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Dynamic process simulation for life cycle inventory data acquisition – Environmental assessment of biological and chemical phosphorus removal

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ABSTRACT

In Sweden, phosphorus is commonly removed from municipal wastewater treatment by chemical precipitation (CP). Recently, such alternatives as enhanced biological phosphorus removal (EBPR) have garnered interest due to the increased risk of chemical shortage. In this study, a life cycle assessment (LCA) was performed to compare EBPR and CP in three scenarios: 1) baseline – precipitation chemicals available, 2) stricter effluent requirements – precipitation chemicals available, and 3) chemical shortage – no precipitation chemicals available. Data acquisition that was based on dynamic process simulation was useful, yielding more site-specific results, in contrast to standard literature values. The results indicated substantial differences in greenhouse gas emissions between configurations (around three times higher methane emissions for EBPR compared to CP configurations – although this finding requires further validation). These differences suggest that different emission factors for EBPR and CP should be considered. Furthermore, it is suggested to include waterline methane emissions, at least when the configuration incorporates anaerobic reactors in the water line. Further validation of emissions is necessary, especially for EBPR plants with side-stream hydrolysis and digester reject water treatment. The LCA results showed a similar overall environmental impact for both configurations, but the results of individual impact categories differed. EBPR caused greater climate impact due to the larger direct emissions of methane. Toxicity was more important for CP, based on the inherent heavy metal content in precipitation chemicals. Freshwater eutrophication was similar for both configurations, assuming that precipitation chemicals were available. However, if the recipient is sensitive, implementing EBPR reduces the freshwater eutrophication potential by 75% during a chemical shortage, and should be considered.

Abbreviations:

ASM1	Activated sludge model no. 1
CH_4	Methane
CO_2	Carbon dioxide
COD	Chemical oxygen demand
CP	Chemical precipitation
EBPR	Enhanced biological phosphorus removal
EU	European Union
GHG	Greenhouse gas
MBBR	Moving bed biofilm reactor
NH_3	Ammonia
NH_4	Ammonium
N_2O	Nitrous oxide
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
PAO	Polyphosphate-accumulating organism
PE	Population equivalents
PO_4	Phosphate
TN	Total nitrogen
TP	Total phosphorus
TS	Total solids
VFA	Volatile fatty acids
WWTP	Wastewater treatment plant

(*continued on next column*)

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1. Introduction

The removal of phosphorus from wastewaters is vital to reduce the risk of eutrophication in receiving waters. At wastewater treatment plants (WWTPs), this is commonly accomplished through chemical precipitation (CP), in which a metal salt is added, precipitating the phosphorus for removal with the sludge. An alternative approach is enhanced biological phosphorus removal (EBPR), which has been implemented in some plants in full scale since the 1970s (Barnard, 1983). EBPR relies on polyphosphate-accumulating organisms (PAOs) to take up phosphorus from the water for growth and energy storage when exposed to alternating anaerobic and aerobic environments. The phosphorus is then removed with the excess sludge.

In Sweden, there are 429 WWTPs serving over 2000 population equivalents (PE) (Villner and Myhr, 2022), approximately 30 of which currently operate using EBPR—often supplemented with chemical post-precipitation (Jönsson et al., n.d.). Recently, concerns have been raised over the risk for shortage of chemicals (Naturvårdsverket, 2023). A priority list for access to precipitation chemicals has thus been generated, with drinking water production as the highest concern, followed by wastewater treatment (Sohlström et al., 2022). Furthermore, upcoming changes in the EU wastewater directive will demand more stringent effluent requirements as well as climate neutrality in the water sector (Council of the EU, 2024). To manage decisions in this context, a systems perspective is necessary in order to identify trade-offs and avoid burden-shifting, and one of the tools available for this is life cycle assessment (LCA).

The foundation of a robust LCA is reliable, relevant and accessible data (Baumann and Tillman, 2004). Niero et al. (2014) compared four types of WWTP configurations (among them both CP and EBPR), based on actual data from existing plants in Denmark. Using data from existing WWTPs ensures that the results are relevant and applicable to the geographical setting. However, that study omitted an estimation of direct air emissions entirely, despite the often substantial contribution of these emissions to the total environmental impact of a WWTP (Zang et al., 2015). Many WWTPs do in fact not measure air emissions, and another method of estimating such emissions is therefore needed. One such option is simulation.

Process model simulation has been used extensively to evaluate and optimise WWTP operations, in academia and in practice, since the publication of Activated Sludge Model No. 1 (ASM1) (Henze et al., 1987). This model was presented as a platform on which to build further, with the possibility of increasing complexity when required to solve a problem or as new knowledge becomes available. Since then, new models have been published, expanding on the general structure of ASM1. Several commercial simulation platforms are also available [BioWin (Envirosim, Canada), GPS-X (Hatch, Canada), Simba# (ifak, Germany), Sumo (Dynamita, France), WEST (DHI, Denmark)], many of which have in-house-developed models that have improved on the short-comings of earlier models. Whereas ASM1 focused on describing processes for the biological removal of organic material and nitrogen, subsequent developments have incorporated processes for biological reactions that are related to phosphorus (Barker and Dold, 1997; Henze et al., 1999), EBPR with side-stream hydrolysis and fermentation (Varga et al., 2018), sulfur (Flores-Alsina et al., 2016), greenhouse gas emissions (Hiatt and Grady, 2008; Pocquet et al., 2016) and chemical phosphorus removal (Hauduc et al., 2015).

The use of process simulation models to acquire data for life cycle inventories is becoming increasingly common (see e.g., Besson et al., 2021; Bisinella de Faria et al., 2015; Foley et al., 2010; Igos et al., 2017; Monje et al., 2022; Ontiveros and Campanella, 2013; Rahmberg et al., 2020). The benefits of using dynamic process simulation for data acquisition include the possibility of examining various operational scenarios for a WWTP in a fraction of the time and effort of gathering data from actual full-scale runs. This potential is especially important for LCAs with a prospective approach where the WWTP of interest might not yet have been built.

In this study, we performed an LCA that compared EBPR and CP at a planned WWTP in Sweden using process simulation as the basis for inventory data acquisition. This article is based on a study by Jönsson et al. (n.d.) that examined the environmental impact of increased implementation of EBPR in Sweden. A similar study by Rahmberg et al. (2020) also compared these two configurations, although they did not consider sludge management or product use. Furthermore, their report focused on climate change and energy use only. In our study, a more extensive system was assessed, covering also the utilisation of energy and nutrient products and our study accounted for other environmental impacts, such as eutrophication and toxicity, in addition to climate change. Moreover, we explored the prospect of a chemical shortage, a perspective that has not yet been highlighted in this field. The aims of our study were therefore both to evaluate the use of a process simulation model for data acquisition in an LCA and to assess the environmental impacts of, and the differences between, EBPR and CP in a Scandinavian setting, representing a cold climate, strict effluent requirements and potential chemical shortage.

2. Methods

2.1. WWTP modelling

Dynamic simulation models over two plant-wide WWTP configurations (EBPR and CP) were developed in Sumo, version 22.1.0, allowing the use of the same influent characteristics. Simulations were performed in the Sumo4N setting, including biological phosphorus removal (Varga et al., 2018), chemical precipitation (Hauduc et al., 2015) and estimates of nitrous oxide emissions (Hiatt and Grady, 2008; Pocquet et al., 2016).

The EBPR configuration was based on a planned, medium-sized (45 000 PE, 14 900 m^3/d) municipal WWTP in Lidköping in southern Sweden, and followed project planning documents with technical descriptions (Dahlberg, 2019). A previous process model (Wärff, 2021) was further developed and used as the basis for this work (Tables SI–1). No model calibration was possible, but location-specific data, such as the characteristics of influent to the current WWTP in the municipality, were available. The developed model includes a primary settler, activated sludge reactors with EBPR, side-stream hydrolysis, sludge thickening, anaerobic digestion, dewatering, struvite precipitation, ozonation with a subsequent moving bed biofilm reactor (MBBR) as biological post-treatment, the possibility of post-precipitation and, finally, disc filters (Fig. 1a).

The CP configuration is based on the same treatment plant, with several alterations: no side-stream hydrolysis, no phosphorus release reactor and no struvite precipitation process (Fig. 1b). Further, the recirculation flow is directed to the first biological tank, rendering it anoxic instead of anaerobic, and additional dosing points for chemical precipitation were added (pre-precipitation and simultaneous precipitation).

Important parameters, such as activated sludge volume, aeration control and sludge content in activated sludge reactors, have been kept identical in the two models, whereas operational differences, such as retention time and efficiency in thickeners, may assume different values depending on configuration. The primary sludge thickener has a 60% efficiency in the EBPR configuration and a 90% efficiency in the CP. The lower efficiency of the former results in more organic matter entering the side-stream hydrolysis, increasing the production of volatile fatty acids (VFAs) that are necessary for EBPR. In the CP configuration, the higher thickener efficiency provides more organic matter to the digesters and, hence, a greater biogas yield. For most model parameters, the default values have been used. The MBBRs, however, were modelled as activated sludge reactors with virtual settlers that was calibrated to obtain the same results as the Sumo biofilm processes in a so-called apparent kinetics approach (Baeten et al., 2019).

The model controllers were tuned to attain the anticipated treatment

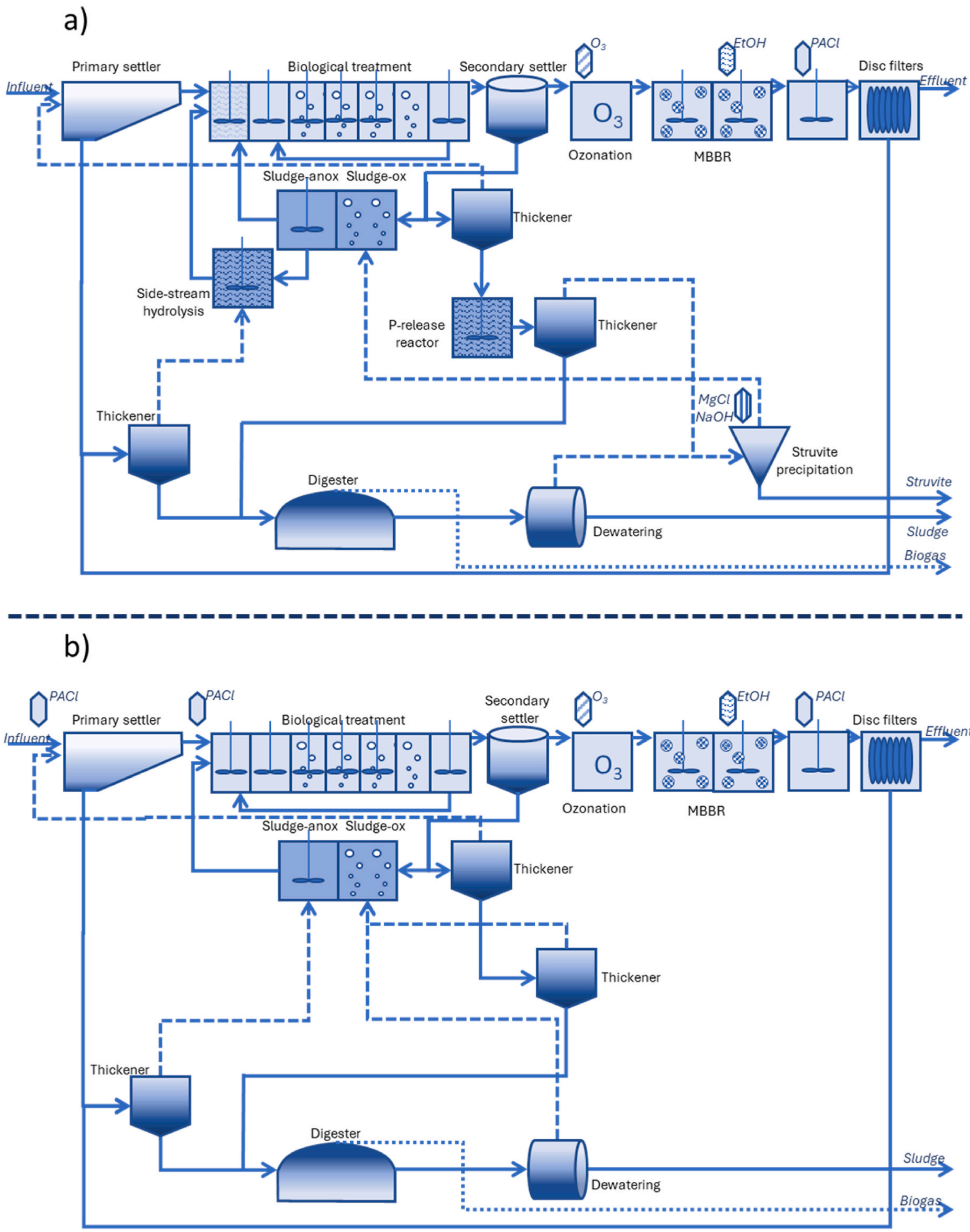

Fig. 1. Process flow diagrams of the two Sumo models: a) EBPR configuration and b) CP configuration.

requirements (Table 1). Model inputs were based on influent flow and temperature (1-h time resolution, from actual data for 2019, then scaled by PE to fit the design). Pollutant loads (COD, TN and TP) were based on measurement campaigns in 2020 and 2023 for influent characterisation, assuming a constant daily average with hourly variations (Table 1, details in SI-section 1.1). The dynamic simulations were run for one year to encompass seasonal variations in flow and temperature. Eight scenarios, as outlined in Table 2, were developed to account for differences in configuration and seasonal patterns.

2.2. *Scope*

This LCA has ISO 14040:2006 as a starting point. A WWTP today generally has several functions, such as treatment of wastewater and production of energy and nutrients; in this study, wastewater treatment was deemed to be the most important. The functional unit for treatment of wastewater is often expressed in flow of incoming or treated amounts (Corominas et al., 2020). Because effluent quality varies between scenarios, a functional unit that is related to the influent was chosen and set

Table 1
Influent water quality and effluent requirements.

Parameter	Influent yearly total	Influent average per m^3	Effluent requirements, annual average
Flow	4 960 000 m^3/year	–	–
COD	2708 t COD/year	551 g COD/m^3	–
BOD	–	–	<10 mg/l
TN	227 t N/year	46.2 g N/m^3	<10 mg/l
NH_4-N	177 t NH_4-N/year	36.2 g NH_4-N/m^3	<3 mg/l
TP	27.7 t P/year	5.6 g P/m^3	<0.2 mg/l
PO_4-P	15.0 t PO_4-P/year	3.0 g PO_4-P/m^3	–

Table 2
The eight scenarios. In the base cases (-B), the control feedback setpoint of the effluent was set to 0.05 mg PO_4/l, whereas in the scenarios with stricter effluent demand (-S), this setpoint was 0.01 mg PO_4/l. In scenarios with chemical shortage (−0), there was no phosphorus setpoint since no precipitation chemicals were added.

Short name	Definition	Use of post-precipitation
EBPR-B	EBPR, Base case	Yes
EBPR-S	EBPR, Stricter effluent demand	Yes
EBPR-0	EBPR, No precipitation chemicals	No
CPp-B	CP with pre-precipitation, Base case	Yes
CPp-S	CP with pre-precipitation, Stricter effluent demand	Yes
CPs-B	CP with simultaneous precipitation, Base case	Yes
CPs-S	CP with simultaneous precipitation, Stricter effluent demand	Yes
CP-0	CP, No precipitation chemicals	No

to “treatment of incoming municipal wastewater for 45 000 PE for 1 year”.

Attributional LCA methodology was applied with regard to data selection (the use of average data, rather than marginal data). Multifunctionality, however, was handled through system expansion by substitution, which can be considered a more consequential approach. This hybridisation of attributional and consequential methodologies is common in wastewater LCA studies (see Heimersson et al. (2019) for reasons and options).

In addition to the WWTPs with either EBPR or CP configurations, the system included the production of input chemicals and energy; transport of chemicals, sludge and struvite; direct emissions of methane (CH_4) and nitrous oxide (N_2O); combined heat and power production from generated biogas; the impact of the effluent on the recipient; sludge storage; and agricultural application of sludge and struvite (Fig. 2).

The study was limited to 1 year of operation, thus excluding the construction and demolition of infrastructure. Other LCA studies of

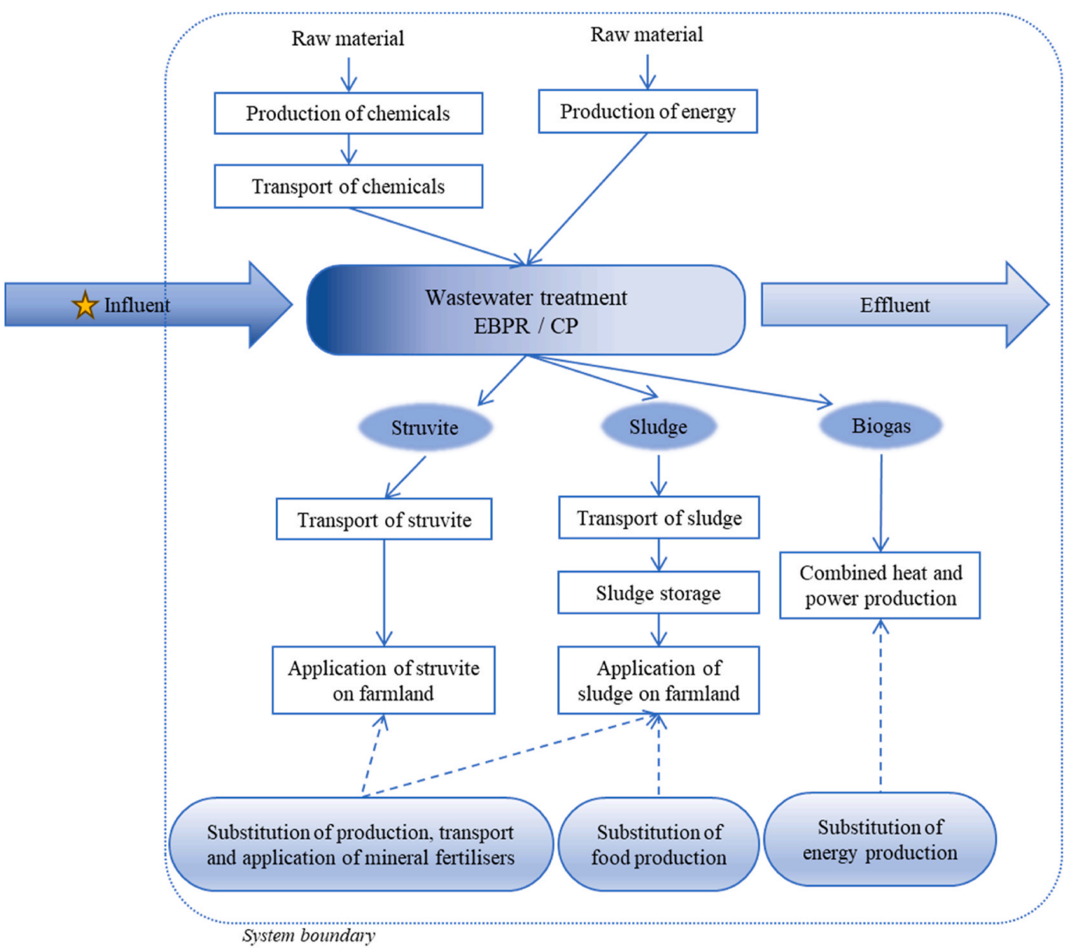

Fig. 2. System boundaries of the LCA for the two WWTP configurations. The study begins when incoming wastewater enters the treatment plant and ends after effluent has been discharged to the recipient. The reference flow, marked with a star, is 4.96 Mm^3/year.

wastewater systems have found the impact of construction to be negligible due to their long life span (Lundie et al., 2004). Yet, the impact of construction may increase in relation to the operation when the share of fossil-free energy input increases. Thus, there are recommendations to include the construction and demolition phases in an LCA (Corominas et al., 2020). In this study, however, the differences between configurations with regard to construction and demolition were deemed to be minor compared with the total demands of the WWTPs, further justifying the decision to exclude construction and demolition from the analysis.

Because the study focused on comparing WWTP configurations, different means of sludge management and disposal options were not evaluated. However, sludge use was included to account for some differences in sludge and nutrient quality. Sludge stabilisation, a common strategy in Sweden, was considered, through outdoor, uncovered storage of digested sludge for a minimum of six months, and subsequent spreading on farmland (Revaq, 2022). In Sweden, 46% of municipal sludge is spread on farmland or forestland (Villner and Myhr, 2022). Nitrogen and phosphorus in sludge and struvite have been assumed to substitute mineral fertilisers (ammonium nitrate and triple superphosphate, respectively). Other nutrients were not considered. The parameters which we considered portrayed differences between EBPR sludge and CP sludge included amount, nutrient content, nutrient bioavailability and heavy metal content. Furthermore, the benefits of carbon sequestration in soil and increased soil organic carbon content were taken into account. Additional details can be found in Sections 3.2.7–8.

The geographic setting was based on Scandinavian characteristics in terms of climate, which affects the wastewater temperature and thus biological processes at the treatment plant, as well as emission factors during storage and after spreading on farmland. A Swedish average electricity mix (40% nuclear, 40% hydro, 11% wind, 6% biomass and 3% miscellaneous (Sphera, 2023)) was used. Datasets for chemical production were based on European data for availability reasons.

The time frame of the study is the immediate present; the one year of operation is assumed to occur in the present or near future, based on the assumption that these types of plant configuration exist today and chemical shortage can happen soon. Thus, the current energy and transport systems were assumed to be relevant. In the process model, seasonal variations of WWTP influent flow and temperature were accounted for, but dynamics on a greater spatial or temporal scale, such as fluctuations in the energy market, were not considered in the study.

2.3. Life cycle inventory

Foreground data were based primarily on the model simulations (i.e., chemical and energy use, direct greenhouse gas (GHG) emissions and effluent and sludge qualities in terms of nutrients and organic matter). Heavy metal content; sludge storage emissions; and farmland application of sludge, struvite and mineral fertilisers were based on data in the literature (see details in Section 4). Background data, such as the production of chemicals and energy, were obtained from the LCA for Experts (formerly GaBi), version 10.7.1.28 (Sphera, 2023) and ecoinvent 3.9.1 (ecoinvent, 2023) databases.

2.4. Life cycle impact assessment

The LCA was modelled in LCA for Experts (Sphera, 2023). Impact assessment was performed according to Environmental Footprint (EF) 3.1, as recommended by the European Commission (2021). This choice was also based on the convention that toxicity-related categories are modelled according to USEtox (Rosenbaum et al., 2008), following the recommendations of UNEP and SETAC, as well as on the availability of recently updated characterisation, normalisation (Andreasi Bassi et al., 2023) and weighting (Sala et al., 2018) factors (Tables SI–12), which were applied as per Zampori and Pant (2019). The normalisation factors were based on Crenna et al. (2019), with global-scale emissions and resource use for the reference year 2010, and EC-JRC (2021) for additional data on non-methane volatile organic compounds (NMVOCs). Sala et al. (2018) based the weighting factors on surveys of and discussions with laymen and experts, also aggregating several sets of weighting factors (a hybridisation of panel-based, evidence-based and expert-judgement weighting approaches). In our study, the characterised results (per functional unit) were divided by 45 000 PE and then normalised by division by the annual environmental impact of an average global citizen. Next, the results were multiplied by the weighting factor. Thus, the unit became dimensionless, and the results show the internal relations between impact categories.

2.5. Interpretation

Sensitivity analysis was performed on three levels: process model, LCA model and LCIA method. The sensitivity of the process model was evaluated by varying three parameters that influence the apparent fermentation rate—a critical design parameter according to Downing et al. (2023). These parameters—anaerobic decay rate ($b_{OHO,\ AN}$), anaerobic hydrolysis rate ($q_{Hyd,\ AN}$) and half-saturation of particulates in hydrolysis (K_{Hyd})—are components of the anaerobic reactions that govern decay, hydrolysis and fermentation (Figure SI-4). Parameter default values from four simulation models were obtained from Downing et al. (2023, Table 9-1) and grouped into four parameter sets. Because parameters may be designed to interact with each other, we assumed that altering the entire set of parameters would render a result to be more relevant than if only one parameter were to be changed at a time. The original Sumo parameter set is referred to as "Default" and the others: "A", "B" and "C" (Tables SI-3). Notably, the hydrolysis rates of "Default" and "C" are substantially greater than for "A" and "B". The parameters within the digester were unchanged. The parameter sets were then inserted in the EBPR-B and CPp-B models and run initially on the process model level; then, the results were implemented at the LCA model level. Another aspect of the process model that was examined in the sensitivity analysis was the presence of PAOs and glycogen accumulating organisms (GAOs) in the EBPR and CP configurations.

Table 3
Scenarios for sensitivity analysis on the process model, LCA model, and LCIA method levels.

Name	Parameter	Base Case	Variation
Process model level scenarios			
A	Apparent fermentation rate	Sumo default values	Parameter set A
B	Apparent fermentation rate	Sumo default values	Parameter set B
C	Apparent fermentation rate	Sumo default values	Parameter set C
noPAO	Presence of EBPR bacteria	Growth rate of PAO and GAO set to 1/d (default value)	Growth rate of PAO and GAO set to 0
LCA model level scenarios			
GHG	Greenhouse gases	CH_4 and N_2O from simulations	CH_4 and N_2O from climate calculation tool
Fossil	Energy sources	Swedish electricity mix Swiss incineration of household waste European production of ethanol from rye	Swedish electricity production from oil Swedish heat production from oil German production of ethanol from fossil sources
HM	Heavy metals	Lidköping WWTP	Swedish average
Marine	Recipient	Freshwater	Marine
LCIA method level scenarios			
–	Toxicity method	EF3.1	USEtox 2.12, ReCiPe 2016
–	Eutrophication method	EF3.1	ReCiPe 2016

The sensitivity analysis on the LCA model level involved GHG emission factors, type of energy source, heavy metal content in the effluent and sludge, and choice of recipient. On the LCIA method level, the impacts of method selection were tested for toxicity and eutrophication. For toxicity, the effects of heavy metal loads in the effluent and sludge for scenario EBPR-B were evaluated using EF3.1 (i.e., USEtox 2.1), USEtox 2.12 and ReCiPe 2016; Huijbregts et al. (2017). For freshwater and marine eutrophication, the use of EF3.1 was contrasted to ReCiPe 2016. Details on all scenarios can be found in Table 3.

3. Process simulation

This chapter presents the results of the process modelling (Section 3.1) and the sensitivity analysis (Section 3.2) and discusses them in relation to the existing literature (Section 3.3).

3.1. Results

Select results from the process simulations are presented in Table 4 (detailed in Tables SI–2). As expected, EBPR resulted in a higher energy demand and lower energy production, whereas precipitation chemical consumption was greater for CP. Notably, CH_4 emissions differed—EBPR produced higher CH_4 emissions than CP. More methanogens were observed in the EBPR waterline compared with CP (approximately 0.8% versus 0.2% in relation to the levels of ordinary heterotrophic organisms). The main difference between the configurations was the inclusion of the three anaerobic tanks in the EBPR-configuration (in the main line, the side-stream hydrolysis and the P-release reactor), see Fig. 1. A substantial increase in dissolved CH_4 (by 2700%, from 0.9 to 25.4 t COD/y) was noted over the side-stream hydrolysis reactor in scenario EBPR-B, followed by an additional rise (by 40%, to 34.7 t COD/y) in the anaerobic zones in the main line. The energy that was used for aeration was also higher for EBPR, indicating a greater potential for CH_4 stripping. CH_4 and N_2O emissions were based on the Sumo4N model with default values (i.e., no CH_4 emissions from digesters or dewatering and 100% efficient combustion in the combined heat and power facility); thus, all CH_4 emissions stemmed from the waterline, and total CH_4 emissions may have been underestimated.

3.2. Sensitivity analysis

Select results from the sensitivity analysis are presented in Fig. 3 (detailed in Tables SI–2). CH_4 emissions varied substantially between simulation runs (Fig. 3a), yielding an inverse correlation with energy production. For the EBPR-B scenarios, the lowest emissions were observed in A (< B < Default < C). The same pattern was seen for CPp-B but with smaller differences between scenarios.

The difference in N_2O emissions were minor, with the lowest emissions noted for the EBPR-B_Default (< A < B < C), and the CPp-B_Default (< B < C ≪ A) (Fig. 3b). For scenario CPp-B_A, it appeared that the denitrification was suboptimal, as evidenced by the increased ethanol consumption, higher N_2O emissions and greater levels of nitrate in the effluent.

Energy consumption for the EBPR-B scenarios did not differ considerably (30-MWh difference), nor did they for the CPp-B scenarios (60-MWh difference). In contrast, energy production varied more widely—by 370 MWh for EBPR-B scenarios (highest for A) and 360 MWh for CPp-B (highest for B).

EBPR function, in terms of PAO levels in the final biology tank before the secondary settler, was best in C for EBPR-B scenarios (> Default > B > A) (Fig. 3c), as observed regarding struvite production. However, effluent phosphorus levels and precipitation chemical consumption were lowest for Default (< C < B < A). Notably, levels of effluent phosphorus and consumption of precipitation chemicals followed the same pattern. For CPp-B scenarios, this was also the case, albeit in another order (A < Default < B < C).

Thus, the choice of hydrolysis rate has a substantial influence. The default value in Sumo is high compared with other models (Downing et al., 2023) and might be the cause of the large CH_4 emissions in the Default scenario. Compared with the long retention time that is required for methanogens to grow during anaerobic digestion, it is unlikely that CH_4 formed during these low sludge retention times (SRTs) of 1.25 days, but actual experimental data are lacking. Downing et al. (2023) reported values of dissolved CH_4 in the return activated sludge (RAS) fermenter of <20 mg COD/l for an SRT of ~1.12 days and <40 mg COD/l for an SRT of ~2.25 days. In scenario EBPR-B_Default, 70 mg COD/l of dissolved CH_4 was produced during the side-stream hydrolysis at an SRT of 1.25 days, compared with 4.7 mg COD/l for EBPR-B_A. Some biogas formation was noted by Park et al. (2018) in their evaluation of a bench-scale

Table 4
Output data from dynamic process simulations per functional unit, unless otherwise stated.

Parameter	Unit	EBPR-0	EBPR-B	EBPR-S	CPp-B	CPp-S	CPs-B	CPs-S	CP-0
Chemicals									
Ethanol	m^3	152	151	150	145	143	128	126	127
Ozone	t	30	30	30	30	30	30	30	30
PACl	m^3	–	26	68	172	206	165	118	–
Al/P ratio	*mol/mol*	*0*	*0.17*	*0.43*	*1.08*	*1.30*	*0.75*	*1.04*	*0*
$MgCl_2$	m^3	88	88	88	–	–	–	–	–
NaOH	m^3	136	133	128	–	–	–	–	–
Energy use									
Electricity	MWh	1709	1709	1708	1588	1588	1657	1656	1655
Heat	MWh	572	572	572	489	488	494	492	479
Direct air emissions									
N_2O	t	1.9	1.9	1.9	1.4	1.4	1.5	1.6	1.4
CH_4	t	38.6	38.5	38.4	10.9	10.9	15.5	15.9	16.0
Effluent									
TP	kg	1550	793	595	711	567	870	664	6566
	mg/l	*0.35*	*0.16*	*0.12*	*0.14*	*0.11*	*0.17*	*0.13*	*1.24*
Struvite production									
Struvite	t	58	56	53	–	–	–	–	–
Energy production									
Electricity	MWh	840	842	844	1120	1122	958	956	937
Heat	MWh	960	962	964	1280	1282	1095	1093	1071

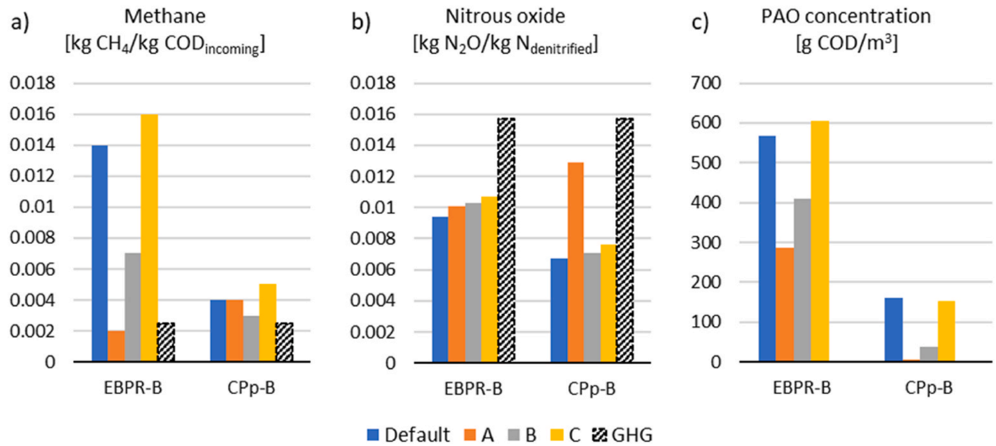

Fig. 3. Results of the process model sensitivity analysis for a) CH_4 emissions, b) N_2O emissions, and c) EBPR function in terms of average PAO concentration in final biology tank before secondary settler. Default, A, B, and C denote the parameter sets. GHG shows emission factors from literature [for CH_4: Gustavsson and Tumlin (2013), for N_2O: Foley et al. (2008, 2010)].

and pilot-scale anaerobic side-stream reactor (ASSR). They found that biogas was produced at an SRT of 2.5 days and at 3 temperatures (21, 37 and 55 °C), although the biogas yield was lower at ambient temperature (10 ml/d or 0.04 ml/g $VSS_{reduced}$).

3.3. Literature outlook

Reported emissions of GHGs from wastewater treatment vary in the literature. In Tables SI–4, some of these emissions are compiled and compared with values from the process model simulations in the current study (see Heimersson et al. (2016) for an earlier, more thorough compilation). In comparing default scenario CH_4 and N_2O emissions with values from Swedish Water's climate calculation tool (Svenskt Vatten, 2023) (Fig. 3a and b), the CH_4 emissions were substantially higher, whereas N_2O emissions were lower. In the calculation tool, the CH_4 emission factor is based on an average of measurements from Swedish WWTPs with covered basins (Gustavsson and Tumlin, 2013), and that for N_2O is derived from literature (Foley et al., 2008, 2010).

Other studies have measured GHG emissions, including Delre et al. (2017), who calculated emissions from five Scandinavian WWTPs. They found a wide range of CH_4 emission factors, from 0.002 to 0.032 kg CH_4/kg $COD_{incoming}$, *including* emissions from sludge line and energy production, where the two latter accounted for a major part of total CH_4 emissions. Daelman et al. (2012) also found that the sludge line generated the largest share of CH_4 emissions (75%) and reported emission levels of 0.0113 (0.0052–0.0120) kg CH_4/kg $COD_{incoming}$ for Dutch conditions. Nevertheless, their value for total CH_4 emissions was lower than the EBPR-B-Default value of 0.014 kg CH_4/kg $COD_{incoming}$ for waterline emissions.

There are several examples of basing GHG emissions in LCA on modelling, such as Rahmberg et al. (2020), who used the ASM1_inCTRL model. However, they did not report their emission factors, and it was unclear whether CH_4 emissions from the waterline were included in their results.

Using literature values for design parameters, Wu et al. (2022) developed carbon footprints for 45 WWTP configurations, based on a statistical model. Among these scenarios, a CP configuration (biological nitrogen removal with ferric chloride precipitation, anaerobic digestion and long-term storage) and an EBPR configuration (Bardenpho with anaerobic digestion and long-term storage) were examined, wherein the direct emissions from the former were greater than the latter, in contrast to our findings. However, waterline CH_4 emissions were not considered. Although Daelman et al. (2012) stated that most CH_4 emissions originated from the sludge line, omitting waterline emissions completely might skew the results, especially for WWTP configurations with anaerobic reactors in the waterline.

Maktabifard et al. (2022) developed carbon footprints for five Polish EBPR plants and four Finnish CP plants. However, they used emission factors from the literature and did not distinguish between plant configurations. In contrast, Fig. 3 notes the differences in CH_4 and N_2O emissions between configurations for all parameter sets. The magnitude of these differences needs to be verified, but it is reasonable to assume that using the same emission factor for EBPR and CP could conceal important differences. Furthermore, using a plant-wide simulation might have the benefit of calculating site-specific emissions more precisely than using standardised literature values, assuming a properly calibrated model.

When comparing emission factors from our study with those in the literature (Tables SI–4), it becomes apparent that the CH_4 emissions from EBPR-B_Default may be overestimated, necessitating further validation of the emissions from actual WWTPs (for both EBPR and CP configurations). In contrast to biological nitrogen removal, biological phosphorus removal is often described differently between simulation software programs, because the governing processes remain incompletely understood (Santos et al., 2020). This deficiency also explains the vast disparities in results between different parameter sets. Consequently, additional research on the biological processes that govern phosphorus removal and methods for modelling them more realistically is needed. Nevertheless, the default parameter set was used in the simulations for the life cycle inventory, but the implications of the selection of emission factors on LCA level were tested in a sensitivity analysis using the values that were suggested in the climate calculation tool mentioned above.

4. Life cycle inventory (LCI)

This section describes the compiled information for the unit processes of the system (Fig. 2). Select process simulation results can be found in Table 4 (detailed in Tables SI–2) and chosen datasets from the databases are listed in Tables SI–5.

4.1. Chemicals

Poly-aluminium chloride (PACl, 7.3% Al) was chosen as the precipitation chemical, given that it will be used at the new Lidköping WWTP. The process model, with its precipitation model designed originally for iron-based coagulants (Hauduc et al., 2015), was thus adjusted accordingly.

Ozone, assumed to be generated on site, was modelled using off-site production and transport of liquid oxygen together with on-site electricity use according to Risch et al. (2022), see Tables SI–6. Ethanol was used as an additional carbon source in the post-denitrification step (MBBR) to reduce effluent nitrogen levels. Cereals were assumed for ethanol production, but fossil ethanol was considered in the sensitivity analysis. For EBPR, additional chemicals (sodium hydroxide and magnesium chloride) were needed for struvite precipitation. For magnesium chloride, no database process was available at the time of the study; instead, it was modelled as the production of sodium chloride (assuming 1 kg $MgCl_2$ = 1 kg NaCl), as in Raymond et al. (2021) and Högstrand et al. (2023). Chemical densities are listed in Tables SI–7.

4.2. Transport

The transport of chemicals was modelled assuming a general distance of 300 km by truck, as in Högstrand et al. (2023). The same distance was assumed for mineral fertilisers, whereas the transport of sludge and struvite was modelled as 100 km by truck, as per Jönsson et al. (2015). All transports were modelled using a database process that was based on distance and weight. Empty returns were not included.

4.3. Energy

For electricity, a Swedish average mix was assumed. For heating, incineration of household waste was considered, but because no dataset for Swedish settings was available, Swiss data were used. On-site combustion of biogas resulted in heat and power being sent to the grid, assuming substitution of the same types of energy as those that were consumed (Swedish electricity mix and Swiss waste incineration). To determine the implications of operating with predominantly fossil-free energy sources, a sensitivity analysis was performed, with fossil energy used and substituted for.

4.4. WWTP air emissions

All carbon in the influent was assumed to be biogenic; thus, direct carbon dioxide emissions were disregarded in the impact assessment, as recommended by IPCC (2019). CH_4 and N_2O emissions were determined using the default parameter values for the apparent fermentation rate, as discussed in Section 3. A sensitivity analysis was performed with emission factors from Swedish Water's climate calculation tool (Svenskt Vatten, 2023). The emission factors are listed in Tables SI–4.

4.5. Effluent

The estimation of the impact of the effluent included total phosphorus, ammonium, nitrate, nitrite, organic nitrogen, COD and heavy metals. All factors except heavy metals were calculated in the process model. The concentration of metals (Cd, Cr, Cu, Hg, Ni, Pb and Zn) in the influent, effluent and sludge was instead based on annual reports from the current WWTP in Lidköping municipality. The current WWTP has a CP configuration; thus, for the modelled EBPR configuration, heavy metal loads in the effluent and sludge were calculated by subtracting the estimated amounts of heavy metals that originated from the precipitation chemical (Tables SI–8). In a sensitivity analysis, average Swedish levels of heavy metals in effluent and sludge (Tables SI–8) were tested.

Organic micropollutants were excluded from this study, due primarily to lack of data. Furthermore, the differences between configurations are expected to be minimal, and the advanced treatment step (ozonation) is anticipated to reduce the toxicity from organic micropollutants to levels well below conventional wastewater treatment. Moreover, several studies have reported a greater impact from heavy metals compared with organic micropollutants in LCAs of wastewater treatment (e.g., Rashid and Liu, 2021; Risch et al., 2022). However, the overall impact of toxicity in the current study may thus be underestimated.

The actual recipient is a river that runs through a town and then enters a larger lake. Most large WWTPs in Sweden, however, release their effluent to the sea (e.g., the brackish Baltic Sea or the marine Øresund, Kattegat and Skagerrak). Lakes are growth-limited primarily by phosphorus, whereas marine environments are usually limited by nitrogen. A sensitivity analysis was performed to determine the implications of the choice of a freshwater versus marine recipient.

4.6. Sludge storage

The digested, dewatered sludge was assumed to be further stabilised by uncovered, outdoor storage for the minimum 6 months, as is common practice in Sweden (Revaq, 2022). N_2O, CH_4 and NH_3 (ammonia) emissions to the air have been included (Tables SI–9).

4.7. Farmland application of sludge and struvite

The following parameters were considered in the application of sludge on farmland: loading and spreading of sludge; direct emissions of NH_3, N_2O and CH_4 to air; direct emissions of NO_3 (nitrate) to water (Tables SI–10); direct emissions of heavy metals to soil (Tables SI–8) and carbon sequestration. For struvite, only emissions of heavy metals to soil were assumed using values from Remy and Jossa (2015) (Tables SI–8). Leakage of phosphorus from agricultural application of sludge, struvite or mineral fertilisers was excluded, because it relates more to soil conditions than type of fertiliser (Johnsson et al., 2019), and any differences between scenarios would likely be minor.

The activity of loading and spreading sludge was based on total sludge weight. The total weight was calculated from the total sludge volume and TS content (4380 m^3/year, 19–21% TS, rendered by the process model) and the density of water and dry sludge solids (assumed to be 1.0 and 1.4 t/m^3, respectively). The application of struvite on farmland was modelled based on area, which in turn was estimated based on the maximum allowed application of phosphorus of 22 kg P/ha per year in Sweden (Jordbruksverket, n.d.).

Inclusion of carbon sequestration is recommended (Corominas et al., 2020) and is modelled in the current study as avoided emissions of carbon dioxide (CO_2) (Tables SI–10). Furthermore, the increased soil carbon content from the application of sludge may have other positive effects, such as greater water holding capacity. This property renders the soil more resistant to droughts and may enhance the yield compared with the application of fertilisers without organic matter, such as struvite and mineral fertilisers (Hedlund, 2012). This potential benefit was modelled as avoided production of wheat, similar to Heimersson et al. (2017), who assumed that the yield of winter wheat rose with 38 kg/ha per year at a 1% increase in soil carbon (based on Hedlund (2012)) and that sludge application increased the soil carbon by 0.9% (based on Börjesson et al. (2012)), boosting the estimated yield with 35 kg winter wheat/ha per year. The estimation of area for wheat production was based on the phosphorus content in the sludge and the maximum allowed level of 22 kg P/ha per year (Jordbruksverket, n.d.).

4.8. Substitution of mineral fertilisers

The impacts of avoided production, transport and application of mineral fertilisers were included. Spreading onto farmland was modelled as for struvite—i.e., based on area and phosphorus content. Phosphorus in the sludge and struvite was assumed to replace triple superphosphate (TSP, 45% P_2O_5), whereas nitrogen replaced ammonium nitrate (AN, 33.5%). The degree of substitution was assumed to be 100% for phosphorus in struvite and EBPR sludge (Gerhardt et al., 2015; Remy and Jossa, 2015), 70% for phosphorus in CP sludge (Gerhardt et al., 2015; Heimersson et al., 2016) and 100% for nitrogen in struvite (Remy and Jossa, 2015) and based on the sludge C/N ratio according to Delin et al. (2012) for nitrogen in EBPR and CP sludge (explained further in SI Section 2.1.1).

Avoided use of mineral fertilisers entails avoided direct emissions.

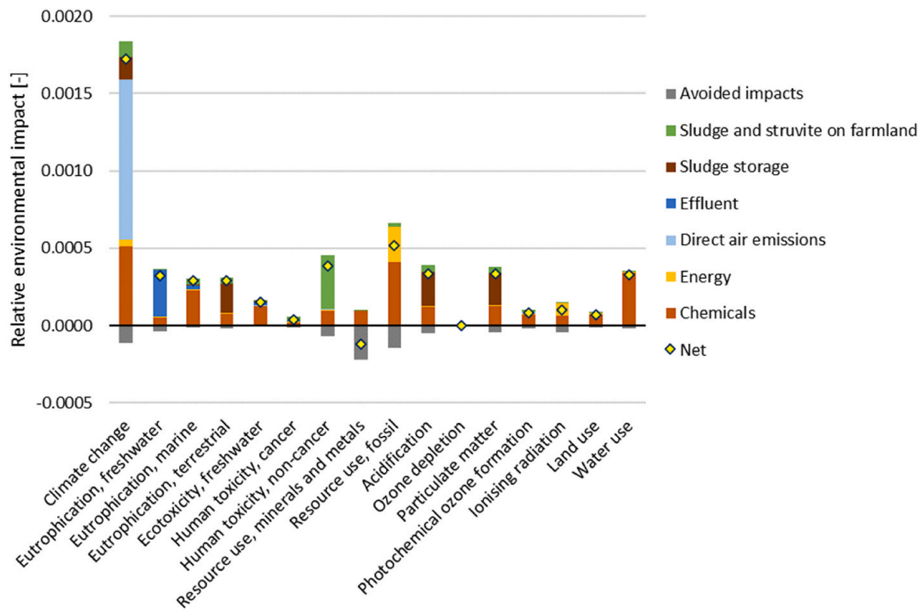

Fig. 4. Normalised and weighted results for scenario EBPR-B, showing the dimensionless, relative impact of each impact category and inventory group. "Chemicals" includes production and transport; "Energy" is electricity and heat production; "Direct air emissions" denotes CH_4 and N_2O emissions from wastewater treatment; "Effluent" reflects direct emissions of nutrients and heavy metals to freshwater; "Sludge storage" is defined as direct emissions during outdoor storage; "Sludge and struvite on farmland" includes transport to and spreading on farmland, as well as direct emissions to air, water, and soil; and "Avoided impacts" includes avoided wheat production, avoided electricity and heat production, avoided production, transport and spreading of mineral fertilisers, as well as avoided direct emissions from fertiliser application.

For phosphorus fertilisers, avoided direct emissions of heavy metals to soil were modelled using the values from Remy and Jossa (2015) (Tables SI–8); phosphorus leakage was excluded, as discussed in Section 4.7. For nitrogen fertilisers, direct emissions to air and water in the form of NH_3, N_2O and NO_3 (Tables SI–11) were included.

5. Life cycle impact assessment (LCIA) results

This section presents and discusses select results; for the full results, the reader is referred to SI Sections 3.1–3.3. As a starting point, Fig. 4 depicts the weighted results of scenario EBPR-B, only, indicating the predominant impact of the climate change category but the substantial contributions from other categories. The same general pattern was seen also for the other scenarios. Selecting a limited number of key categories [as is often done in LCA studies and recommended by Corominas et al. (2020)] might skew the results, at least in this study, increasing the risk for misinterpretation and confounding recommendations for decision-making.

In general, the overall normalised and weighted environmental impacts were rather similar for the different scenarios (Fig. 5). Climate change was the largest individual contributor for all scenarios, except for CP-0 where the impact from freshwater eutrophication was the primary cause of impact. This high level of eutrophication also meant that CP-0 attained the highest weighted environmental impact of all scenarios.

In the following, the results of the individual impact categories are presented and discussed. It should be kept in mind that although the WWTP configurations themselves are standard processes available in full scale around the world, the results are here affected by the Scandinavian climate (i.e. colder water temperature and specific GHG emission factors), the Swedish energy system (i.e. largely fossil-free), the relatively low phosphorus concentrations in influent wastewater (due to restrictions on phosphorus in detergents) as well as strict effluent requirements on phosphorus (due to eutrophication-sensitive recipients).

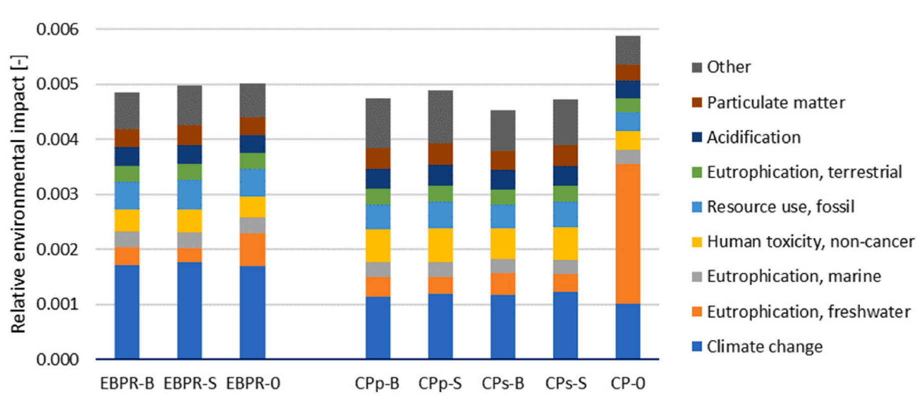

Fig. 5. Normalised and weighted net results for the eight main scenarios, showing the dimensionless, relative impact of each category and scenario. "Other" consists of the categories Ecotoxicity, freshwater; Human toxicity, cancer; Resource use, minerals and metals; Ozone depletion; Photochemical ozone formation; Ionising radiation; Land use and Water use. Scenario labels are defined in Table 2.

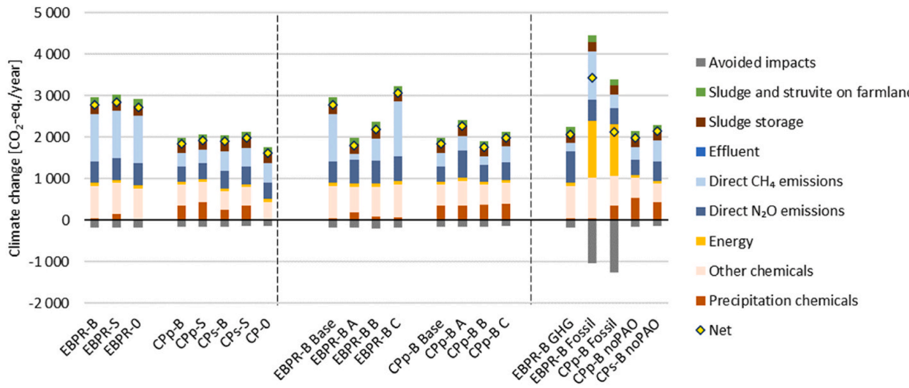

Fig. 6. Characterised results of the category climate change in tonne CO_2-eq./year. Legend is explained in Fig. 3, except that "Chemicals" is divided into "Precipitation chemicals" and "Other chemicals" and that the direct emissions are divided into CH_4 and N_2O. Scenario labels are defined in Table 2 and 3.

5.1. Climate change and fossil resource use

Climate change had the greatest impact in the weighted results, due primarily to it having the highest weighting factor (Tables SI–12). The climate impact for scenario EBPR-B was estimated to be 58 kg CO_2-equivalents per person per year, corresponding to 0.7% of the consumption-based emissions of an average Swedish citizen of 8000 kg CO_2-equivalents per person per year (Naturvårdsverket, n.d.) (detailed in Tables SI–13).

As shown in Fig. 6, direct CH_4 and N_2O emissions from wastewater treatment are the main contributors to this category. The results indicate a difference between the EBPR and CP configurations, wherein the former has higher direct emissions. Because climate change has the highest weighting factor and because the impact from direct emissions is large, these differences are apparent in the overall results (Fig. 5). As discussed in Section 3.1, CH_4 emissions vary in literature and change with fermentation rate (as tested in the process model sensitivity analysis). The sensitivity of the process model to the hydrolysis rate thus has a considerable impact on the LCA-level results (Fig. 6, middle section). Moreover, when emission factors from the literature for CH_4 and N_2O emissions from the waterline were used (scenario EBPR-B_GHG), the impact of climate change was markedly lower compared with scenario EBPR-B.

Notably, the emission factor for CH_4 that was used in the EBPR-B_GHG scenario was based on Foley et al. (2010), who used the process model software BioWin to simulate their scenarios. Similar low emissions of CH_4 can be seen with the EBPR-B_A scenario, in which the parameter set was based on the default parameter values in BioWin. Bisinella de Faria et al. (2015) also used BioWin for their simulation and reported merely a small contribution of CH_4 to the total direct GHG emissions, excluding sludge line emissions, similar to the EBPR-B_GHG and EBPR-B_A cases. Thus, the choice of simulation program may have a large impact on the results. Further research should examine and validate the mechanisms that are related to the apparent fermentation rate, especially the hydrolysis rate.

These results should be tested and verified in full scale to confirm the difference in emissions between configurations. Nevertheless, mitigating direct emissions is important, regardless of configuration, in reaching the target of net zero climate impact in the Swedish wastewater sector (Svenskt Vatten, 2023).

In addition to direct emissions, the impact of chemical production on climate was substantial, due primarily to ethanol production (N_2O emissions and mineral fertilisers in rye production and natural gas for heating in subsequent ethanol production) and precipitation chemicals. For scenario CPp-B, the impact from precipitation chemicals constituted nearly 20% of the total effect. Background data were based on five years of production (2011–2015) in a German production facility, which was assumed to have global relevance (ecoinvent, 2023), resulting in a climate impact of 1.69 kg CO_2-eq./kg product, or 226 g CO_2-eq./mol. Johansson and Liljenroth (2023) recently reported carbon footprints for several precipitation chemicals, wherein aluminium chloride-based compounds ranged from 78 to 102 g CO_2-eq./mol Al, indicating that the values that were used in the current study may be outdated and overestimated. Furthermore, the choice of precipitation chemical influences the results, and selecting an iron-based coagulant may decrease the carbon footprint further (Johansson and Liljenroth, 2023), although this aspect was not evaluated here.

It is, however, possible that the *consumption* of precipitation chemicals was underestimated, because PAOs were also present in the CP model, thus affecting the chemical consumption. In one of the process model-level sensitivity analyses, the model parameter for PAO growth was set to 0, resulting in a 50–75% increase in chemical consumption (scenarios CPp-B_noPAO and CPs-B_noPAO; Fig. 6). This meant that the Al/P ratio[1] rose from 1.08 to 1.59 mol/mol for scenario CPp-B, with and without PAO bacteria, respectively (Tables SI–2).

Fossil resource use constituted the second largest bar in the weighted results for EBPR-B (Fig. 4), with electricity use contributing substantially, although the Swedish electricity mix is largely fossil-free. The choice of energy source also had a profound impact on the climate change results (compare EBPR-B and CPp-B with EBPR-B_Fossil and CPp-B_Fossil; Fig. 6). In this study, largely fossil-free energy sources were selected as the base case, resulting in little climate impact from energy consumption and minor benefits from biogas production. When fossil-based energy sources were chosen instead, potential impacts and benefits related to substitutions were considerably affected. Similar effects have been reported, such as in Rahmberg et al. (2020), who found that climate impact was greater for EBPR than CP when energy sources were fossil-based, due to the higher energy demand and lower energy production for the former.

The types of energy sources that are considered for substitution depend on the purpose of the study and the boundaries of the system of interest. It can be difficult to identify the de facto opportunities that exist in a real-life case, which may also vary over time. Furthermore, the WWTP is part of a greater energy system, and decisions made at the WWTP level might impact another actor's capacity to reduce its climate impact (Grewatsch et al., 2023). These considerations are factors that could be examined further in a consequential LCA study. The current study demonstrates the importance of minimising energy consumption and increasing fossil-free energy production to mitigate the need for fossil-based energy.

[1] Al/P-ratio: a measure of the need for metal ions for phosphorus precipitation; for a CP plant, this value usually exceeds 1.5 mol/mol but is typically 0–1 mol/mol for an EBPR plant.

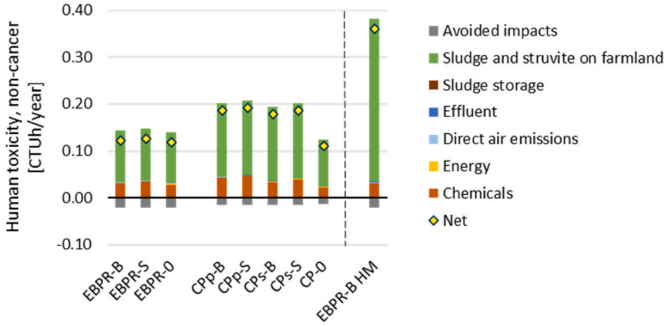

Fig. 7. Characterised results of the category Human toxicity, non-cancer in CTUh/year. Legend is explained in Fig. 3. Scenario labels are defined in Table 2 and 3.

5.2. Human toxicity, non-cancer

Human toxicity (non-cancer) was the second and third most impactful category for the CP and EBPR configurations, respectively (Fig. 5), largely due to the heavy metal content in the sludge spread on farmland. The greater impact of the CP configuration was attributed to the additional heavy metals in the precipitation chemicals. However, the heavy metal content in the influent was also substantial, as indicated by scenario EBPR-B_HM (Fig. 7), in which average Swedish values of heavy metal content in sludge and effluent were applied, doubling the toxicity potential of scenario EBPR-B. This finding highlights the importance of choosing precipitation chemicals wisely or avoiding their use altogether, in addition to continued upstream removal of heavy metals, especially if the sludge is to be used on farmland. Heavy metal content in Swedish wastewaters has decreased markedly over the past several decades (Börjesson and Kätterer, 2018, 2019), but although such levels in the EBPR scenarios were far below the Swedish average (Villner and Myhr, 2022) and although this category had the smallest weighting factor (Tables SI–12), the environmental impact of this category remained substantial (Fig. 5).

Notably, however, the models for toxicity estimation are still not very robust (Sala et al., 2018); thus, the results should be interpreted with caution. The importance of various heavy metals depends on the impact model that is selected. Three impact methods (EF 3.1, USEtox 2.12 and ReCiPe, 2016) were thus contrasted to examine the impact of the heavy metal content in the effluent and sludge in the EBPR-B scenario. For EF 3.1, mercury, lead and zinc in the sludge contributed nearly equally to the environmental impact, whereas for USEtox 2.12 and ReCiPe 2016, zinc in sludge accounted for 85% and 94% of the impact, respectively (Fig. 8, left). For the two other toxicity-related categories (human toxicity, cancer and ecotoxicity), the differences in which heavy metals contributed most to the impact were even greater (Fig. 8, middle and right). Thus, further development of these methods is warranted and until they are sufficiently robust, it would perhaps be wise to use several of them to contrast the results to capture the strengths of each method. As each method highlights different metals, the recommended mitigation measures may vary accordingly.

5.3. Terrestrial eutrophication, acidification and particulate matter formation

Sludge storage was the main contributor (in all scenarios) to terrestrial eutrophication, acidification and particulate matter formation, chiefly due to NH_3 emissions. As discussed, sludge storage also contributed notably to climate impact through CH_4 and N_2O emissions. The future of outdoor, uncovered sludge storage has earlier been discussed and alternatives evaluated (Svanström et al., 2016). The results of the present study further stress that identifying alternatives and implementing new guidelines for sludge management could mitigate a notable part of the total impact of wastewater treatment.

5.4. Aquatic and marine eutrophication

Eutrophication of freshwater systems was primarily affected by phosphorus in the effluent. For the scenarios without chemical addition (EBPR-0 and CP-0) this category was the second largest and largest, respectively, of the weighted results (Fig. 5). For CP-0, the elevated levels of phosphorus in the effluent resulted in the highest total weighted score of all scenarios, although for most other categories, CP-0 had the lowest environmental impact. The difference between EBPR-B and EBPR-0 compared with that between CPp-B and CP-0 is notable, demonstrating the (unsurprisingly but remarkably) greater sensitivity of CP configurations to the availability of precipitation chemicals. Neither EBPR-0 nor CP-0 fulfilled the annual average effluent requirements of 0.2 mg P/l; however, EBPR-B merely required one-sixth of the amount of that of CPp-B to meet these standards.

In discussions over effluent concentrations of phosphorus in relation to chemical use, the impacts of even stricter effluent requirements (EBPR-S, CPp-S and CPs-S) are relevant. To attain lower effluent levels of phosphorus, markedly greater amounts of precipitation chemicals are needed. Comparing EBPR-B with EBPR-S, average annual effluent

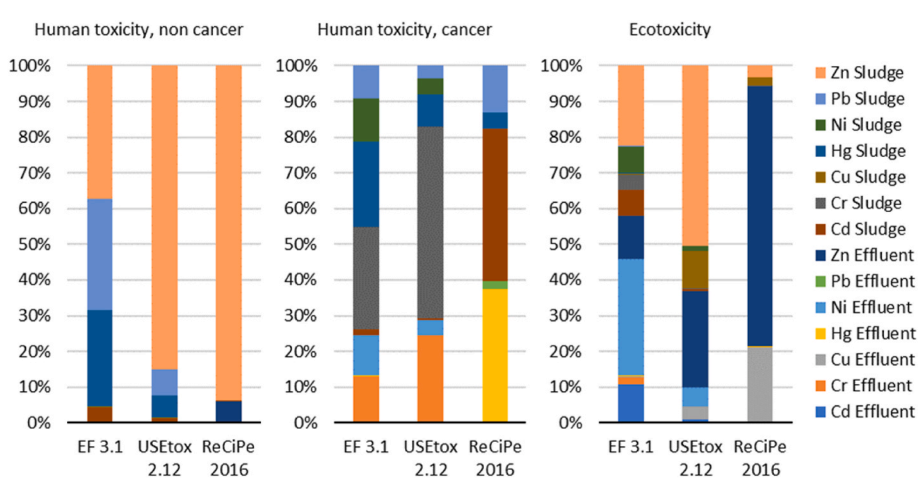

Fig. 8. Contribution analysis of heavy metals in effluent and sludge for the EBPR-B scenario in the 3 toxicity-related impact categories. Three impact methods were compared: EF 3.1, USEtox 2.12 and ReCiPe 2016.

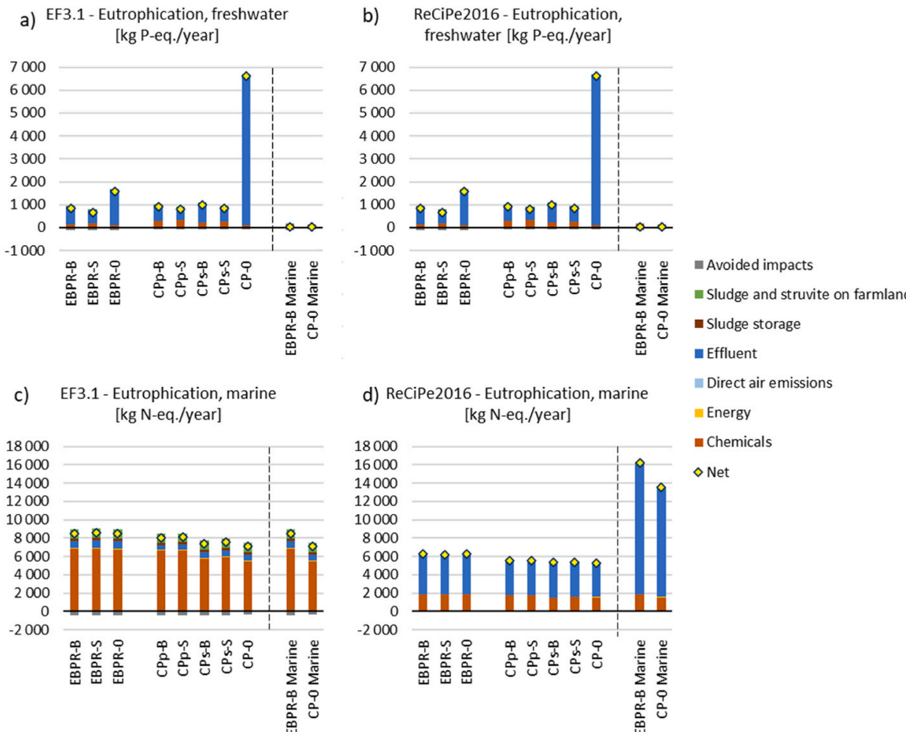

Fig. 9. Characterised results of freshwater (a, b) and marine (c, d) eutrophication, based on EF3.1 and ReCiPe 2016. Legend is explained in Fig. 3. Scenario labels are defined in Table 2 and 3.

phosphorus levels decreased from 0.16 to 0.12 mg P/l, whereas chemical consumption more than doubled (from 26 to 68 m^3/year) (Tables SI–2). Overall, the stricter requirements effected a marginal increase in the environmental impact (Fig. 5).

The choice of recipient had a major impact on the eutrophication results (Fig. 9a–d), because freshwater and marine systems are modelled to be affected only by phosphorus and nitrogen compounds, respectively—i.e., higher levels of effluent phosphorus do not increase eutrophication when the recipient is marine. In the EF3.1 method, eutrophication of freshwater and marine systems is based on an earlier version of ReCiPe, in which marine eutrophication is modelled without differentiating between freshwater or marine compartments (Fazio et al., 2018) (Fig. 9c). In the more recent version, ReCiPe 2016; Huijbregts et al. (2017), the impact of nitrogen emissions depends on whether the compounds are directly or indirectly emitted to marine environments (Fig. 9d). Thus, when changing to a marine recipient, freshwater eutrophication disappears, whereas marine eutrophication remains constant (EF3.1) or doubles in size (ReCiPe, 2016). Consequently, the choice of method may impact the overall results and lead to disparate conclusions. For both methods, we conclude that the availability of precipitation chemicals does not affect marine eutrophication (i.e., for CP plants with marine recipients, chemical shortage may not be an issue). However, two culprits appear with regard to marine eutrophication—the use of ethanol (EF 3.1) and any remaining nitrogen in the effluent (ReCiPe 2016)—leading to diverse recommendations for mitigation depending on selected LCIA-method.

Furthermore, the current level of pollution in the recipient is another important parameter, because it affects the recipient's sensitivity to additional nutrients. Many lakes in southern Sweden, as well as the Baltic Sea, are heavily eutrophicated and thus sensitive to further addition of phosphorus (Havs- och vattenmyndigheten, 2023). To conclude, if the recipient is sensitive to phosphorus emissions and if the availability of precipitation chemicals is deemed to be uncertain, there is reason to evaluate the choice of WWTP configuration and methods for lowering the dependence on chemicals.

6. Conclusions and recommendations

This study performed an environmental assessment of municipal wastewater treatment comparing biological phosphorus removal (EBPR) with chemical precipitation (CP), for standard technology but for a Scandinavian setting in terms of e.g. cold climate and strict water quality standards. Data acquisition through process models proved to be useful, and results are likely more site-specific than when using standard literature values. Our findings suggest that different emission factors for direct emissions should be applied to EBPR and CP, as evidenced by the disparities in greenhouse gas emissions. It is furthermore suggested that waterline emissions should not be excluded for configurations with anaerobic reactors in the waterline. However, uncertainties that are related to apparent fermentation rates seem to result in wide differences between process models, affecting the overall LCA results. Thus, further validation of hydrolysis rates is necessary to obtain more robust estimations of CH_4 emissions. Also, experimental evidence on CH_4 emissions from EBPR plants with side-stream hydrolysis and digester reject water treatment require further research to validate the findings of this study. The inability to calibrate the model to an actual treatment plant makes it difficult to draw firm conclusions; however, this shortcoming often arises in prospective LCA studies, and lessons can be learned also from uncertain results—early indications of the environmental impact of a WWTP before its commissioning are valuable.

The LCA results indicate that the weighted environmental impacts of operating a WWTP with CP or EBPR under normal circumstances are similar. For individual impact categories, however, the results differed. For climate change, EBPR had a greater impact, due to its increased direct air emissions, although verification of CH_4 emissions is needed, as earlier pointed out. For toxicity-related categories, CP performed worse due to increased heavy metal content from the precipitation chemicals. These chemicals, however, were necessary for attaining low effluent limits of phosphorus in the CP and EBPR systems. If the recipient ecosystem is growth-limited with regard to phosphorus and if the supply of precipitation chemicals is deemed uncertain, the possibility of implementing EBPR should be evaluated to reduce the risk of eutrophication. Building on the findings of this study, suggestions for future work include an evaluation of the implications of shifting the bulk of Swedish WWTPs from CP to EBPR. To add to this, an assessment of which treatment plants can shift without massive re-construction should be done, as well as a thorough cost analysis. Furthermore, it is of high relevance to look into the probabilities and characteristics of a chemical shortage. This could lead up to well-informed mitigation and adaptation schemes that would ensure sustainable wastewater treatment in times of global instability and uncertain world development.

CRediT authorship contribution statement

Sofia Högstrand: Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Christoffer Wärff:** Writing – review & editing, Software, Methodology. **Magdalena Svanström:** Writing – review & editing, Supervision, Conceptualization. **Karin Jönsson:** Writing – review & editing, Supervision, Funding acquisition.

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.

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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.2024.144047.

Data availability

Data will be made available on request.

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