact categories provided by Simapro® for the ReCiPe® impact assessment, and these categories are also shown in Table 1, together with abbreviations and units.

According to Ryberg et al. (2018), there are some correlations between the two types of LCAs, but not all impact categories can be considered for both methods. To facilitate analysis of both methods and their pros and cons, we adapted our results according to the correlations identified in that study (Ryberg et al., 2018). Table 2 also aggregates the categories per this correlation.

2.2.5. Assigning SoSOS

The calculations on SoSOS were based on the defined safe operating space (SOS) for humanity for the different PBs, as shown in Table 3. Note that the SOS for humanity is defined by subtracting the share needed by nature from the boundaries.

As mentioned, when using a geographically resolved method such as LCA, regional environmental impacts need to be considered (Bjørn et al., 2020b). Therefore, based on the work of Hjalsted et al. (2021) and Ryberg et al. (2021), we calculated SoSOS for the impact categories presented in Table 3 and for the following four functions: wastewater treatment, water supply (from recovery), fuel for buses, and mineral fertilizer production. Using recommendations from Ryberg et al. (2020), we applied an egalitarian approach for allocation of the share to personal level and a utilitarian approach based on final consumption expenditure for upscaling to systems level. That means that we up-scaled the very small share obtained for each person to the systems level, and hence 1568 people were supplied by the functions provided by the systems assessed (wastewater treatment, water supply, fuel production, mineral fertilizer production). We then looked at direct and indirect spending on the functions and at the total production provided by the wastewater treatment system. All calculations for assigning SoSOS to wastewater treatment can be found in the SM (section 3).

2.2.6. Uncertainty

After modeling both assessments in Simapro®, a Monte Carlo uncertainty analysis was performed on both full systems. We used 10,000 runs, as is common practice, and a 95% confidence interval.

Table 2
Impact categories considered in the two LCA methods and their correlations.

Traditional -LCA categories	Units	PB-LCA categories	Unit
Global warming potential (GWP)	kg CO_2 eq	Climate change – Energy imbalance	Wm^{-2}
		Climate change as atmospheric CO_2 concentration	ppmCO_2/kg
		Ocean acidification	$\Omega_{aragonite}$ [mole]
Stratospheric ozone depletion (SOD)	kg CFC11 eq	Stratospheric ozone depletion	Dobson units
Ionizing radiation (IR)	kBq Co-60 eq		
Ozone formation - Human health (OFh)	kg NOx eq	Atmospheric aerosol loading	Aerosol optical depth [AOD; dimensionless]
Ozone formation - Terrestrial ecosystems (Oft)	kg NO_x eq		
Fine particulate matter formation (PMT)	kg PM2.5 eq		
Terrestrial acidification (TAD)	kg SO_2 eq		
Freshwater eutrophication (FEP)	kg P eq	Biogeochemical flows – P	Tg P to soil per year
Marine eutrophication (MEP)	kg N eq	Biogeochemical flows – N	Tg N fixated per year
Terrestrial ecotoxicity (TEC)	kg 1,4-DCB		
Freshwater ecotoxicity (FEC)	kg 1,4-DCB		
Marine ecotoxicity (MEC)	kg 1,4-DCB		
Human carcinogenic toxicity (HCT)	kg 1,4-DCB		
Human non-carcinogenic toxicity (HNCT)	kg 1,4-DCB		
Land use (LU)	m^2a crop eq	Land-system change	% of potential forest
Mineral resource scarcity (MRS)	kg Cu eq		
Fossil resource scarcity (FRS)	kg oil eq		
Water consumption (WC)	m^3	Freshwater use – Humid basins	Fraction of maximum annual water withdrawal [dimensionless]

Source: Adapted from Huijbregts et al. (2017), Ryberg et al. (2018, 2020).

3. Results and discussion

3.1. Complete results for PB-LCA and traditional LCA

3.1.1. PB-LCA

Table 4 shows the direct PB-LCA results obtained from modeling in Simapro®. Since normalization was not performed in the PB-LCA method, very small values were obtained. Values highlighted in green (and with *) in Table 3 represent the best performing results for each category. As can be seen, the H+ system had slightly higher impacts than the conventional wastewater system in most categories and two of the land-system categories (boreal and temperate) had zero impacts. Based on the PB-LCA methodology, the SoSOS assigned to both wastewater systems was calculated to be 7.59×10^{-10} (see SM). Although the H+ system had more functions considered in the SoSOS, due to very small values this made no difference to the end result, so both systems had similar SoSOS.

Table 3
Planetary boundaries and the remaining safe operating space (SOS) for human activities.

Planetary Boundary	Units	SOS for human activities
Climate change – Energy imbalance	Wm^{-2}	1
Climate change as atmospheric CO_2 concentration	ppmCO_2	72
Ocean acidification	$\Omega_{aragonite}$ [mole]	0.688
Stratospheric ozone depletion	Dobson units	15
Land-system change – Global forest	% of potential forest	25%
Land-system change – Boreal forest*	% of potential forest	15%
Land-system change – Tropic forest*	% of potential forest	15%
Land-system change – Temperate forest*	% of potential forest	50%
Freshwater use – Global	km^3 consumptive water per year	4000
Freshwater use – Semidry basins*	Fraction of maximum annual water withdrawal [dimensionless]	1
Freshwater use – Dry basins*	Fraction of maximum annual water withdrawal [dimensionless]	1
Freshwater use – Humid basins*	Fraction of maximum annual water withdrawal [dimensionless]	1
Atmospheric aerosol loading	Aerosol optical depth [AOD; dimensionless]	0.11
Biogeochemical flows – P	Tg P to soil per year	6.2
Biogeochemical flows – N	Tg N fixed per year	62

Source: Adapted from Ryberg et al. (2021). *represents regional adapted boundaries.

From the table it is important to note that even though Helsingborg is located in a temperate climate, in the land-system change categories, the one representing the tropics was the one with impacts. The reason for that is that for some processes used, such as electricity for biogas upgrading using wood chips, the energy source is coming from tropical forests, which consequently affects that category and not the temperate climate. Hence, this is one of the weakness of the assessment method since the inventory cannot be refined to reflect where the land-system change is occurring as Ecoinvent does not contain this information. In this way, most land-system change are, per default, attributed to tropical forest.

Estimated environmental sustainability ratio (ESR) is shown in Fig. 3 (logarithmic scale), where values greater than 1.00 (red shading) exceed the PBs and values lower than 1.00 are within the PBs (green shading). In PB-LCA, the specific contributions of six categories exceeded the PBs: climate change-energy imbalance, climate change-CO_2 concentration, ocean acidification, atmospheric aerosol loading, and biogeochemical flows (regional P and N).

As Fig. 3 demonstrates, neither sanitation system can be considered absolutely sustainable according to PB-LCA, as both systems exceeded eight of the 15 categories assessed. This is similar to findings in Ryberg et al. (2021), where water supply and wastewater treatment system exceeded 10 boundaries. In their case, reductions were needed in greenhouse gases (GHGs) that contribute to climate change and N and P emissions from sewers and WWTP, which may also apply in the present study. However, as discussed by Ryberg et al. (2021), absence of sanitation services is not an option and would most definitely have even worst effects, so actions are needed in relation to the hotspots and improving the existing systems.

Besides excess biogeochemical flows, both systems also exceeded the PBs in the climate change categories, ocean acidification, atmospheric aerosol loading, and freshwater use (dry and humid basins). Further, the H+ system had higher emissions in six impact categories, due to an added number of processes in the WWTP, but the emissions associated with nutrient flows were reduced.

Since the PBs are already exceeded for biogeochemical flows of N and P (Steffen et al., 2015), initiatives to improve recovery of these nutrients and avoid emissions to water are particularly important. Ultimately, the emissions trend needs to be reversed.

As shown in the specific contributions chart (Fig. 3), according to PB-LCA the main impact contributions were from operation of the WWTP in both systems, especially when adding the separate processes of ammonium stripping and pelletization in H+. This was due mainly to electricity consumption in the WWTP and to high use of chemicals (ammonium stripping) and transportation (pellets). However, the pelletization process only gave a significant burden in climate change and ocean acidification. Other significant emissions derived from construction of the WWTP (conventional system) due to concrete and steel use, and biogas upgrading in both systems, mainly from the CO_2 emitted from buses using the upgraded biogas and the propane used in upgrading.

Since no substitution is considered or evaluated in PB-LCA, this type of assessment and its representation focus on emissions from systems relative to the PBs. Such a focus is important, since burdens cannot be avoided once emissions have been released, and provides a perception of the direct impacts of the system assessed. However, the calculation of SoSOS could be improved, since the method considered in this work was based on current expenditure on wastewater functions by a country or region, and not necessarily on what should be spent, and the source separation system did not have a larger SoSOS despite delivering more functions. One way of fixing this issue would be by adding to the conventional system water supply, bus fuel and mineral fertilizer as inputs, to make these avoided burdens more visible, however it could be perceived as unnecessary since the system boundary does not provide this, and that the SoSOS should cover the issue.

3.1.2. Traditional LCA - ReCiPe® impact assessment

Some values obtained in the ReCiPe® method were negative, due to the avoided burdens, which indicates environmental savings in the impact category, whereas positive values represented environmental impacts (burdens). Table 5 shows the net results (balance between positive and negative impacts) for all impact categories considered in the traditional LCA method, and the respective units. The best results for each category are highlighted in green (and with *). Ozone formation-terrestrial ecosystems and freshwater eutrophication had very similar values for both wastewater systems. The H+ system performed best in 13 categories, while the conventional system performed best in five. However, both systems showed net savings in four categories.

For additional investigation, we have further treated the results from the Recipe assessment and presented them in detailed, differentiating burdens and savings and focusing on just the emissions, similarly to the PB-LCA. The information is presented in the SM (Fig. S13) where it shows the emissions obtained in the different impact categories, in characterized values, without savings.

As for the numbers presented in Table 4, in order to facilitate interpretation, the results were normalized and plotted in graphs (Fig. 4). This showed the net results for each impact category and the specific contributions for six very relevant categories: GWP, stratospheric ozone depletion, freshwater eutrophication, land use, marine eutrophication, and water consumption.

In the normalized results of the traditional LCA (represented by the ReCiPe® method), the H+ system performed better than the conventional system in six of the 18 impact categories assessed. The specific contributions chart (Fig. 4) showed that both systems gave high emissions in the different categories, but the environmental savings were significant and capable of bringing the net results down, e.g., for GWP, stratospheric ozone depletion (SOD), human carcinogenic toxicity, and water consumption (WC). Use of biofertilizer in agriculture and

Table 4
Complete results of planetary boundaries life cycle assessment (PB-LCA) for the conventional and H+ wastewater treatment systems in Helsingborg. Green highlights and * represent the best performing results.

Impact category	Unit/p.e.year	Conventional	H+
Climate change - Energy imbalance	Wm^{-2}	1.42E-11*	2.31E-11
Climate change - CO_2 concentration	ppm	7.00E-10*	1.33E-09
Ocean acidification	Omega Aragon	2.14E-12*	4.06E-12
Stratospheric ozone depletion	DU	2.97E-14*	1.76E-13
Freshwater use - Basin semidry	-	4.11E-13*	6.00E-13
Freshwater use - Basin dry	-	3.67E-12*	5.95E-12
Freshwater use - Basin humid	-	3.05E-12*	3.60E-12
Freshwater use – Global	km^3	2.97E-10*	5.32E-10
Atmospheric aerosol loading	AOD	2.20E-13*	5.48E-13
Biogeochemical flows - Regional P	Tg P	6.98E-08	4.79E-09*
Biogeochemical flows – N	Tg N	4.38E-07	1.48E-07*
Land-system change – Global	%	4.04E-13	9.40E-16*
Land-system change – Boreal	%	0.00	0.00
Land-system change – Tropic	%	1.14E-12*	2.49E-12
Land-system change – Temperate	%	0.00	0.00

avoidance of fuel from the biogas upgraded were mainly responsible for the environmental savings. Further, the district heating avoided in the conventional system gave considerable savings in FEP, due to avoided district heating usage based on the Swedish mix, as the treated water recovered in the H+ system had a significant impact on WC.

The ecotoxicity categories (freshwater, marine, human carcinogenic) showed the highest values in the normalization results, as is common in these assessments due to incomplete coverage of the substance flows in the normalization factors (Pizzol et al., 2017). Modeling fate and exposure, as well as unclear definition of the toxic effects, cause large uncertainties in these impact categories, which may be overestimated in calculation of emissions (Zhang et al., 2022). Therefore, in order to provide more complete results in terms of toxicity, more specific assessments (such as risk assessment) are needed as LCA studies do not usually explore hotspots based on these (Goedkoop et al., 2013; Zhang et al., 2022). The high toxicity emissions in our case, reflected the effect of WWTP construction in the conventional system (copper and steel) and of construction of collection system in buildings in the H+ system, plus ammonium stripping and, to a lesser extent, pelletization. In WWTP operation, most toxicity emissions were from the chemicals consumed, fuel consumption in transportation, and district heating.

As found in PB-LCA, the ammonium stripping process also gave the most significant individual burden in the six categories considered in traditional LCA. The most significant savings were in replacement of mineral fertilizer in agriculture, the upgraded biogas used as bus fuel, and the treated water used for irrigation in the H+ system.

The lowest normalized impacts were obtained in the WC category, in which the H+ system performed better according to traditional LCA (Fig. 4). For GWP, H+ more than halved the normalized impacts compared with the conventional system, mainly due to the avoided emissions from fertilizer application (N fertilizer and N_2O emissions), but it was not enough to give net savings. The main burdens in stratospheric ozone depletion (SOD) came from N_2O emissions from the WWTP in the conventional system, which were greatly reduced in the H+ system.

The eutrophication categories (FEP and MEP) reflect nutrient emissions (P and N, respectively) to water. The results showed that for FEP the environmental savings were higher in H+, but not enough to give net savings since the burden also increased, due mainly to the ammonium stripping and pelletization processes, and also transportation and electricity consumption. For MEP, the main emissions from the conventional system were from WWTP discharge, while total impact was significantly lower in the H+ system. Note that this is the opposite of the MEP results in PB-LCA (Fig. 3), represented by biogeochemical flows-regional P and biogeochemical flows-N, where the impacts decreased in the H+ system. This is explained by the ReCiPe® method considering irrigation emissions to the soil as affecting freshwater and by PB-LCA not taking any soil processes into account.

For LU in traditional LCA, the conventional system showed better environmental performance (net savings) due to less use of district heating, which was slightly higher in the H+ system with the added burden contribution from ammonium stripping. As also shown in

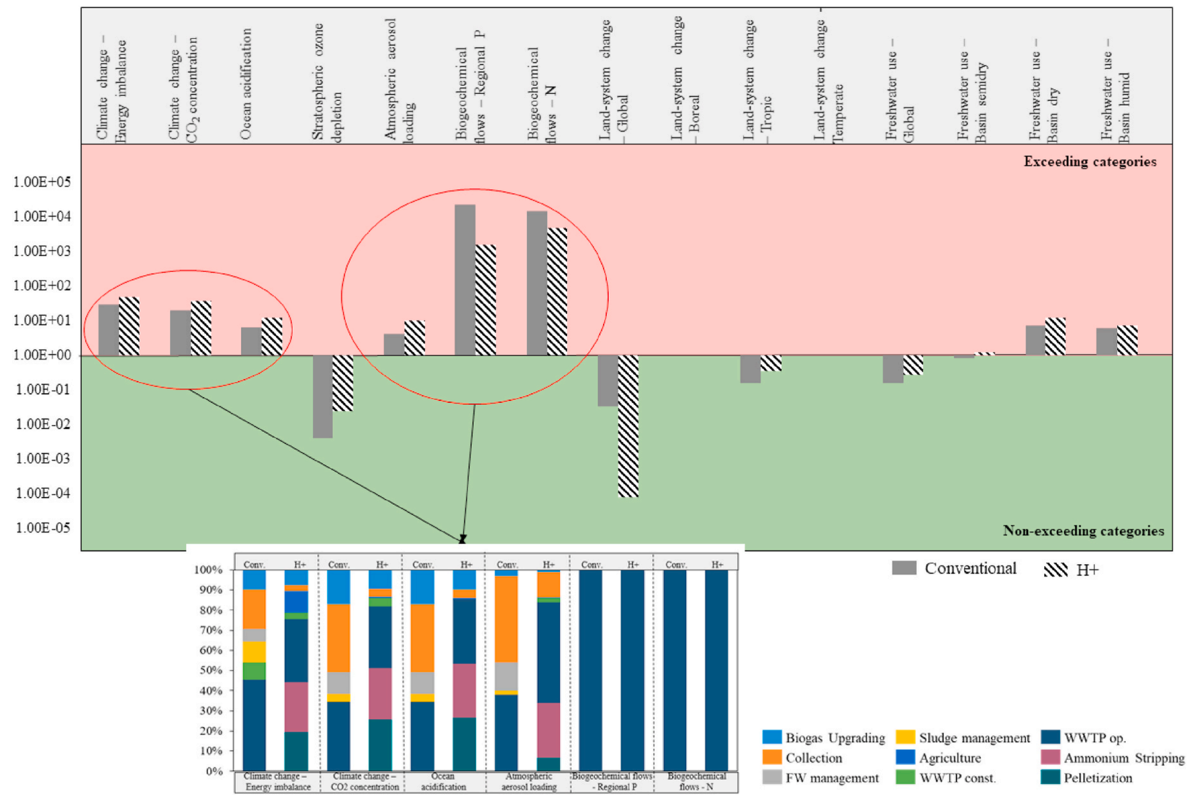

Fig. 3. Exceedance of the planetary boundaries by the H+ and conventional wastewater systems, represented by the ESR (red shading indicates exceedance, green shading non-exceedance) and (lower panel) the specific contributions of some categories exceeded. Note: WWTP const. refers to construction and WWTP op. to operation of the wastewater treatment plant.

Table 5
Complete characterized results of the traditional life cycle assessment using the ReCiPe® method (ReCiPe-LCA) for the conventional and H+ wastewater treatment systems in Helsingborg. Green highlights and * represent the best performing results.

Impact category	Unit	Conventional	H+
Global warming potential	kg CO_2 eq	27.9	11.4*
Stratospheric ozone depletion	kg CFC11 eq	7.31E-04	6.24E-05*
Ionizing radiation	kBq Co-60 eq	47.1*	88.1
Ozone formation - Human health	kg NOx eq	-0.0179	-0.00241*
Ozone formation - Terrestrial ecosystems	kg NO_x eq	-0.0199*	1.79E-04
Fine particulate matter formation	kg PM2.5 eq	0.0482	0.0440*
Terrestrial acidification	kg SO_2 eq	0.291	0.0925*
Freshwater eutrophication	kg P eq	6.08E-03	2.38E-03*
Marine eutrophication	kg N eq	0.512	0.0745*
Terrestrial ecotoxicity	kg 1,4-DCB	458*	513
Freshwater ecotoxicity	kg 1,4-DCB	9.18	7.81*
Marine ecotoxicity	kg 1,4-DCB	11.5	9.68*
Human carcinogenic toxicity	kg 1,4-DCB	10.7	-2.52*
Human non-carcinogenic toxicity	kg 1,4-DCB	75.6	48.9*
Land use	m^2a crop eq	-10.0*	6.28
Mineral resource scarcity	kg Cu eq	0.596	-0.145*
Fossil resource scarcity	kg oil eq	-8.57*	-6.53
Water consumption	m^3	0.907	-29.7*

Table 5, WC displayed large differences between the two systems, with H+ having significantly higher savings due to water reuse from the WWTP, avoided in the irrigation process.

The traditional LCA representation showed more clearly the benefits gained by adding different approaches to resource recovery, although the burdens were somewhat hidden at times.

3.2. *Specific contributions – correlation*

Fig. 5 shows the relative impact contributions of the two wastewater systems according to both assessment methods, considering the correlation between the categories shown in Table 1.

At first glance, the main difference between the two representations was that ReCiPe® LCA gave negative values and PB-LCA did not. Otherwise, the impact contributions were distributed quite equally, with the same processes having similar importance. The WWTP represented

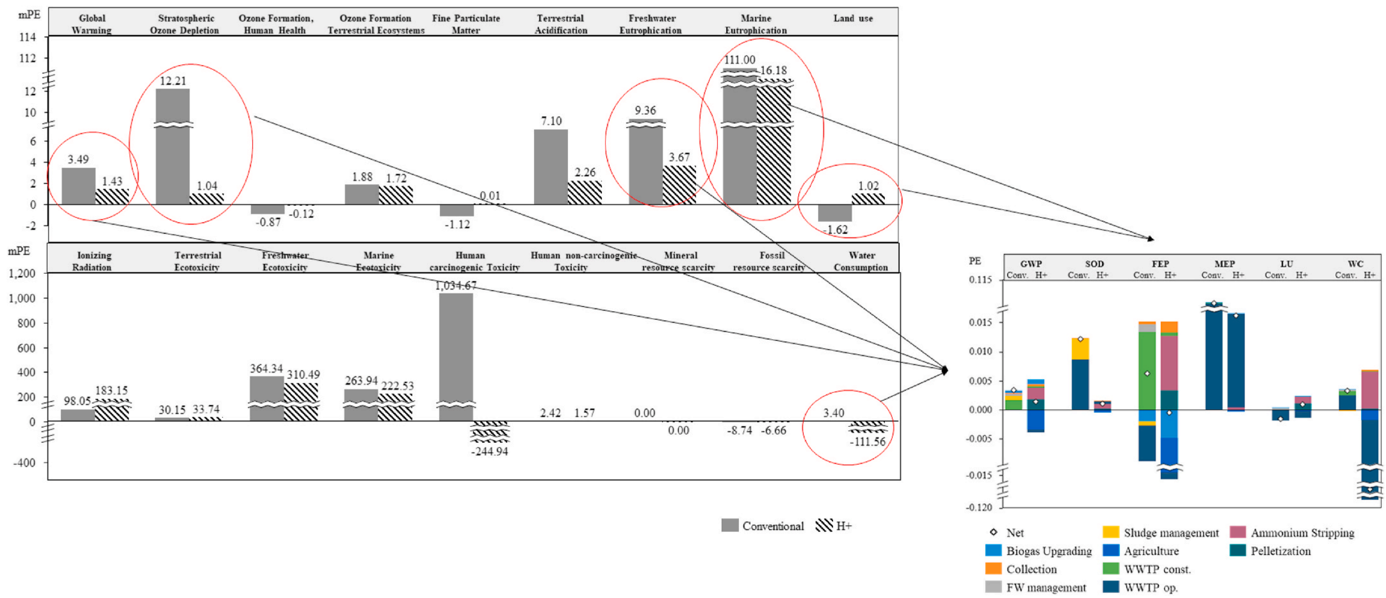

Fig. 4. Complete net normalized results (in milli person-equivalents, mPE) for the H+ and conventional wastewater systems with the ReCiPe® impact assessment methodology, and (lower panel) specific contributions of six important categories: Global warming potential (GWP), stratospheric ozone depletion (SOD), freshwater eutrophication (FEP), land use (LU), marine eutrophication (MEP) and water consumption (WC). Note: WWTP const. refers to construction and WWTP op. to operation of the wastewater treatment plant. NB: The diagrams have different scales.

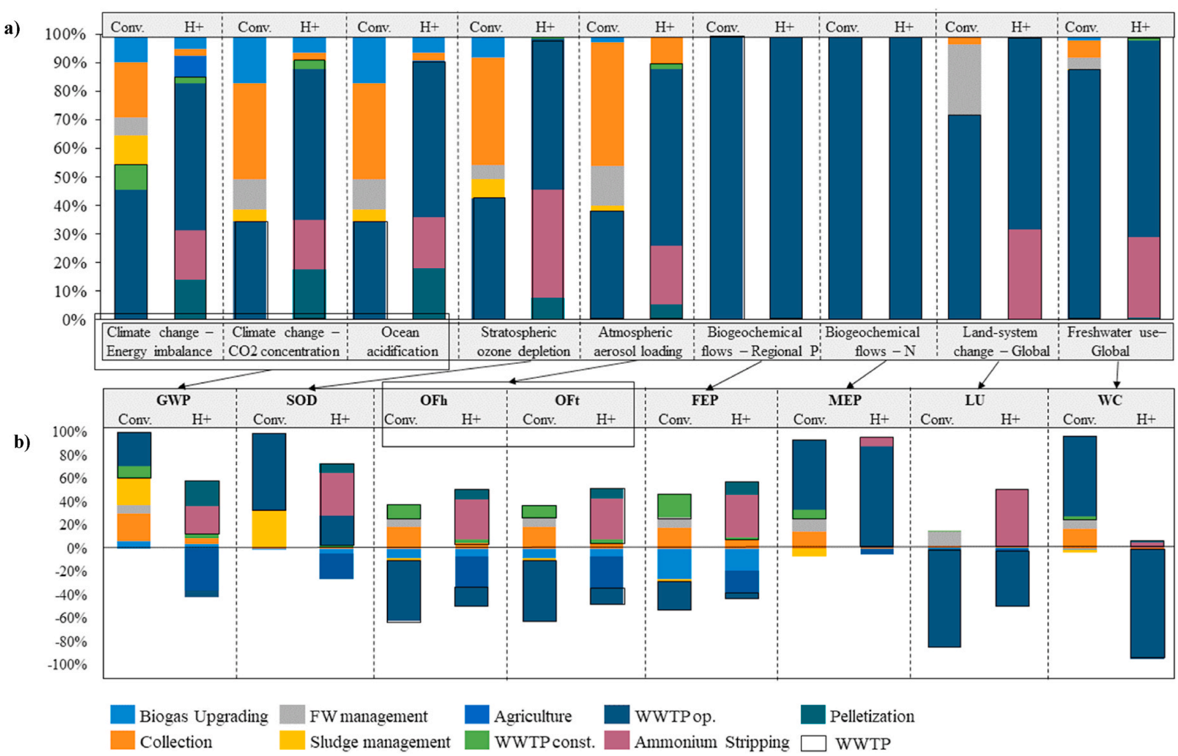

Fig. 5. Analysis of specific relative contributions of the conventional and H+ wastewater treatment systems to the categories: Global Warming Potential (GWP), Stratospheric Ozone Depletion (SOD), Ozone Formation - human health (OFh), Ozone Formation - terrestrial ecosystems (Oft), Freshwater Eutrophication (FEP), Marine Eutrophication (MEP), Land Use (LU) and Water Consumption (WC) according to a) PB-LCA and b) traditional LCA.

most emissions in both systems, but its operation was more significant in H+ and its construction was more significant in the conventional system.

For the climate change categories considered (energy imbalance, CO_2 concentration, ocean acidification, GWP), the results from traditional LCA and PB-LCA were very similar for energy imbalance and GWP in the conventional system, while WWTP construction impact was not visible for CO_2 concentration and ocean acidification in PB-LCA. For the H+ system, the small savings in GWP from WWTP op. represented the savings obtained in the WWTP and reduced all its burdens, resulting in net savings. The same applied for biogas upgrading, which appeared as a burden in PB-LCA and as savings in GWP, which is not the case, i.e., the burdens are hidden in the net savings. As verified in several LCAs of sanitation systems (Landry and Boyer, 2016; Lima et al., 2022; Shi et al., 2018), electricity consumption had high emissions, which were reflected in WWTP op. for both systems, followed by district heating.

For SOD the two assessment methods use different units, but they still showed similar ratios of contributions. In the traditional LCA the collection is not visible, but it was quite clear in the PB-LCA results. This is because the collection contribution in the ReCiPe® method represented 0.65% of the overall impacts for the conventional wastewater treatment system and the different units led to different characterization factors for the process.

Atmospheric aerosol loading also showed similar ratios with the two different assessment methods. It has two representative categories in the ReCiPe® method. Both ozone formation categories are measured in NO_2 emissions, as aerosol loading by aerosol optical depth (AOD), which takes into account different substances. However, the aspect mentioned above was also evident for these categories in terms of WWTP operation. Aside from ammonium stripping and small contributions from pelletization and WWTP construction, it seemed that the WWTP only gave savings in oxidation formation in traditional LCA, which was clearly not the case in PB-LCA, where WWTP operation was part of the burden.

For both biogeochemical flows (P and N), the corresponding categories in ReCiPe® (FEP and MEP) gave a very different visual representation. The PB-LCA chart only showed the contributions from nutrients directly discharged in treated water from the WWTP, as the characterization factors from the method are limited to those, which is one of the weaknesses that needs to be overcome. The ReCiPe® method considers other emissions in FEP and MEP resulting in more contributing processes being shown in the chart. It is difficult to draw conclusions from analyzing both methods, but it is possible to see that for FEP there are enough savings to reduce the burden from the WWTP, while for MEP this does not occur and the savings are minimal.

Land-system change is calculated in terms of percentage in PB-LCA, while LU in ReCiPe® is calculated in m^2a crop equivalents. Even with the different methods of calculation, the ratio of the contributions was rather similar, with the difference lying in the avoided processes in the ReCiPe® method that are not covered by the PB-LCA system boundaries.

Water consumption showed the opposite behavior to land use, with the H+ system having much lower impacts in traditional LCA due to the savings from recovered water, as the conventional system's net burdens were mainly from WWTP operation. The savings were so large that the negative impact from ammonium stripping appeared very small, but the PB-LCA clearly showed that use of chemicals in the ammonium stripping process alone was almost as significant as the remaining operations of the WWTP.

From this analysis, both assessments appear to complement each other. While it is important to consider the environmental savings provided by a system, the emissions deriving from it are no less important and need to be considered as hotspots in system improvement and future implementations.

3.2.1. Practical applications of the methods

By performing the two assessments methods for the same case study, we aimed to identify areas where each method should be applied. However, as the results showed, both methods have their advantages and disadvantages and our main finding is that they can be used to complement each other. Since PB-LCA is still a new approach, there are several improvements that should be performed, such as characterization factors considering more parameters and substances and fairer ways to calculate SoSOS.

If the objective of a study is to compare two or more alternatives and identify the best among these, then traditional LCA is better suited to this purpose due to the detailed verification of burdens and savings and potential hotspots. If the objective is to determine how close the alternatives are to being environmentally sustainable, then PB-LCA is preferable. PB-LCA is also preferable when performing eco-design, to show how much better the technology needs to be.

In our specific case, the Oceanhamnen district in Helsingborg, the better-performing wastewater system identified differed between the assessment methods, mainly due to fertilizer substitution. This indicates a need for valuation of biofertilizer production when calculating SoSOS, as the current value is probably too low. The fact that performance of a product is dependent on what it substitutes in LCA can also be problematic, as it is often not well known exactly what (if anything) is being substituted. Hence, the best (and most diplomatic) outcome from our case is to say that the methods should also be used in a complementary way to test whether conclusions from the comparison align. If the conclusions differ, further analyses will be required to determine the reason for this.

When performing the assessments as a complement to each other as one comparative study, they should ideally have the same system boundaries, to make the comparison fair. That means that the differences in avoided products should be considered, either by removing them in the Recipe assessment or by adding the equivalent consumptions in the PB-LCA as a way of obtaining the subtraction.

Hence, in future projects seeking to provide empirical support for decision-makers when considering sustainable alternatives in their entirety and system hotspots, both assessment methods should be used, taking into account the aspects mentioned and discussed here.

3.3. Uncertainty analysis

From the Monte Carlos runs in Simapro®, we obtained the values shown in Fig. 6 for ReCiPe® assessment and those in Fig. 7 for PB-LCA. According to traditional LCA, both sanitation systems appeared fairly similar in terms of uncertainty apart from water consumption, which had the highest uncertainty in the conventional system (range 1500 to −2250%) and ozone formation that had the highest in the H+ (range 25,000 to −35000%).

The PB-LCA gave much smaller uncertainties compared with the ReCiPe® method (see Fig. 7), aside from Land-system change – Global which had a wide uncertainty for both systems.

3.4. Source separation and environmental performance

Several studies have shown that source separation is an efficient way to recover nutrients and reduce the burden on oceans, but also to recover energy, nutrients, and water for re-use (Besson et al., 2021; Suvi Lehtoranta et al., 2022a, b; Lima et al., 2022). While the H+ source separation system did not show the best overall performance in the two assessment methods, we confirmed the advantages of nutrient recovery, provided it is done considering other factors and all trade-offs.

According to Andersson et al. (2020), there is a pressing need for humanity to manage its own resources in a way that can meet current and future needs. This includes linking food production with sanitation, as the nutrients and organic matter potentially recovered can be returned to agriculture. Source separation sanitation systems can increase N and P recovery by 3- to 10-fold (S. Lehtoranta et al., 2022a, b). This was verified in the H+ case, which showed 3-fold higher P recovery (to agricultural land) and 7-fold higher N recovery.

The actual environmental benefits, particularly the climate benefits, of any mitigation measure are dependent on policies and decisions on how the avoided emissions will be accounted for (S. Lehtoranta et al., 2022a, b). The main goal of the three-pipe H+ system in Helsingborg is to increase recovery of resources, namely fuel, nutrients, and water. If greywater were to be recirculated in the system or simply used for non-potable purposes, instead of treating it to potable water quality, some of the electricity impacts of the H+ system could be avoided and potentially some overall burdens. However, in that case, some valuable resources (water) could be lost or not fully exploited.

The technologies chosen for the H+ system, such as ammonium stripping and struvite precipitation to increase nutrient recovery, added

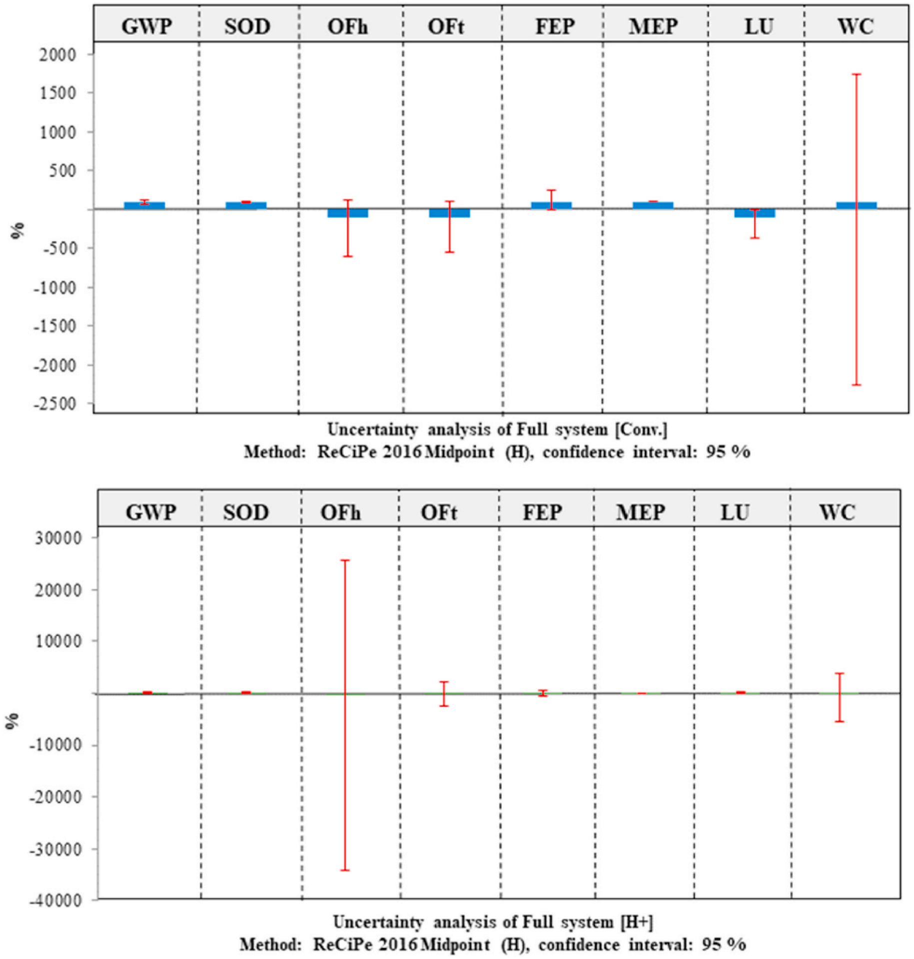

Fig. 6. Results of uncertainty analysis for the ReCiPe® impact assessment method. Top: conventional system; bottom: H+ system.

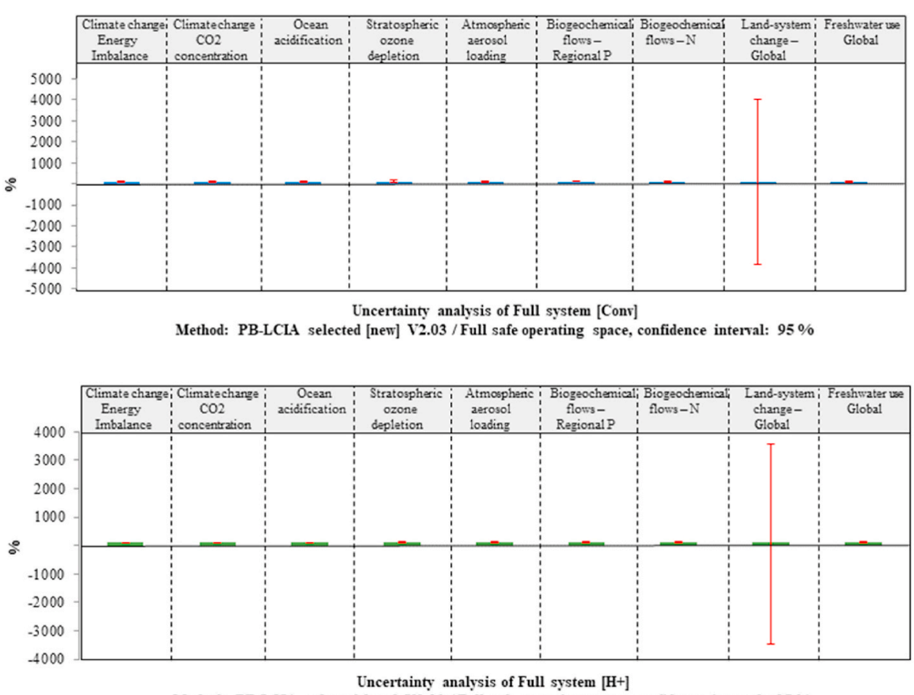

Fig. 7. Results of uncertainty analysis for the PB-LCA impact assessment method. Top: conventional system; bottom: H+ system.

considerable burdens, due to the use of chemicals and increased electricity consumption. Lam et al. (2020) concluded that while chemicals help increase nutrient recovery, the environmental burdens they bring are difficult to offset completely, as shown in studies by Bradford-Hartke et al. (2015) and Ishii and Boyer (2015). Further, electricity consumption is a pressing issue in this type of recovery system (Lima et al., 2022; Shi et al., 2018). Due to ambitions to phase out fossil fuel usage, Sweden is currently using biogas as bus fuel, but this trend can change in the future, e.g., electricity and heat generation is an alternative for biogas recovery. With the rise of electric vehicles and consequently electric buses, fuel trends might change and systems might need to adapt (Borén, 2020). A surplus of biogas could open other possibilities for its utilization, e.g., substituting for fossil carbon in other sectors, such as agriculture, heat, and electricity (Hamelin et al., 2021), which could potentially reduce the environmental impacts and help transition towards a low-carbon economy.

4. Conclusions

In terms of overall environmental performance, the conventional wastewater system in Sweden already has benefits from resource recovery (nutrients, gas). The new system increases resource recovery quantitatively and qualitatively (potable water quality), increasing the environmental performance in several impact categories. For even better performance, energy and chemical consumption and transportation hotspots identified in the H+ system should be addressed in future resource recovery systems.

Operation of the WWTP, especially the process of ammonium stripping, gave the most significant impacts in the new H+ system, due to consumption of chemicals. However, the increased recovery of biofertilizer, biogas, and water reduced the impacts significantly for climate change, freshwater eutrophication, and water consumption, respectively.

The analysis of the two LCA methods showed that PB-LCA and traditional LCA (ReCiPe impact assessment) have different representations and potentially different purposes. According to the PB-LCA results, neither of the sanitation systems assessed is absolute sustainable, as both exceeded SoSOS for eight out of the 15 boundaries studied. In fact, the new system showed higher emissions than the conventional system in six categories, and reduced impacts only for biogeochemical flows of N and P. According to the traditional ReCiPe® LCA results, the H+ system performed better in 10 out of 18 climate impact categories, due to the avoided emissions, but net savings were achieved in only four categories.

These discrepancies in results arose because PB-LCA leaves out avoided burdens (such as mineral fertilizer production and fossil fuel for buses), and thus it does not consider a very important part of the resource recovery system that offsets some of the environmental burdens. This can be rectified with better methods for assigning SoSOS. The savings are shown in the ReCiPe® method, but the overall burdens can be hidden and not taken into consideration. Therefore, the assessments can be used to complement each other. However, PB-LCA still needs further development to improve its utility as a decision-making tool. Combined use of PB-LCA and conventional LCA can guide future technology development to become both more efficient and sustainable for the planet.

CRediT authorship contribution statement

Priscila de Morais Lima: Conceptualization, Methodology, Investigation, Writing – original draft. **Gertri Ferrer:** Conceptualization, Methodology. **Hamse Kjerstadius:** Conceptualization, Methodology, Supervision. **Morten Ryberg:** Methodology, Validation. **Jennifer Rae McConville:** Conceptualization, Validation, Writing – review & editing.

Declaration of competing interest

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

Data availability

I have shared the data as a Supplementary Material file

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2023.138632.

References

Andersson, K., Rosemarin, A., Lamizana, B., Kvarnström, E., McConville, J., Seidu, R., Dickin, S., Trimmer, C., 2020. SANITATION , WASTEWATER MANAGEMENT AND SUSTAINABILITY, second ed.

Benetto, E., Nguyen, D., Lohmann, T., Schmitt, B., Schosseler, P., 2009. Life cycle assessment of ecological sanitation system for small-scale wastewater treatment. Sci. Total Environ. 407 (5), 1506–1516.

Besson, M., Aguas, J., Berger, S., Tiruta-barna, L., Paul, E., Spérandio, M., 2021. Life Cycle Assessment of Wastewater Source Separation Scenario : Case Study on a New District in.

Bjørn, A., Chandrakumar, C., Boulay, A.M., Doka, G., Fang, K., Gondran, N., Hauschild, M.Z., Kerkhof, A., King, H., Margni, M., McLaren, S., Mueller, C., Owsianiak, M., Peters, G., Roos, S., Sala, S., Sandin, G., Sim, S., Vargas-Gonzalez, M., Ryberg, M., 2020a. Review of life-cycle based methods for absolute environmental sustainability assessment and their applications. Environ. Res. Lett. 15 https://doi.org/10.1088/1748-9326/ab89d7.

Bjørn, A., Richardson, K., Hauschild, M.Z., 2019. A framework for development and communication of absolute environmental sustainability assessment methods. J. Ind. Ecol. 23, 838–854. https://doi.org/10.1111/jiec.12820.

Bjørn, A., Sim, S., King, H., Patouillard, L., Margni, M., Hauschild, M.Z., Ryberg, M., 2020b. Life cycle assessment applying planetary and regional boundaries to the process level: a model case study. Int. J. Life Cycle Assess. 25, 2241–2254. https://doi.org/10.1007/s11367-020-01823-8.

Borén, S., 2020. Electric buses' sustainability effects, noise, energy use, and costs. Int. J. Sustain. Transp. 14, 956–971. https://doi.org/10.1080/15568318.2019.1666324.

Bradford-Hartke, Z., Lane, J., Lant, P., Leslie, G., 2015. Environmental benefits and burdens of phosphorus recovery from municipal wastewater. Environ. Sci. Technol. 49, 8611–8622. https://doi.org/10.1021/es505102v.

Diaz-Elsayed, N., Rezaei, N., Guo, T., Mohebbi, S., Zhang, Q., 2019. Wastewater-based resource recovery technologies across scale: a review. Resour. Conserv. Recycl. 145, 94–112. https://doi.org/10.1016/j.resconrec.2018.12.035.

Dixon, A., Simon, M., Burkitt, T., 2003. Assessing the environmental impact of two options for small-scale wastewater treatment: comparing a reedbed and an aerated biological filter using a life cycle approach. Ecol. Eng. 20 (4), 297–308.

Gallego, A., Hospido, A., Moreira, M.T., Feijoo, G., 2008. Environmental performance of wastewater treatment plants for small populations. Resourc. Conserv. Recycl. 52 (6), 931–940.

Garfí, M., Flores, L., Ferrer, I., 2017. Life cycle assessment of wastewater treatment systems for small communities: activated sludge, constructed wetlands and high rate algal ponds. J. Clean. Prod. 161, 211–219.

Goedkoop, M.J., Heijungs, R., Huijbregts, M.A.J., Schryver, A. De, Struijs, J., van Zelm, R., 2013. Category indicators at the midpoint and the endpoint level ReCiPe 2008. ResearchGate 126.

Hamelin, L., Møller, H.B., Jørgensen, U., 2021. Harnessing the full potential of biomethane towards tomorrow's bioeconomy: a national case study coupling sustainable agricultural intensification, emerging biogas technologies and energy system analysis. Renew. Sustain. Energy Rev. 138 https://doi.org/10.1016/j.rser.2020.110506.

Hjalsted, A.W., Laurent, A., Andersen, M.M., Olsen, K.H., Ryberg, M., Hauschild, M., 2021. Sharing the safe operating space: exploring ethical allocation principles to operationalize the planetary boundaries and assess absolute sustainability at individual and industrial sector levels. J. Ind. Ecol. 25, 6–19. https://doi.org/10.1111/jiec.13050.

Huijbregts, M.A.J., Steinmann, Z.J., Elshout, P.M., Stam, G., Verones, F., Vieira, M., Van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. The International Journal of Life Cycle Assessment 22, 138–147.

Ishii, S.K.L., Boyer, T.H., 2015. Life cycle comparison of centralized wastewater treatment and urine source separation with struvite precipitation: focus on urine nutrient management. Water Res. 79, 88–103. https://doi.org/10.1016/j.watres.2015.04.010.

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

ISO, 2006b. ISO 14044-Environmental management - Life cycle assessment - Requirements and guidelines.

Jönsson, H., Baky, A., Jeppsson, U., Hellström, D., Kärrman, E., 2005. Composition of urine, feaces, greywater and biowaste for utilisation in the URWARE model. Report 2005:6. Urban Water Rep 6, 1–49, 2005.

Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2016. Life cycle-based decision support tool for selection of wastewater treatment alternatives. J. Clean. Prod. 117, 64–72.

Kjerstadius, H., Bernstad Saraiva, A., Spångberg, J., Davidsson, 2017. Carbon footprint of urban source separation for nutrient recovery. J. Environ. Manag. 197, 250–257. https://doi.org/10.1016/j.jenvman.2017.03.094.

Kjerstadius, H., Haghighatafshar, S., Davidsson, A., 2015. Potential for nutrient recovery and biogas production from blackwater, food waste and greywater in urban source control systems. Environ. Technol. 36, 1707–1720. https://doi.org/10.1080/09593330.2015.1007089.

Lam, K.L., Zlatanović, L., van der Hoek, J.P., 2020. Life cycle assessment of nutrient recycling from wastewater: a critical review. Water Res. 173 https://doi.org/10.1016/j.watres.2020.115519.

Landry, K.A., Boyer, T.H., 2016. Life cycle assessment and costing of urine source separation: focus on nonsteroidal anti-inflammatory drug removal. Water Res. 105, 487–495. https://doi.org/10.1016/j.watres.2016.09.024.

Larsen, T.A., Gruendl, H., Binz, C., 2021. The potential contribution of urine source separation to the SDG agenda – a review of the progress so far and future development options. Environ. Sci. Water Res. Technol. 7, 1161–1176. https://doi.org/10.1039/d0ew01064b.

Lehtoranta, Suvi, Laukka, V., Vidal, B., Heiderscheidt, E., Postila, H., Nilivaara, R., Herrmann, I., 2022a. Circular economy in wastewater management—the potential of source-separating sanitation in rural and peri-urban areas of northern Finland and Sweden. Front. Environ. Sci. 10, 1–18. https://doi.org/10.3389/fenvs.2022.804718.

Lehtoranta, S., Malila, R., Särkilahti, M., Viskari, E.L., 2022b. To separate or not? A comparison of wastewater management systems for the new city district of Hiedanranta, Finland. Environ. Res. 208 https://doi.org/10.1016/j.envres.2022.112764.

Lima, P. de M., Lopes, T.A. de S., Queiroz, L.M., McConville, J.R., 2022. Resource-oriented sanitation: identifying appropriate technologies and environmental gains by coupling Santiago software and life cycle assessment in a Brazilian case study. Sci. Total Environ. 837, 155777 https://doi.org/10.1016/j.scitotenv.2022.155777.

Lundin, M., Bengtsson, M., Molander, S., 2000. Life cycle assessment of wastewater systems: influence of system boundaries and scale on calculated environmental loads. Environ. Sci. Technol. 34 (1), 180–186.

Piao, W., Kim, Y., Kim, H., Kim, M., Kim, C., 2016. Life cycle assessment and economic efficiency analysis of integrated management of wastewater treatment plants. J. Clean. Prod. 113, 325–337.

Pizzol, M., Laurent, A., Sala, S., Weidema, B., Verones, F., Koffler, C., 2017. Normalisation and weighting in life cycle assessment: quo vadis? Int. J. Life Cycle Assess. 22, 853–866. https://doi.org/10.1007/s11367-016-1199-1.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., De Wit, C.A., Hughes, T., Van Der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature 461, 472–475. https://doi.org/10.1038/461472a.

Ryberg, M.W., Andersen, M.M., Owsianiak, M., Hauschild, M.Z., 2020. Downscaling the planetary boundaries in absolute environmental sustainability assessments – a review. J. Clean. Prod. 276 https://doi.org/10.1016/j.jclepro.2020.123287.

Ryberg, M.W., Bjerre, T.K., Nielsen, P.H., Hauschild, M., 2021. Absolute environmental sustainability assessment of a Danish utility company relative to the Planetary Boundaries. J. Ind. Ecol. 25, 765–777. https://doi.org/10.1111/jiec.13075.

Ryberg, M.W., Owsianiak, M., Richardson, K., Hauschild, M.Z., 2018. Development of a life-cycle impact assessment methodology linked to the Planetary Boundaries framework. Ecol. Indicat. 88, 250–262. https://doi.org/10.1016/j.ecolind.2017.12.065.

Shi, Y., Zhou, L., Xu, Y., Zhou, H., Shi, L., 2018. Life cycle cost and environmental assessment for resource-oriented toilet systems. J. Clean. Prod. 196, 1188–1197. https://doi.org/10.1016/j.jclepro.2018.06.129.

Spångberg, J., Tidåker, P., Jönsson, H., 2014. Environmental impact of recycling nutrients in human excreta to agriculture compared with enhanced wastewater treatment. Sci. Total Environ. 493, 209–219. https://doi.org/10.1016/j.scitotenv.2014.05.123.

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., De Vries, W., De Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: guiding human development on a changing planet. Science 80, 347. https://doi.org/10.1126/science.1259855.

United Nations, 2015. Transforming Our World: the 2030 Agenda for Sustainable Development.

Zhang, S., Ericsson, N., Hansson, P.A., Sjödin, M., Nordberg, Å., 2022. Life cycle assessment of an all-organic battery: hotspots and opportunities for improvement. J. Clean. Prod. 337 https://doi.org/10.1016/j.jclepro.2022.130454.