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Life cycle assessment of AnMBR technology for urban wastewater treatment: A case study based on a demo-scale AnMBR system

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ABSTRACT

This study aims at assessing the environmental performance of a projected full-scale anaerobic membrane bioreactor (AnMBR) treating urban wastewater (UWW) at ambient temperature. To this aim, data from an AnMBR demonstration plant equipped with commercially available equipment, including industrial hollow fiber and degassing membranes, was used for projecting a full-scale facility. The use of real operation data allows to obtain robust results that contribute to improve the knowledge of the environmental performance of this technology, pointing out its strengths and the challenges that still need to be addressed. Life cycle assessment (LCA) was applied by means of Ecoinvent data base and ReCiPe2016 methodology considering 1 kg of removed COD as functional unit. Additionally, sensitivity and uncertainty analysis were conducted. Energy balance showed AnMBR performing as energy producer (net energy surplus up to −0.688 kWh·kg COD_{rem}^{-1}) and carbon sink (emissions credit up to 0.223 $kgCO_{2eq}\cdot kgCOD_{rem}^{-1}$). Results also showed energy recovery, heavy metals in sludge, dissolved methane in the effluent, and effluent nutrient content as the most important aspects affecting LCA outcome. Construction phase affected some impact categories significantly (e.g., 51–71% in mineral resource scarcity, 18–27% in fossil resource scarcity, 21–28% in water consumption), therefore its exclusion should be carefully evaluated. CHP efficiency, dissolved methane recovery, filtration productivity, membrane scouring, reactor mixing, HRT and SRT appeared most influencing parameters. Finally, actions leading to increase the recovery and valorization of dissolved methane and/or of nutrients through, for instance, fertigation, improve the environmental performance of AnMBR for UWW treatment.

1. Introduction

The study of the environmental performance of water management has been consolidated as a key element to promote innovation and sustainability within this sector [5]. In this context, Life Cycle Assessment (LCA) has been established as a suitable instrument for evaluating the environmental impacts of the water infrastructure (catchment, purification, distribution, sewer systems, wastewater treatment, reclamation, discharge, etc.), providing valuable information on systems design, operation of facilities and policy development [1].

Despite the unquestionable benefits of wastewater treatment (WWT), the conventional processes applied to obtain adequate effluent quality and guarantee the protection of the environment and public health are based on energy-intensive and chemical-dependent technologies [21] with a considerable associated impact. Furthermore, increased pressure on the environment is foreseen since wastewater production and the associated treatment are expected to increase [28] while the authorities are tightening the regulations on waste [4]. Therefore, sewage management strategies should focus on saving energy and chemicals. The progress towards sustainability in the wastewater management sector should also include shifting from the conventional pollutant-removal concept to the resource-recovery paradigm, which offers advantages in reclaimed water, energy and the valuable materials embedded in wastewater, in line with the principles of the Circular Economy [33].

Anaerobic membrane bioreactors (AnMBR) have been proposed as an alternative technology to conventional treatments based on aerobic processes [14,32] and could overcome some of the drawbacks

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mentioned above and contribute to enhance sewage treatment sustainability. While anaerobic microorganisms produce methane from organic matter, which acts as an energy carrier and improves the energy balance [18], they also provide a nutrient-rich effluent that can be fed to nutrient recovery processes (struvite crystallization, membrane contactors, photobioreactors…) or directly reused through fertigation, i.e., direct agriculture reuse of the effluent [16]. On the other hand, membrane technology ensures reduced downstream solids and pathogen concentrations making reclaimed water attractive for other uses (irrigation, aquifer recharge, etc.), while the compactness of membrane modules reduces a facility's footprint and thus favors the retrofitting of existing treatment plants operating in a limited space [39]. Membrane units also decouple sludge retention time (SRT) and hydraulic retention time (HRT), enabling AnMBRs to operate at low/middle temperatures since the membranes avoid microorganisms wash-out from the reactor [38].

To the best of author's knowledge, few LCAs have been carried out on AnMBRs for urban wastewater treatment, while the comparison of published studies is somewhat challenging due to the diversity of methods (ReCiPe, CML-IA, TRACI, etc.), functional units (m^3 of treated water, kg PO_4^{-3} removed, etc.) and the initial assumptions used. For example, the facility construction phase has traditionally been excluded from the LCA since its impact was considered negligible compared to those related to the operating phase [30]. However, Morera et al. [23] more recently concluded that the environmental loads of the construction phase have been underestimated due to a lack of detail when drawing up the inventories. These authors found a significant impact of the construction phase (over 5%) on most of the categories evaluated, and especially in the case of metal depletion (63%) using the ReCiPe method. Efforts are now being made to enhance LCA studies on wastewater treatment aimed at clarifying these uncertainties and improving their quality and comparability [6].

A wide implementation of AnMBR for UWW is still hampered by the phenomenon of membrane fouling, whose management may imply an increase in membrane surface requirements and energy and chemicals consumption, thus affecting the economic and environmental viability of this technology. Fouling is governed by different interrelated factors, among which the concentration and nature of the mixed liquor solids, membrane scouring intensity, SRT or operating temperature can be mentioned [19]. In this sense, high temperatures diminish mixed liquor viscosity, which boosts filtration but also could make microorganism to release extracellular polymeric substances (EPS) and/or Soluble microbial products (SMP), which increments fouling [20,31]. Fouling-prevention strategies based on gas sparging are effective, but they increase the filtration energy demand. On the other hand, increased SRT might enhance EPS and SMP consumption, although it implies increasing the solids concentration, which hinders the filtration process.

Energy demand, chemical consumption, nutrient release and the methane dissolved in the effluent have been identified as key factors in driving AnMBR environmental performance and determine its comparison with other technologies. Pretel et al. [26] estimated the environmental impact of different aerobic-based wastewater treatment plants (WWTP) (conventional activated sludge-CAS; aerobic membrane bioreactors-AeMBR) and AnMBR both alone and combined with post-treatment for nutrient removal. The results showed that AnMBR alone presented the lowest environmental impact within all the categories evaluated except eutrophication, although eutrophication loads could also be reduced if AnMBR is combined with a CAS-based post-treatment or nutrient recovery processes. Smith et al. [38] also compared AnMBR to various aerobic-based WWTPs (high rate activated sludge with anaerobic digestion-HRAS+AD; CAS and anaerobic digestion-CAS+AD; AeMBR and anaerobic digestion-AeMBR+AD). The findings showed that the methane dissolved in AnMBR effluent accounted for 75% of the global warming impact, which indicates that dissolved methane recovery is a key point to be addressed. The results also demonstrated improved AnMBR environmental performance when treating high-loaded wastewater, being comparable to HRAS+AD and lower than CAS+AD in some impact categories. Cashman et al. [3] compared decentralized AeMBR and AnMBR for different population densities and treatment flows and found that, for all the scales evaluated, AnMBR at 20 °C presented the highest net energy benefits and concluded that AnMBRs operated at lower temperatures in warm climates were a promising technology. Xu et al. [40] found that AnMBR had a lower environmental impact than AeMBR when the system boundary was expanded to include effluent reuse and concomitant nutrient recovery through crop uptake. Without this boundary expansion and in disagreement with Pretel et al. [26], AnMBR showed worse performance but, as commented in regard to Smith et al. [38], this contradiction may be due to differences in influent quality: Pretel et al. [26] evaluated medium/high-loaded wastewater, while Xu et al. [40] considered a low-loaded influent. Finally, Pretel et al. [27] assessed the economic and environmental performance of an AnMBR treating UWW and food waste. The results showed that codigestion allowed a net energy production, improved the economic sustainability (savings up to €0.023 per m^3 of treated water) and reduced overall environmental impacts of AnMBR technology.

The main goal of this study was thus to apply the LCA method to evaluate the environmental performance of a projected full-scale AnMBR system (treatment capacity of 16000 $m^3 \cdot d^{-1}$) for urban wastewater (UWW) treatment. Unlike other studies, mainly based on pilot plants and simulation data, the results were obtained using real data from an AnMBR demonstration plant treating UWW under different operating conditions (see Jiménez-Benítez et al. [14]; Robles et al. [32], (2020b)). Due to the dimensions and equipment of this facility, the results can be regarded as robust information for scaling up AnMBR technology for industrial-scale urban WWTPs.

2. Materials and methods

2.1. AnMBR demo-plant description

The demonstration plant mainly consisted of a 40 m^3 anaerobic reactor (AnR) (34.4 m^3 working volume) connected to three 0.8 m^3 membrane tanks (MT-X) of 0.7 m^3 working volume. Each MT was equipped with an ultrafiltration (UF) hollow-fiber membrane module (PURON® PSH41, 0.03-μm pore size, filtration area of 41 m^2). The plant also included a degassing membranes (DM) system to recover the methane dissolved in the effluent (PermSelect® Silicone Hollow Fiber Membrane; filtration area of 2.1 m^2). The demonstration plant also included different on-line sensors and measurement equipment to monitor and register data on flow rates, liquid levels, pressure (transmembrane, inlet and outlet for blowers, and gas), temperature and methane production and recovery. For further details, see Robles et al. (2020b) and e-supplementary materials.

2.2. AnMBR demo-plant operation

Effluent from the pre-treatment unit of the "Alcázar de San Juan" WWTP (screening and sand removal) was pumped into the anaerobic reactor after further pre-treatment in the rotofilter (RF) and homogenization in the equalization tank (ET). The mixed liquor was continuously recycled through the MTs, where the final effluent was obtained by vacuum filtration. In order to maintain appropriate mixing conditions in the anaerobic reactor, part of the biogas produced was constantly recirculated from the headspace to the bottom of the reactor through coarse bubble diffusers, while another fraction of the biogas produced was pumped to the bottom of the MTs for membrane scouring. For further details, see Robles et al. (2020b).

The demo-plant was operated during 580 days under different operating conditions from which five operating periods have been selected to design and simulate the performance of a projected full-scale AnMBR system to treat the Alcázar de San Juan WWTP flow rate (16000 $m^3 \cdot d^{-1}$). The operating periods have been named by combining the

Table 1
Materials and processes selected for the Life Cycle Inventory.

Construction materials, kg·kg COD_{rem}^{-1}	Concrete *(1 m3 Concrete, normal {RoW}| market for | APOS, S (Ecoinvent 3))*	
	Chromium steel *(1 kg Steel, chromium steel 18/8 {GLO}| market for | APOS, S (Ecoinvent 3))*	
	Cast iron (*1 kg Cast iron {GLO}| market for | APOS, S (Ecoinvent 3)*)	
	Polyester resin (*1 kg Polyester resin, unsaturated {GLO}| market for | APOS, S (Ecoinvent 3)*)	
	Polypropylene (*1 kg Polypropylene, granulate {GLO}| market for | APOS, S (Ecoinvent 3)*)	
	PVC *(1 kg PVC pipe E (Industry data 2.0))*	
	Silicone *(1 kg Silicone product {GLO}| market for | APOS, S (Ecoinvent 3))*	
	Transport *(1 tkm Transport, freight, lorry 16–32 metric ton, EURO6 {GLO}| market for | APOS, S (Ecoinvent 3)*	
Energy consumption, kWh·kg COD_{rem}^{-1}, *(1 kWh Electricity, medium voltage {ES}| market for | APOS, S (Ecoinvent 3))*		
Energy avoided (energy recovery from methane), kWh·kg COD_{rem}^{-1} *(1 kWh Electricity, medium voltage {ES}| market for | APOS, S (Ecoinvent 3))*		
Reagent consumption, kg·kg COD_{rem}^{-1}	Polyelectrolyte *(1 kg Polyacrylamide {GLO}| market for | APOS, S (Ecoinvent 3))*	
	NaOCl *(1 kg Sodium hypochlorite, without water, in 15% solution state {GLO}| market for | APOS, S (Ecoinvent 3))*	
	Citric acid *(1 kg Citric acid {GLO}| market for | APOS, S (Ecoinvent 3))*	
	Transport *(1 tkm Transport, freight, lorry 3.5–7.5 metric ton, EURO6 {GLO}| market for | APOS, S (Ecoinvent 3))*	
Discharge to air, kg·kg COD_{rem}^{-1}	Methane CH_4 *(AnMBR, 1 kg emissions of CH4 to air)*	
Discharge to water, kg·kg COD_{rem}^{-1}	Ammonium NH_4^+ *(AnMBR, 1 kg emissions NH_4^+ to water)*	
	Total phosphorous, PO_4^{-3} *(AnMBR, 1 kg emissions PO_4^{-3} to water)*	
Discharge to soil, kg·kg COD_{rem}^{-1}	Disposal to agriculture *(1 kg Solid manure loading and spreading, by hydraulic loader and spreader {GLO}| market for | APOS, S (Ecoinvent 3))*	N to soil *(AnMBR, 1 kg emissions N to soil through fertigation)*
		PO_4^{-3} to soil *(AnMBR, 1 kg emissions PO_4^{-3} to soil through fertigation)*
		N-based fertilizer (*1 kg Nitrogen fertiliser, as N {GLO}| market for | APOS, S)*
		P-based fertilizer (*1 kg Phosphate fertilizer, as P2O5 {GLO}| market for | APOS, S (Ecoinvent 3))*
		Cd to soil *(AnMBR, 1 kg emissions Cd to soil through sludge spreading)*
		Co to soil *(AnMBR, 1 kg emissions Co to soil through sludge spreading)*
		Cr III to soil *(AnMBR, 1 kg emissions Cr^{+3} to soil through sludge spreading)*
		Cu to soil *(AnMBR, 1 kg emissions Cu to soil through sludge spreading)*
		Zn to soil *(AnMBR, 1 kg emissions Zn to soil through sludge spreading)*
		Ni to soil *(AnMBR, 1 kg emissions Ni to soil through sludge spreading)*
		Pb to soil *(AnMBR, 1 kg emissions Pb to soil through sludge spreading)*
		N_2O to air *(AnMBR, 1 kg emissions N_2O to air through fertigation)*
	Disposal to landfill *(1 kg Municipal solid waste {RoW}| treatment of, sanitary landfill | APOS, S (Ecoinvent 3))*	
	Disposal to incineration *(1 kg Digester sludge {GLO}| treatment of digester sludge, municipal incineration | APOS, S (Ecoinvent 3))*	

Table 1 (*continued*)

Demolition and materials recycling, kg·kg COD_{rem}^{-1}	Transport *(1 tkm Transport, freight, lorry 3.5–7.5 metric ton, EURO6 {GLO}| market for | APOS, S (Ecoinvent 3))*
	Steel and Iron (*1 kg Steel and iron (waste treatment) {GLO}| recycling of steel and iron | APOS, S (Ecoinvent 3))*
	Concrete demolition *(1 kg Waste reinforced concrete {Europe without Switzerland} | treatment of waste reinforced concrete, recycling | APOS, S (Ecoinvent 3))*
	Polyethylene treatment *(1 kg Waste polyethylene/ polypropylene product {Europe without Switzerland} | treatment of waste polyethylene/polypropylene product, collection for final disposal | APOS, S (Ecoinvent 3)*
	PVC recycling *(1 kg PVC (waste treatment) {GLO}| recycling of PVC | APOS, S (Ecoinvent 3))*

letter "H" to indicate the HRT followed by its average value (subindex) and the letter "T" to indicate the temperature followed by its average value (subindex): i) $H_{41}T_{27}$; ii) $H_{25}T_{24}$; iii) $H_{26}T_{19}$; iv) $H_{26}T_{27}$; and v) $H_{41}T_{18}$. Complete characterization of the operating periods evaluated in this study are provided in e-supplementary materials (see Table S1).

The influent wastewater was characterized by a high COD (755 ± 224–1403 ± 532 mg·L^{-1}) and sulfate (125.4 ± 47.6–172.2 ± 28.5 mg SO_4^{-2}-S·L^{-1}) concentrations due to dairy- and wine-industry contributions. Despite the high influent COD concentration, COD removal efficiencies of 90–92% were achieved with concentrations below the European discharge limits (125 mg·L^{-1}). Further details of the biological performance of the demo-plant can be found in Robles et al. [32].

Regarding the filtration process, membrane modules were operated at 20 °C-standardized gross transmembrane fluxes ($J_{20\ gross}$) between 15 and 21 L·m^{-2}·h^{-1} and specific gas demands per volume of permeate (SGD_P) between 6 and 14 Nm^3_{biogas}·$m^{-3}_{permeate}$. These conditions resulted in fouling rates of between 0.3 and 4.7 mbar·d^{-1}. Details on filtration process of the demo-plant can be found in Jiménez-Benítez et al. [15].

2.3. System boundary and functional unit

According to ISO 14040:2006, the LCA framework applied in this study was subdivided into four stages: (1) goal and scope definition phase, (2) inventory analysis phase, (3) impact assessment phase, and (4) interpretation phase. The life cycle inventories (LCI) of individual materials and processes were compiled using the *Ecoinvent 3.0* and *Industry data 2.0* databases accessed via SimaPro (PRé Consultants; The Netherlands) (see Table 1). A functional unit of 1 kg of removed COD (kg COD_{rem}) was selected. A total facility life span of 20 years was set.

The following system boundaries were considered:

- The flow diagram considered in this LCA begins with the rotofilter treatment step. Previous screening and sand removal systems are not included since they are common to most treatment systems like CAS.
- Construction (anaerobic reactors, membrane tanks, pipes and equipment), facility operation, and demolition phases (materials disposal and recycling) were considered. Transport of materials, chemicals and sludge (assuming a distance of 20 km for construction and operation and 30 km for demolition) were also incorporated.
- The building work and infrastructure life cycles were set to 20 years [6]. Blower and pump lifespans were estimated from the total working hours provided by the manufacturers (50000 and 75000 h, respectively). A 10-year lifetime was set on other equipment (diffuser, reciprocating engines). UF membrane lifetime was estimated according to the total chlorine contact specified by the manufacturer (500000 h). A 10-year lifespan period was set to DM.
- Neither reuse of nutrients in the effluent for fertigation purposes nor nutrient recovery or removal by other technologies were considered.
- The waste sludge fate was set as follows: 90% to agricultural application, 10% to landfill (in line with data published by the Spanish Ministry of Ecological Transition and Demographic Challenge [22]). Nitrogen and phosphorus sludge content was considered available

Table 2
Relative LCA results for midpoint impact categories in % (The largest impacts are indicated in red and the minor ones in green. Negative values indicate environmental benefits).

Impact category	$H_{41}T_{27}$	$H_{25}T_{24}$	$H_{26}T_{19}$	$H_{26}T_{27}$	$H_{41}T_{18}$
Global warming	-100	-45	1	9	-22
Stratospheric ozone depletion	-100	-64	-92	-65	-63
Ionizing radiation	-100	-61	-43	-39	-70
Ozone formation, human health	-100	-61	-53	-42	-65
Fine particulate matter formation	-100	-62	-52	-43	-66
Ozone formation, terrestrial ecosystems	-100	-61	-53	-42	-65
Terrestrial acidification	-100	-62	-55	-45	-66
Freshwater eutrophication	76	81	96	100	86
Marine eutrophication	51	54	78	100	49
Terrestrial ecotoxicity	-100	-65	-74	-53	-56
Freshwater ecotoxicity	69	62	96	100	87
Marine ecotoxicity (kg 1,4-DCB)	69	62	96	100	86
Human carcinogenic toxicity	65	58	94	100	87
Human non-carcinogenic toxicity	69	62	96	100	87
Land use	-100	-63	-81	-61	-66
Mineral resource scarcity	-100	-77	-90	-67	-45
Fossil resource scarcity	-100	-61	-50	-39	-64
Water consumption	-100	-61	-72	-60	-70

for crops, therefore equivalent avoidance of mineral fertilizer production was contemplated.
- N_2O emissions due to sludge agricultural application were included as 1.18% of total nitrogen content, in accordance with Doka [8].
- Disinfection treatment step was considered implicit in UF performance as per Ferreiro et al. [10], Foglia et al. [11] and Yang et al. [41], among others.
- According to the IPCC Guidelines for National Greenhouse Gas Inventories [9], biogenic CO_2 emissions from wastewater treatment were not taken into account to calculate greenhouse gases (GHG).
- According to Kidd et al. [17], sludge heavy metal content from municipal wastewater treatment was considered, i.e., the industrial contribution is considered to be low (see e-supplementary materials). Displacement of heavy metals in the avoided mineral fertilizers was

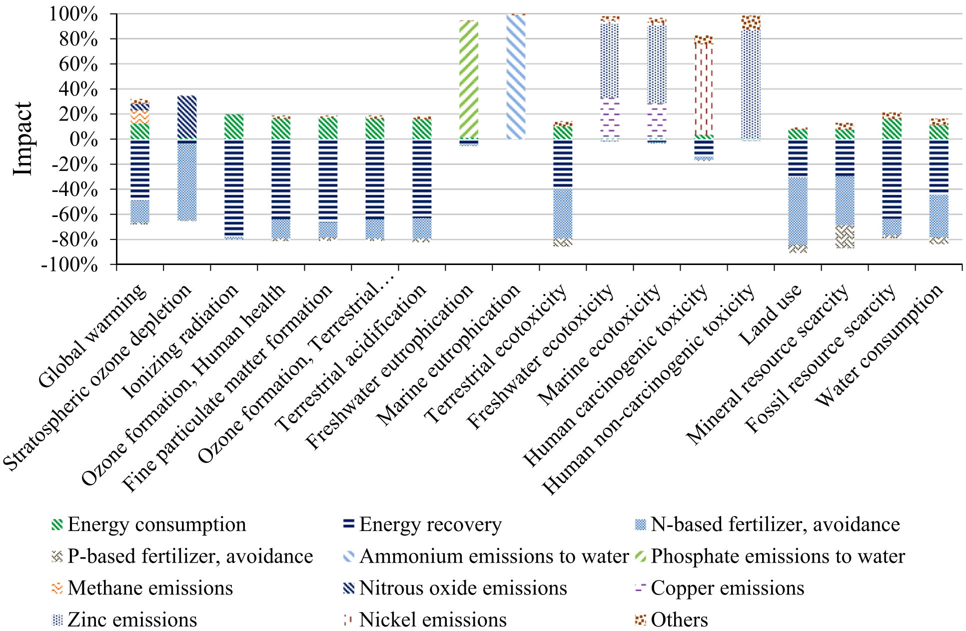

Fig. 1. Example of allocation of loads per impact category for $H_{41}T_{27}$.

not taken into account, due to a lack of information about their content in commercial fertilizer.

- The energy balance in each operating period was calculated as shown in Jiménez-Benítez et al. [15]. The material impact of DM and reciprocating engines for energy recovery were thus included. Moreover, it is considered that the energy recovered is used in its entirety, i.e., if the energy recovered exceeds the plant requirement, it is considered that the energy surplus is discharged into the electrical grid and is consumed outside the facility.
- Dissolved methane recovery by means of DM was included. Recovery percentages in the 80–84% range was considered based on the environmental optimum performance obtained by Sanchis-Perucho et al. [37].
- Fugitive methane emissions due to the stripping of non-captured dissolved methane in the effluent to the atmosphere were considered. These emissions have been estimated as the difference between the methane content in the effluent and that recovered in accordance with the previous point.
- Potential biogas treatments to adapt its quality to CHP requirements (e.g., removal of hydrogen sulfide or siloxanes) have been excluded from the scope of the analysis due to the lack of information on the concentration of these compounds.
- Emissions to air of given compounds (e.g., CO, SO_2, NO_2, non-methane volatile organic compounds) resulting from biogas combustion (through CHP) were excluded due to a lack of information.

LCA was based on the ReCiPe 2016 method and a hierarchist approach. A general discussion based on the 18 midpoint impact categories provided by ReCiPe 2016 method is offered. Moreover, global warming, mineral and fossil resource scarcity, freshwater eutrophication, marine eutrophication and water consumption were selected in this work as key process indicators and a deeper analysis of them was conducted. The endpoint areas (damage to human health; damage to ecosystems; and damage to resource availability) proposed in ReCiPe 2016 methodology were also considered in this study. The results of all midpoint and end-point impact categories can be found in e-supplementary materials.

2.4. Global sensitivity and uncertainty analysis

Sensitivity analysis (SA) and uncertainty analysis (UA) were conducted to identify influential parameters on model outputs and related uncertainty. 2000 model runs (Monte Carlo method) were performed for both SA and UA. In both cases, the Latin hypercube sampling (LHS) method was used to construct the sampling matrix, applying a variability of 5% for the membrane and equipment weight quoted by manufacturers and 10% for the rest of parameters considered (further details can be found in e-supplementary materials). The SA was carried out by means of the Standardized Regression Coefficients (SRC) method applied to all variables and constants considered for modelling the projected full-scale AnMBR system. An absolute value of 0.1 was selected as the threshold value for the standardized regression slope (b_i) for identifying influential input model factors. The UA was conducted to evaluate the propagation of uncertainty in the results, which were assessed by means of i) the 5th and 95th percentiles and ii) the empirical cumulative distribution function (eCDF). The above-mentioned midpoint impact categories (global warming, mineral and fossil resource scarcity, freshwater eutrophication, marine eutrophication and water consumption) were used as model outputs for both SA and UA.

3. Results and discussion

3.1. Energy balance

First of all, it should be noted that in a previous life cycle costing (LCC) study carried out for the same full-scale AnMBR plant [14], the energy balance was conducted considering the economic optimum for the operation of the DM unit. This optimum, calculated based on Sanchis-Perucho et al. [37], was estimated in the 47–55% range of dissolved methane recovery. Sanchis-Perucho et al. [37] also included the estimation of the environmental optimum, which raises the % of dissolved methane recovered to the 80–84% range. The present study, focused on the environmental assessment of the AnMBR technology, has been carried out based on this last recovery percentage range. However, differences in energy balance results are low and it is possible to consult both cases in the e-supplementary materials (Table S2 and Table S3).

Since the energy balance has a crucial influence on many of the impact categories evaluated, main results are summarized in this

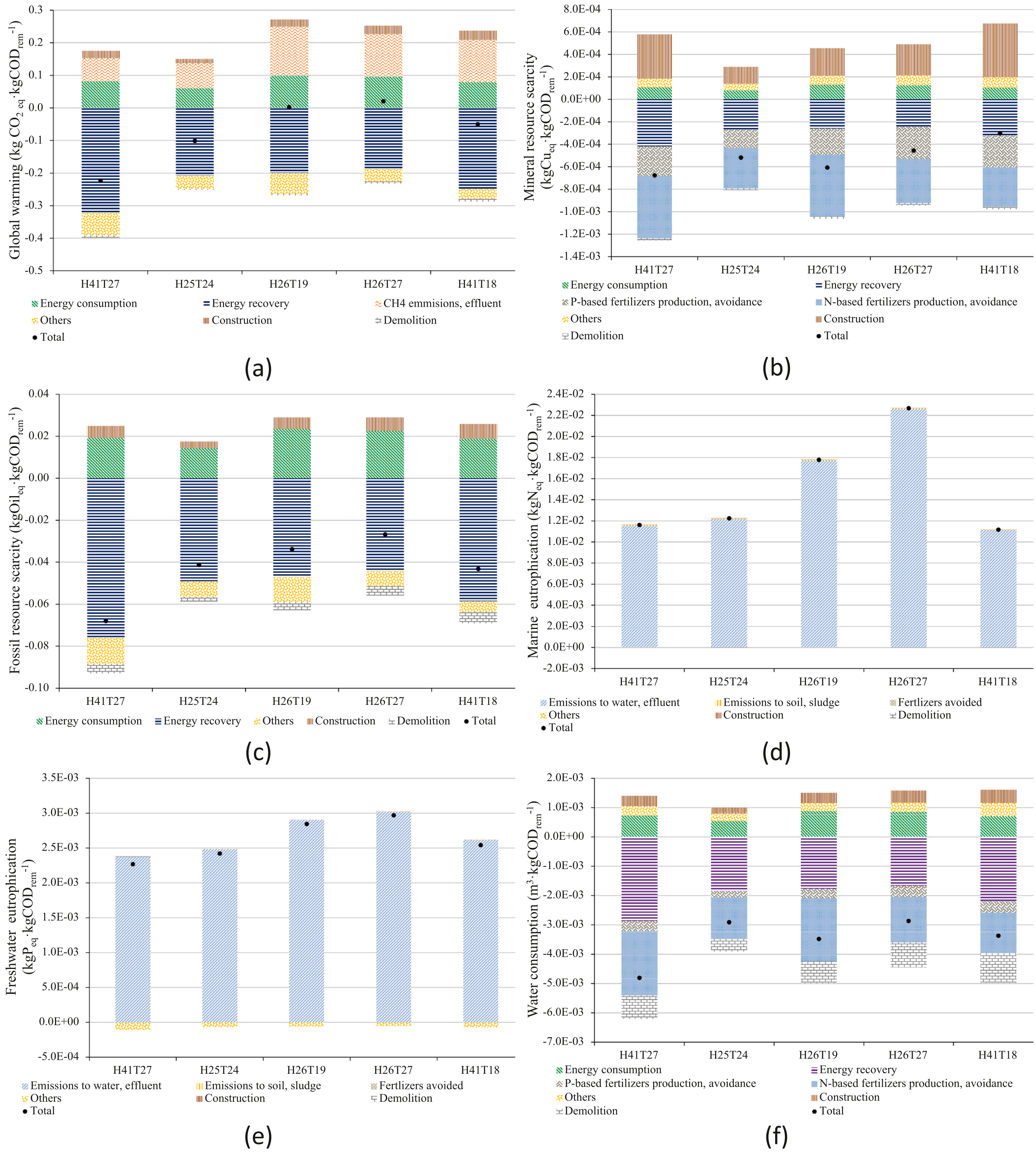

Fig. 2. Environmental impact through: (a) global warming potential, (b) mineral resource scarcity, (c) fossil resource scarcity, (d) marine eutrophication, (e) freshwater eutrophication, and (f) water consumption. Figures include allocations for operating phase jointly with impacts for construction phase and demolition phase.

section. All the evaluated operating periods performed as energy producers, especially $H_{41}T_{27}$, $H_{25}T_{24}$ and $H_{41}T_{18}$ (net energy demands (NED) of − 0.668, − 0.409 and − 0.468 kWh·kg COD_{rem}^{-1}, respectively). $H_{26}T_{27}$ showed the less favorable NED (−0.254 kWh·kg COD_{rem}^{-1}), while $H_{26}T_{19}$ obtained an intermediate value (−0.277 kWh·kg COD_{rem}^{-1}). These results were mainly associated with HRT, which affects reactor mixing requirements; membrane scouring; filtration strategy to minimize membrane fouling; reactor temperature; organic loading rate (OLR) and the COD:SO_4^{-2} ratio, which influences methane production and thus recovered energy. AnMBR, thus, appears as a potential alternative to conventional treatments at moderate temperatures for medium-high strength urban wastewater. For further details, see Table S2 in e-supplementary materials.

3.2. Life cycle assessment

3.2.1. Midpoint categories

Table 2 shows the LCA results for all the midpoint categories included in ReCiPe 2016 for the five evaluated operating periods. Overall, $H_{41}T_{27}$ resulted in the lowest overall environmental impact of the evaluated operating periods in almost all categories (i.e., global warming, stratospheric ozone depletion, ionizing radiation, ozone formation-human health, fine particulate matter formation, ozone formation-terrestrial ecosystem, terrestrial acidification, freshwater eutrophication, terrestrial ecotoxicity, land use, mineral resource scarcity, fossil resource scarcity and water consumption). However, $H_{41}T_{27}$ also was found to have a significant impact among the studied operating periods for freshwater and marine ecotoxicity, and human carcinogenic and non-carcinogenic toxicity. Table 2 also shows that, in general, the worst environmental results were reached in $H_{26}T_{27}$.

Fig. 1 shows the effect of the most influential processes on each impact category for $H_{41}T_{27}$ (similar information related to the rest of the operating periods is provided in e-supplementary material). The main results given in Fig. 1 generalized to all the operating periods can be summarized as follows:

- **Energy recovery:** high organic loading rate jointly with methane production allowed a higher energy recovery in $H_{41}T_{27}$, $H_{25}T_{24}$ and $H_{41}T_{18}$. Improved total energy recovery offset energy consumption, thus reducing the environmental impact of the following categories: global warming, ionizing radiation, ozone formation (human health and terrestrial ecosystem), fine particulate matter formation, terrestrial acidification and ecotoxicity, mineral and fossil resources scarcities, land use and water consumption.
- **Nutrient recovery:** direct reuse of phosphorus and especially the nitrogen embedded in sludge for fertilizing purposes contributed to reducing the environmental loads associated with the avoidance of mineral fertilizer production. This influence was particularly relevant for stratospheric ozone depletion, terrestrial ecotoxicity, land use, mineral resource scarcity and water consumption. Other categories (e.g., global warming, ozone formation (human health and terrestrial ecosystem), fine particulate matter formation, terrestrial acidification, and fossil resource scarcity also reduced their impact due to nutrient recovery. It is important to highlight that Fig. 1 foresees that the environmental benefits of nitrogen recovery in agriculture through sludge spreading would compensate for the drawbacks related to the N_2O emissions derived to air (for the categories of global warming and stratospheric ozone depletion), resulting in positive net balances due to this agricultural practice.
- **Release to the environment of the effluent nutrient content:** the concentration of nutrients in the effluent is the factor that determines the eutrophication potential of the technology. However, since functional unit is defined as 1 kg of removed COD, the environmental load for each operating period was also affected by the influent COD:N and COD:P ratios and the organic matter removal efficiency. As a way of example, $H_{41}T_{27}$ and $H_{26}T_{19}$ obtained a very similar total nitrogen concentrations in the effluent (47.9 $\pm$ 6.2 and 47.3 $\pm$ 4.3 mg N·L^{-1}, respectively) but COD removal in $H_{41}T_{27}$ (equivalent to 19138 kg·d^{-1} for 16000 m^3·d^{-1} of treatment flow) was significantly higher than in $H_{26}T_{19}$ (equivalent to 12428 kg·d^{-1}). Therefore, the latter showed higher marine eutrophication. The eutrophication results demonstrated that nutrient recovery in a downstream post-treatment step (e.g., microalgae cultivation, ion exchange, membrane contactors, etc.) or by fertigation (i.e. nutrient-loaded water reclamation for agricultural purposes) should be considered in order to further enhance the environmental feasibility of AnMBR for UWW treatment [16] and fully leverage resource recovery from the AnMBR effluent. Indeed, reduced environmental loads in other categories influenced by avoiding mineral fertilizer production (e.g., global warming, stratospheric ozone depletion, land use, etc.) could be expected if these nutrients are recovered.
- **Direct GHG emissions due to dissolved methane:** fugitive methane released from the effluent had a significant impact on global warming potential, so that the capture of this gas is a key challenge to be addressed for AnMBR environmental improvement. As way of example, Fig. 1 shows that 40% of the global warming load in $H_{41}T_{27}$ was associated with dissolved methane emissions. Increasing its recovery would reduce this impact while improving the NED.
- **Heavy metals:** the heavy metal content considered in the sludge (see e-supplementary materials) mainly affected freshwater ecotoxicity, marine ecotoxicity, and human toxicity, both carcinogenic and non-carcinogenic. Zinc showed a significant impact on human non-carcinogenic toxicity and freshwater and marine ecotoxicity. The two latter categories were also affected by the copper concentration. Nickel appeared as the main metal influencing human carcinogenic toxicity. Fig. 1 reveals that the benefits of recovering energy and nutrients did not offset the impact of metals associated with human carcinogenic toxicity and the freshwater and marine ecotoxicity impact categories. $H_{26}T_{27}$ obtained the worst performance within these categories due to the combined effect of its moderate dry sludge production (847 t·year^{-1}) and its lower COD removal performance (10815 kg·d^{-1}), while $H_{25}T_{24}$ produced the lowest environmental loads for these categories due to its lower dry sludge production (632 t·year^{-1}) and higher COD removal efficiency (20631 kg·d^{-1}).

As mentioned above, to better analyze the LCA results from the full-scale AnMBR, the following categories were selected as key indicators of the process: global warming potential, mineral and fossil resource scarcity, freshwater and marine eutrophication, and water consumption.

3.2.1.1. Global warming potential. Fig. 2a shows the global warming potential obtained for the evaluated operating periods, considering construction, operating, and demolition phases. The operating phase was the main contributor to this impact category, representing 87–92% of the direct environmental impact (without considering emission avoidance due to energy recovery and others, e.g., avoidance of mineral fertilizer production) for all the evaluated operating periods. The construction phase weight represented between 8% and 13% of the total direct impact, values slightly higher than those reported by Morera et al. [23] (5–10%) for a large CAS WWTP using 1 m^3 of treated water as the functional unit. The main contributors to construction phase are concrete applications (44–47%) followed by PVC of pipes (32–34%).

Most of the direct environmental load in this category was related to fugitive methane emissions due to downstream stripping of dissolved methane in the effluent (40–55%). This percentage is lower to that obtained by Smith et al. [38]: 75% using as functional unit the treatment of 18,950 m^3·d^{-1}). Pretel et al. [25] evaluated the environmental performance of an AnMBR for urban wastewater treatment and also identified methane emissions as a key point. Reduction of AnMBR carbon footprint therefore necessarily involves improving dissolved methane recovery. In turn, recovering the dissolved methane to produce energy would improve the energy balance. $H_{41}T_{27}$, $H_{25}T_{24}$ and $H_{41}T_{18}$ acted as a carbon sinks (−0.223, −0.101 and −0.050 kgCO$_{2eq}$·kgCOD$_{rem}^{-1}$, respectively) when considering the dissolved methane capture efficiencies applied in this study (80–84%), while $H_{26}T_{19}$ and $H_{26}T_{27}$ showed almost carbon neutral performance: 0.003 kgCO$_{2eq}$·kgCOD$_{rem}^{-1}$ and 0.020 kgCO$_{2eq}$·kgCOD$_{rem}^{-1}$. In this regard, Rodriguez-Garcia et al. [34] evaluated the operation of different WWTPs, among others, based on conventional activated sludge obtaining values between 0.206 kgCO$_{2eq}$·kgCOD$_{rem}^{-1}$ and 0.551 kgCO$_{2eq}$·kgCOD$_{rem}^{-1}$ for facilities without nutrient removal processes and discharging to non-sensitive areas. In the case of the more restrictive situation of conventional activated sludge with nutrient removal and discharging to sensitive areas, the same authors obtained emission values between 0.413 kgCO$_{2eq}$·kgCOD$_{rem}^{-1}$ and 0.688 kgCO$_{2eq}$·kgCOD$_{rem}^{-1}$. It can thus be concluded that AnMBR can be regarded as an alternative treatment to reduce GHG emissions in sewage management when dissolved methane in the effluent is captured.

Enhanced dissolved methane capture efficiency is possible from a technical perspective by increasing the DM filtration area, although this would entail an increase in investment costs, which would affect

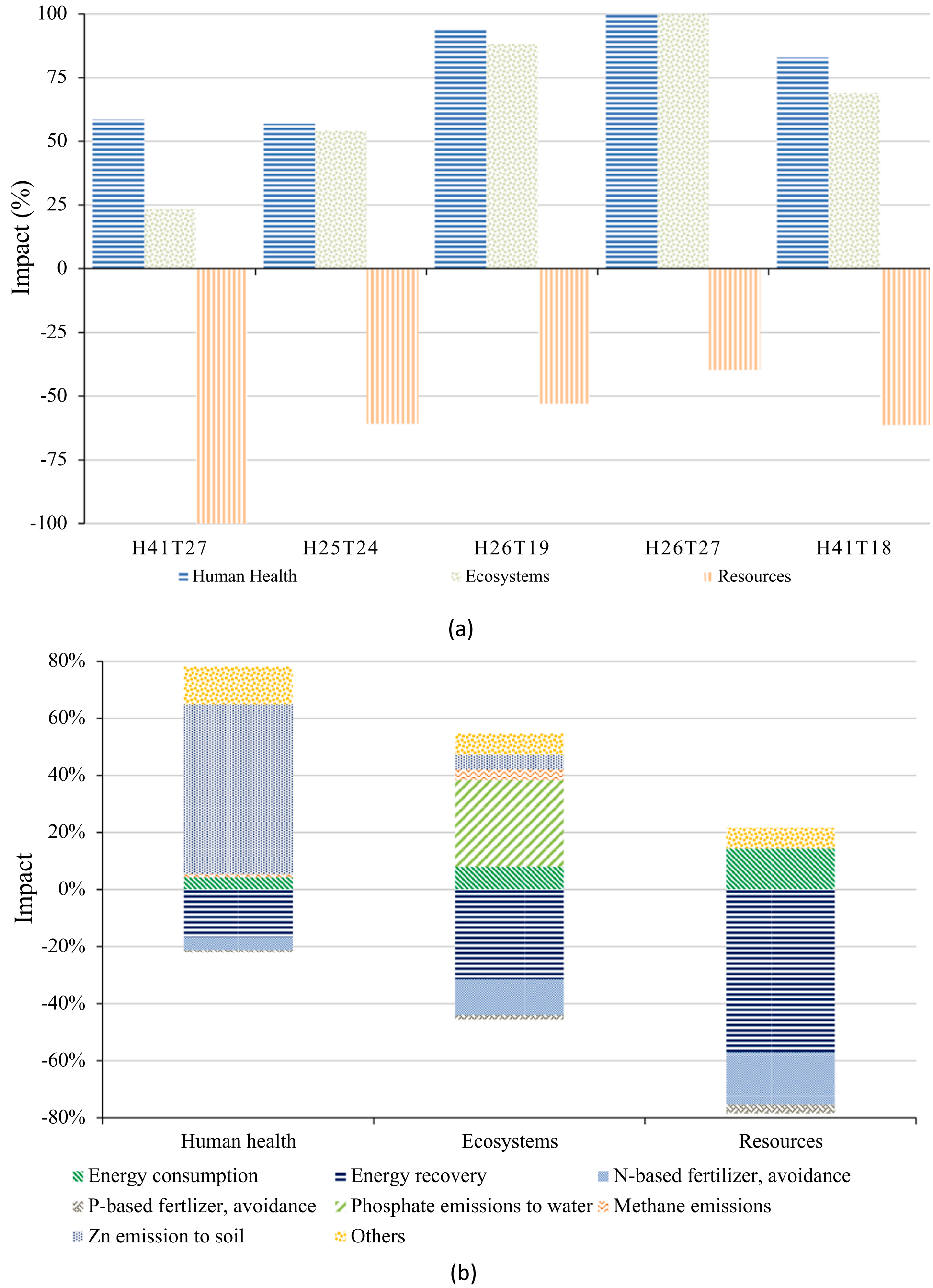

Fig. 3. (a) LCA results for endpoint impact categories. (b) Example of allocation of loads per impact category for $H_{41}T_{27}$.

AnMBR's economic feasibility [36]. Moreover, increasing DM filtration area would increment the environmental impacts associated with the use of DM materials. However, results showed that environmental loads of construction phase were significantly lower than operation phase. Therefore, it is expected that the contribution to the environmental impact associated with an increase in the DM membrane surface would be low. Finally, DM can be combined with additional technologies to prevent dissolved methane release. In this sense, Sanchis-Perucho et al. [35] reached up to 100% of methane capture by adding a degassing tower after DM with low investment cost.

On the other hand, it is important to highlight that global warming potential results strongly depended on the energy mix considered. The larger the fossil fuel burden, the higher the CO_{2eq} emissions associated with energy consumption. For example, according to the International Energy Agency [13], in 2018 the main energy sources in Norway, France and Poland were hydro (≈96%), nuclear (≈72%) and coal (≈78%), respectively. Consequently, the global warming potential impact factor in the Ecoinvent database reflects this diversity of technologies and establishes 0.0289 kg $CO_{2eq}\cdot kWh^{-1}$ for the Norwegian mix, 0.0533 kg $CO_{2eq}\cdot kWh^{-1}$ for the French mix and 1.01 kg $CO_{2eq}\cdot kWh^{-1}$ for the Polish one. The appropriate global warming potential comparisons between the WWT technologies should thus take location into consideration [6].

3.2.1.2. Mineral and fossil resource scarcity. Mineral resource scarcity was mainly influenced by NED and nutrient recovery through sludge spreading due to avoiding production of P-based and especially N-based fertilizers (see Fig. 2b). Hence, there is still a great potential for reducing the environmental loads in this category by reducing resource depletion if effluent nutrient content is also recovered and/or reused for agriculture. $H_{41}T_{27}$ showed the lowest environmental impact in this category among studied operating periods due to the high energy recovery and the combination of high total nitrogen and phosphorus concentration in the mixed liquor (505.8 ± 139.4 mg N·L^{-1} and 96.8 ± 12.3 mg P·L^{-1}), the highest dry sludge production (1049 t·year^{-1}) and its good COD removal performance (19138 kg·d^{-1}). It should also be mentioned that the construction phase composed between 51% and 71% of the direct mineral resource scarcity life cycle impact, when depletion offsets due to energy, nutrient recovery and demolition and recycling are not considered. These results are in line with those obtained by Morera et al. [23], who evaluated the construction phase contribution to mineral resource scarcity as over 60%, but considering 1 m^3 of treated water as functional unit. The major contributors to construction phase load are associated to steel chromium used (77–87%) and concrete applications (12–20%). Environmental WWT assessment of scenarios with reduced energy and material recovery may therefore present a non-negligible construction phase impact, at least for the mineral resource scarcity category.

The environmental impact on fossil resource scarcity followed a similar pattern to the global warming potential (see Fig. 2c), since energy production in Spain still depends strongly on the consumption of fossil fuels [7]. NED is thus the most important factor in the fossil resource scarcity impact category and, accordingly, $H_{41}T_{27}$ presented the lowest environmental load, while $H_{26}T_{27}$ showed the lowest fossil resource preservation. However, this analogous pattern between global warming and fossil resource scarcity may change according to the energy mix selected, as already mentioned. It is also remarkable that both net mineral and fossil resource depletion obtained negative loads in all the operating periods, which indicates that AnMBR contributes to preserving these resources by recycling and valorizing the resources embedded in wastewater and hence contributing to avoid non-renewable resources extraction from conventional sources. These results are in line with Pretel et al. [25], who also obtained negative loads for abiotic depletion category for wastewater treatments based on AnMBR. Conversely, Niero et al. [24] and Canaj et al. [2] obtained positive loads for fossil depletion and fossil and mineral resource scarcity, respectively, assessing the environmental performance of WWTP based on CAS and AD technologies, which confirms the environmental benefits of AnMBR-based treatment over conventional ones.

Even though the operation phase was the major contributor to the fossil resource scarcity input, the construction phase burden forms 18–27% of the direct load, with PVC of pipes (47–51%) and concrete applications (27–28%) as main contributors. Again, this contribution is similar to the results reported by Morera et al. [23], i.e., ≈ 20%, and confirmed that the construction phase could be important in some cases.

3.2.1.3. Freshwater and marine eutrophication. Effluent characterization was the main factor affecting both freshwater and marine eutrophication due to its nitrogen and phosphorus content (see Fig. 3d and e) in line with results obtained by Niero et al. [24], Pretel et al. [25] and Rebello et al. [29], among others. Since freshwater eutrophication is characterized in kg P_{eq}, the trends in this impact category are marked by the effluent phosphorus content and the organic matter removal efficiency in each OP. Results for freshwater eutrophication are in the $2.30{\cdot}10^{-3}$-$2.97{\cdot}10^{-3}$ kg $P_{eq}{\cdot}kgCOD_{rem}^{-1}$ range, which are higher than those reported by Canaj et al. [2] for aerobic-based WWTP ($9.55{\cdot}10^{-4}$ kg $P_{eq}{\cdot}m^{-3}$), since aerobic-based treatments allows reducing the nutrient content in the effluent.

Despite $H_{26}T_{19}$ and $H_{26}T_{27}$ showed the lowest effluent phosphorus content (6.9 ± 2.8 and 6.3 ± 0.5 mg P·L^{-1}, respectively), they had the highest impact within this category due to their comparatively low COD removal (12428 kg·d^{-1} in $H_{26}T_{19}$ and 10815 kg·d^{-1} in $H_{26}T_{27}$). On the other hand, $H_{41}T_{27}$ and $H_{25}T_{24}$ showed the highest phosphorus discharge concentration (8.8 ± 1.9 and 9.9 ± 1.9 mg P·L^{-1}, respectively), but their high COD removal allowed them to reduce their relative impact.

Similarly, although marine eutrophication is characterized in kg N_{eq}, the relative impact of each operating period also depended on the COD removal. Results in this category were in the 1.12·10–2–2.27·10–2 kg N_{eq} $kgCOD_{rem}^{-1}$ range, also higher than the one obtained by Canaj et al. [2], 2.73·10–4 kg N_{eq} m^{-3}. This is coherent with de nitrogen content in its effluent (23.5 mg N·L^{-1}), which is half that in this study (37–54 mg N·L^{-1}). $H_{41}T_{18}$, which combined the lowest effluent N_T concentration (37.0 ± 11.1 mg N·L^{-1}) with a moderate COD removal performance (15376 kg·d^{-1}) showed the lowest marine eutrophication burden. On the other hand, despite presenting similar effluent N_T concentrations, $H_{25}T_{24}$ showed a significantly lower marine eutrophication than $H_{26}T_{27}$, since the COD removal performance of the former (20631 kg·d^{-1}) was double that the obtained in the later (10815 kg·d^{-1}).

The influence of the construction phase on marine eutrophication was found to be negligible, in agreement with Morera et al. [23]. These authors reported a ≈ 5% construction phase freshwater eutrophication burden, unlike the present study, in which its contribution was found to be below 1%. The importance of the N and P sludge concentration on the impact of the agriculture use of biosolids was found to be insignificant.

Finally, it is important to highlight that, although the nutrient concentrations of the effluents from anaerobic treatments present a potential for eutrophication to be considered, it is also true that they represent an opportunity to recover these resources in considerable quantities, which entails obvious environmental and economic benefits. As an example, Jiménez-Benítez et al. [16] carried out an analysis of the combination of AnMBR and fertigation applied to two case studies in Spain and Italy. The results showed that proper irrigation planning allows reducing the discharges of both N and P into the environment by up to 74%, which also means a potential of reducing the need for chemical fertilizers by up to 100%. In addition, significant economic and CO_2 savings were recorded in the AnMBR+fertigation systems compared to the base scenarios made up of a conventional WWTP and a separate irrigation system.

3.2.1.4. Water consumption. Fig. 2f shows the results of the water consumption impact category. The results indicated that reducing NED through biogas and dissolved methane recovery had a positive effect on the environmental loads. On the basis of the results, it could be also stated that taking advantage of nutrient content and thus avoiding P-based and especially N-based fertilizer production also contributed to a relevant reduction of the total impact in this category.

All the evaluated operating periods resulted in negative environmental loads (between $-4.81{\cdot}10^{-3}$ and $-2.87{\cdot}10^{-3}$ $m^3{\cdot}kgCOD_{rem}^{-1}$), which indicates a global water consumption saving. Conversely, Canaj et al. [2] obtained a water consumption of 0.38 m^3 per m^3 of treated water, but considering groundwater withdrawal for irrigation.

According to the results depicted in Fig. 1, energy recovery and avoidance of N-based fertilizer production showed a high influence in water consumption impact. Therefore, $H_{26}T_{27}$ comparatively had the worst performance in this impact category since it showed the lowest energy recovery potential and relatively low avoidance of mineral fertilizer production. The opposite results were achieved in $H_{41}T_{27}$, which had the highest methane production and fertilizing potential through sludge spreading. It is important to highlight that water reuse was not considered in this study. Including this practice in the LCA boundaries would prevent freshwater extraction, thus improving this impact category, in line with the results obtained by Cashman et al. [3] and Canaj et al. [2].

The construction phase weight in the direct water consumption category made up 21–28% of its total life-cycle impact (e.g., associated with minerals extraction and processing), without considering the load credits provided by energy recovery, avoidance of fertilizer production and material recycling after dismantling. Demolition also had a significant effect and contributed to reducing the water consumption burden in all the operating periods. Excluding the construction and demolition

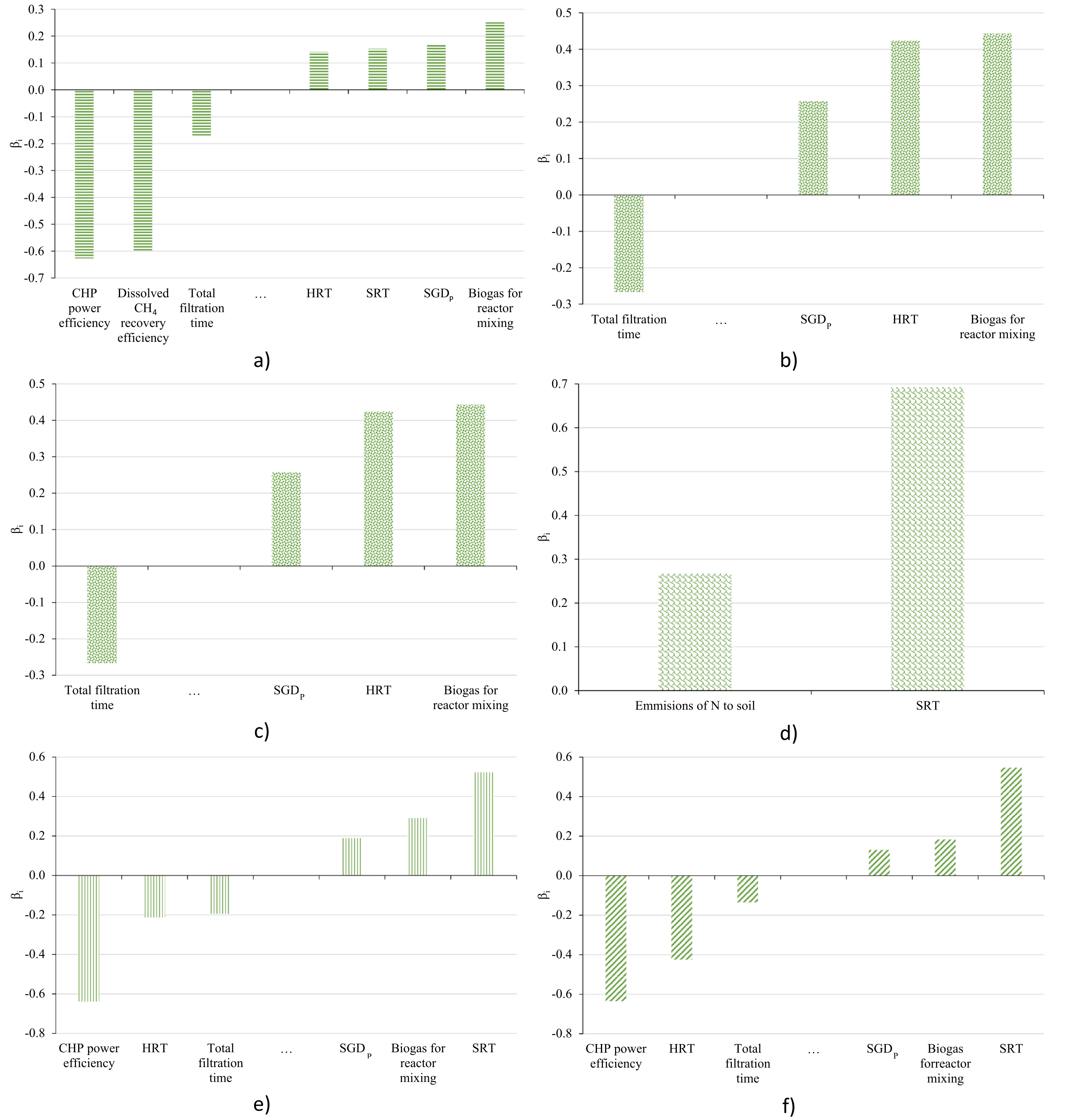

Fig. 4. Sensitivity analysis of $H_{41}T_{27}$ for (a) global warming potential, (b) mineral resource scarcity, (c) fossil resource scarcity, (d) marine eutrophication, (e) freshwater eutrophication, and (f) water consumption. "…" separates positive from negative results.

phases from the LCA boundary for this impact category should be therefore carefully evaluated so as not to affect the results.

3.2.2. *Endpoint categories*

Fig. 3a shows the results for the three endpoint impact categories included in ReCiPe 2016: human health, ecosystems, and resources.

3.2.2.1. Human health. The highest impact on human health was reached in $H_{26}T_{27}$, which is consistent with the midpoint results. In this respect, $H_{26}T_{27}$ resulted in the highest impact in terms of human carcinogenic and non-carcinogenic toxicity. As can be seen in Fig. 3b, the main factor influencing human health was the zinc emissions associated with sludge spreading. Reducing sludge production and its agricultural use or reducing its heavy metal content would address this issue. For example, a reduction of 10% in dry sludge production in $H_{26}T_{27}$ (from 847 t·year^{-1} to 762 t·year^{-1}) would reduce its human health impact by 10% in the operation phase, from $5.070 \cdot 10^{-6}$ to $4.538 \cdot 10^{-6}$ DALY (Disability-Adjusted Life Year). On the other hand, A 10% reduction in zinc concentration in sludge (from 500 mg·kg^{-1} to 450 mg·kg^{-1}) would reduce human health impact to $4.629 \cdot 10^{-6}$ DALY, which represents a reduction of 9%. In Section 3.2.2.3 it will be seen that, unlike in the human health, the reduction in sludge production has a detrimental effect on the resources category due to a lower potential use in agriculture of the nutrients present in the sludge. Finally, AnMBR also contributes to human health by the physical disinfection provided by the UF membranes. Despite this effect is not captured by ReCiPe2016, it is clear that preventing pathogens discharge into water bodies enhances human health protection.

3.2.2.2. Ecosystems. As regards ecosystems, $H_{26}T_{27}$ presented the highest relative impact, while $H_{41}T_{27}$ had the lowest environmental burden. The biggest influence on ecosystems came from phosphorus release, methane emissions associated with effluent discharge, NED, and from zinc released to the soil in a lesser extent. In $H_{26}T_{27}$, the lowest effluent P_T concentration (6.3 ± 0.5 mg P·L^{-1}), the relative moderate dry sludge production (847 t·year^{-1}), and the low dissolved methane concentration in the effluent (2.6 ± 0.5 mg·L^{-1}, mainly due to operating at high temperature: 27 ± 1 °C) were not able to offset the combination of the lowest resulting NED (−0.254 kWh·kg COD_{rem}^{-1}) and the lowest COD removal performance (10815 kg·d^{-1}) of this OP. On the other hand, $H_{41}T_{27}$ presented the minimum impact due to a more favorable NED (−0.668 kWh·kg COD_{rem}^{-1}) and low dissolved methane release (2.6 ± 0.5 mg·L^{-1}, operating temperature of 27 °C), despite its highest dry sludge production (1049 t·year^{-1}) and high effluent P_T concentration (8.8 ± 1.9 mg P·L^{-1}).

3.2.2.3. Resources. The results obtained for the impact on resources were mainly affected by NED and nutrient recovery for agriculture, especially nitrogen and its associated avoidance of mineral fertilizer production (see Fig. 3b). Both items gave to all the operating periods evaluated a net impact reduction in this impact category, as previously mentioned for the former midpoint impact categories (mineral and fossil resource scarcities). Again, $H_{41}T_{27}$ showed the best results due to the resulting favorable NED (−0.668 kWh·kg COD_{rem}^{-1}) and high dry sludge production (1049 t·year^{-1}), suitable for nutrient recovery through sludge spreading. Conversely, $H_{26}T_{27}$ resulted in a less favorable NED (−0.254 kWh·kg COD_{rem}^{-1}), produced less sludge (847 t·year^{-1}) and removed less COD (10815 kg·d^{-1}), with the worst results for ecosystems. It is important to highlight here that sludge production and spreading has opposite effects when analyzing the impact on human health and ecosystems. This is due to the effect of different substances on different areas of protection is considered. In the case of the human health area, the impact is given by the presence of heavy metals in the sludge, therefore, an increase in the amounts destined for agricultural use means increasing the amount of metals deposited in the soil and, consequently, an increased potential for health damage. On the contrary, when the area of resources is considered, the substances of influence are nutrients, so that a greater use of sludge in agriculture reduces the needs of industrial fertilizers and, consequently, a lower consumption of resources.

3.2.3. Sensitivity analysis

By way of example, the results from the SA of $H_{41}T_{27}$ are shown in Fig. 4, providing the main influencing parameters (b_i ≥0.1) for the key midpoint impact categories selected in this work. These factors are CHP electric efficiency; dissolved methane recovery efficiency; total filtration time; SGD_P; rector mixing; HRT; SRT; and emissions of nitrogen to soil.

3.2.3.1. CHP electric and dissolved CH_4 recovery efficiencies. CHP efficiency had a significant influence on global warming potential, fossil resource scarcity, freshwater eutrophication and water consumption, which is consistent with the results given in Fig. 1. A high energy recovery potential reduces CO_2 emissions associated with fossil fuel use and prevents fossil resource depletion. Same effect would produce an increase in dissolved CH_4 recovery and the consequent increase in energy production, jointly with a direct reduction in the global warming potential associated with diminishing CH_4 emissions from the effluent. Moreover, reducing the consumption of electricity produced from fossil fuels also makes it possible to improve the water consumption results, since the energy produced from conventional sources (i.e., coal, gas, nuclear, etc.) requires significant water consumption. The relationship between freshwater eutrophication and energy is not so obvious and could be associated with phosphorus emissions during fossil fuel extraction [12]. Therefore, the improvements associated with energy recovery go firstly through the development of CHP technology that allows increasing its efficiency and, secondly, through the optimization of the operation of methane recovery technologies.

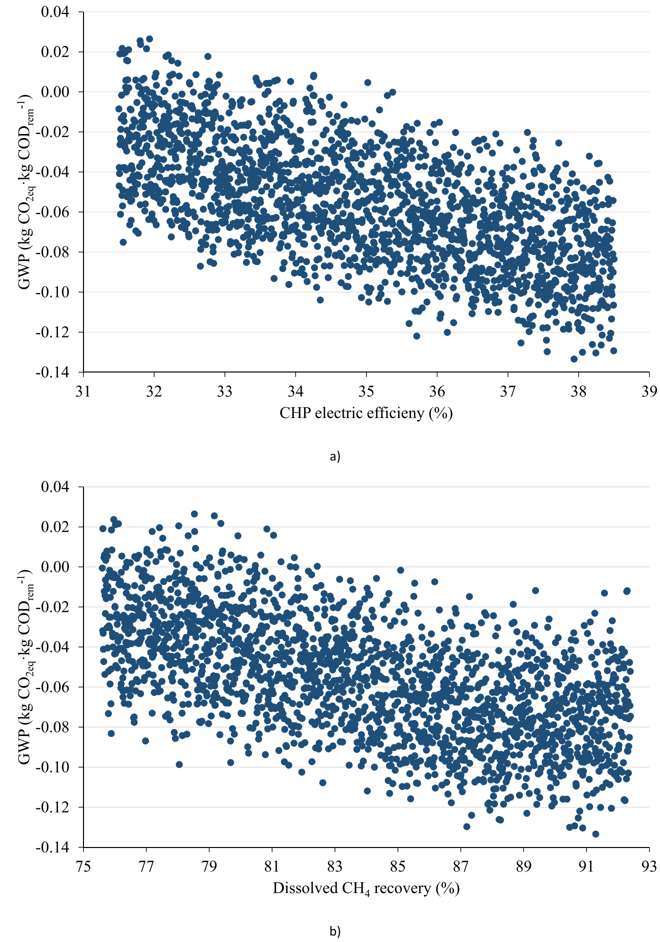

Fig. 5. Monte Carlo simulation results. Effect on global warming potential (GWP) output of: a) CHP electric efficiency and b) dissolved methane recovery efficiency for $H_{41}T_{27}$.

Fig. 5 shows the Monte Carlo simulation results for global warming potential with regard to CHP power efficiency (Fig. 5a) and dissolved methane recovery efficiency (Fig. 5b) in $H_{41}T_{27}$. As already mentioned, higher CHP power efficiencies leads to reduced CO_2 emissions, since the external energy requirements from the grid are lower. The trend of the results shows emissions below − 0.4 kg $kgCO_{2eq}$·$kgCOD_{rem}^{-1}$ with CHP efficiencies of 38.5%. Additionally, dissolved methane recovery over 83% (see Fig. 5b) would also guarantee negative net GHG emissions. Therefore, the combination of AnMBR and CHP shows a promising performance to contribute to reduce the carbon footprint of wastewater treatment.

3.2.3.2. Downtime for physical cleaning. Reducing the downtime for physical cleaning would increase filtration productivity (i.e., total filtration time), while decreasing SGD_P if keeping the same gas sparging intensity for membrane scouring. Under this premise, increasing filtration productivity would reduce energy needs and improve global warming potential, fossil resource scarcity, freshwater eutrophication and water consumption. Therefore, downtime for physical cleaning

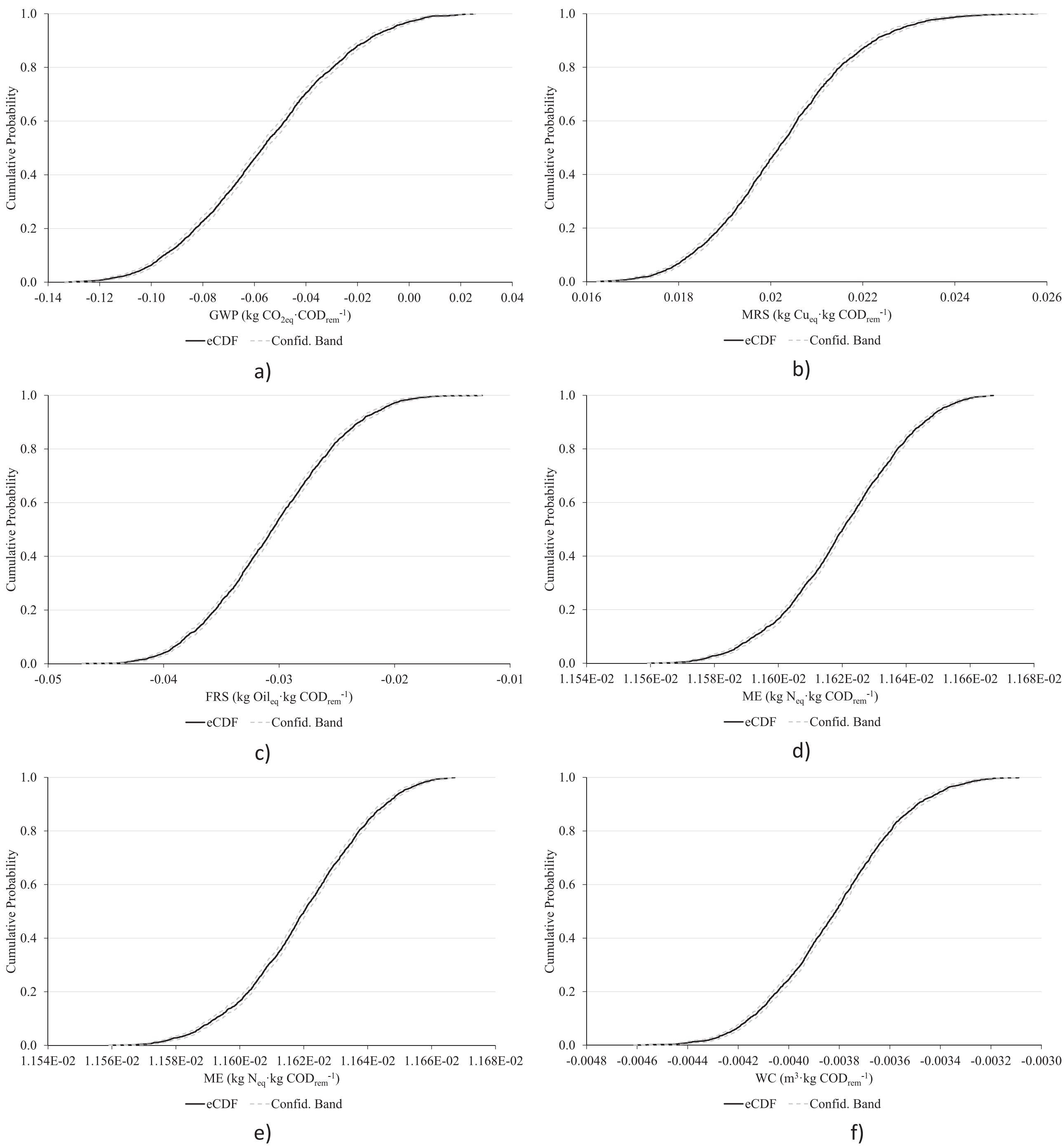

Fig. 6. Uncertainty analysis results for $H_{41}T_{27}$. Empirical cumulative probability for: (a) global warming potential (GWP), (b) mineral resource scarcity (MRS), (c) fossil resource scarcity (FRS), (d) marine eutrophication (ME), (e) freshwater eutrophication (FE), and (f) water consumption (WC).

should be optimized to improve filtration productivity without increasing TMP. Moreover, an enhanced filtration productivity would lead to a reduction in the membrane surface which would also prevent mineral resource scarcity since less material is required in their manufacture.

3.2.3.3. Biogas sparging for membrane scouring and reactor mixing. As commented in Section 3.1, as membrane scouring and reactor mixing are included among the main energy consumption drivers, increasing them also raises the environmental loads associated with the impact categories significantly influenced by the generation and use of energy: global warming potential, fossil and mineral resource scarcity, freshwater eutrophication and water consumption. Reducing energy consumption for membrane scouring requires going through the optimization of the filtration process, e.g., minimizing gas sparging at constant flux or vice versa while guaranteeing low membrane fouling propensities. On the other hand, an optimization of the reactor mixing would also contribute to minimize the environmental impact associated with this process. This optimization would be based on maintaining the minimum gas flow to avoid sedimentation of the solids in the reactor, considering the rheological and hydrodynamic conditions of the system, e.g., solids concentration, viscosity, gas rising, etc.

3.2.3.4. HRT. HRT was identified as an influential input factor on three

Table 3
Uncertainty results for $H_{41}T_{27}$: 5th-95th percentile and 5th-95th range.

	GWP[a] kg CO_{2eq}.kg^{-1} COD_{rem}	MRS[b] kg Cu_{eq}·kg COD_{rem}^{-1}	FRS[c] kg Oil_{eq}·kg COD_{rem}^{-1}	ME[d] kg N_{eq}·kg COD_{rem}^{-1}	FE[e] kg P_{eq}·kg COD_{rem}^{-1}	WC[f] m^3·kg COD_{rem}^{-1}
5th percentile	-0.126	0.017	-0.045	0.012	0.002	-0.004
95th percentile	0.019	0.026	-0.014	0.012	0.002	-0.003
5th-95th range	0.144	0.009	0.031	≈ 0.000	≈ 0.000	0.001

[a] global warming potential;
[b] mineral resource scarcity;
[c] fossil resource scarcity;
[d] marine eutrophication;
[e] freshwater eutrophication;
[f] water consumption

impact categories. HRT was positively correlated with global warming potential, mineral and fossil resource scarcity. Increasing HRT means increasing the reactor volume for a given treatment capacity. Thus, the emissions associated with material consumption (e.g., concrete) and energy consumption for reactor mixing would increase as the HRT is increased, resulting in increased environmental loads within mineral and fossil resource depletion impact categories. Hence, reducing HRT is a suitable option for improving the environmental performance in these three categories. However, reducing HRT for a given SRT would entail an increase in the mixed liquor total suspended solids (MLTSS) concentration in the reactor, which eventually could increment membrane fouling propensity for a given filtration operating mode (e.g., downtime for physical cleaning, gas sparging intensity for membrane scouring, etc.). Although fouling propensity could be controlled by modifying the filtration operating mode when raising MLTSS, it may impact other dimensions (e.g., energy consumption), thus both filtration process configuration and operation and HRT might be considered jointly for process optimization purposes.

3.2.3.5. SRT. SRT was identified to mainly affect methane production, reactor volume, wasting flow rate and nutrients concentration in the effluent. Regarding impact categories, it is positively correlated with global warming potential, fossil resource scarcity, marine and freshwater eutrophication and water consumption. Increasing SRT generally implies an increase in methane production. However, the operation of the AnMBR demo-plant was already carried out at long SRT (70 d), so the availability of additional methanizable organic matter was not significant and the potential increase in energy recovery was low. On the other hand, maintaining the MLTSS concentration by increasing the SRT implies an increase in the reactor volume. As commented above, increasing the reactor volume would result in increased consumption of construction materials and energy requirements for reactor mixing. In addition, the increase in the SRT implies a slight decrease in the waste sludge and, therefore, in the potential for nutrient recovery through biosolids agricultural use, which limits the environmental benefits associated with the displacement of the manufacture of industrial fertilizers. These combined effects explain why, in this case, an increase in SRT leads to an increase in global warming potential and fossil resource scarcity. However, it is important to highlight that in the case of facilities operating at short SRTs, increasing the SRT could imply a significant increase in methane production, even enough for the effect of increasing SRT to be the opposite of that obtained in this study, thus decreasing global warming potential as increasing SRT. Finally, an increase in the SRT means increasing the mineralization of organic nutrients, increasing the concentration of inorganic nitrogen and phosphorus that is discharged into the environment along with the effluent, although it had a negligible effect in this work due to the long SRT applied. Additional releases of nutrients to the environment would increase the environmental load within eutrophication impact categories.

3.2.3.6. Emissions of nitrogen to soil. Emissions of nitrogen to soil from sludge spreading contributes to marine eutrophication (quantified in kg of N_{eq}). On the one hand, A fraction of the nitrogen can percolate through the soil and reach groundwater, through which it is finally transported to the marine environment. On the other hand, runoff can also carry a fraction of nitrogen that will flow towards the sea through surface water. To limit this detrimental effect, the application of agricultural sludge must be carried out following good agricultural practices and adequately balancing the nutrient needs of crops and the amount of sludge to be applied.

3.2.4. Uncertainty analysis

Fig. 6 shows the empirical cumulative function (eCDF) and confidence interval for the six selected midpoint impact categories in $H_{41}T_{27}$ as representative of the study. It can be seen that eCDF follows a standard normal distribution. Table 3 shows that the lowest 5th-95th range was obtained for marine and freshwater eutrophication (both almost 0.0 kg N_{eq}·kg COD_{rem}^{-1} and kg P_{eq}·kg COD_{rem}^{-1}, respectively) followed by water consumption, with $1.0\cdot10^{-3}$ m^3·kg COD_{rem}^{-1}. The highest 5th-95th range was for global warming potential (0.144 kg CO_{2eq}·kg COD_{rem}^{-1}) and fossil resource scarcity (0.031 kg Oil_{eq}·kg COD_{rem}^{-1}), while the mineral resource scarcity showed intermediate values (0.009 kg Cu_{eq}·kg COD_{rem}^{-1}). These results therefore indicate that the global warming potential is subject to greater uncertainty and that it is necessary to strengthen the mechanisms to guarantee that the data used in constructing the LCI that influence this impact category are sufficiently robust.

4. Conclusions

AnMBR performed as energy producer (NED between −0.277 and −0.688 kWh·kg COD_{rem}^{-1}), being HRT, membrane scouring, filtration strategy, reactor temperature, OLR, and the COD:SO_4^{-2} ratio the main influential parameters on NED.

Energy and nutrients recovery were identified as key factors for improving the environmental performance of a full-scale AnMBR for UWW designed using real data from a demo-scale AnMBR plant. Methane and N_2O release to air, and nitrogen and phosphorus release to soil and water bodies are major drivers for environmental impacts, jointly with heavy metal content in sludge. Degassing membranes for dissolved methane capture appear as a key technology to enhance AnMBR environmental performance, while post-treatment technologies for nutrient recovery/removal (e.g., microalgae cultivation, anammox, ion exchange, membrane contactors, etc.) may be necessary for reducing marine and freshwater eutrophication.

Recovering the dissolved methane with efficiencies of 80–84% led AnMBR to act as carbon sink (emissions avoidance up to 0.223 kgCO_{2eq}·kgCOD_{rem}^{-1}). Construction phase contribution to overall LCA was relevant in some operating periods and/or impact categories: 8–13% in global warming potential, 51–71% in mineral resource scarcity, 18–27% in fossil resource scarcity, 21–28% in water consumption, but negligible in marine and freshwater eutrophication: < 1%. Therefore, its exclusion from the system boundary should be carefully evaluated.

Finally, the SA confirms that there is room to improve the environmental performance of the AnMBR technology by increasing the potential for energy recovery (improvement in the efficiency of CHP technologies and optimization of dissolved methane recovery technologies), optimization of filtration, membrane scouring and mixing processes, adjustment of HRT and SRT and use of appropriate agricultural practices.

CRediT authorship contribution statement

A. Jiménez-Benítez: Writing - Original Draft, Investigation, Methodology, Formal analysis, Writing - Review & Editing. **J.R. Vázquez** Investigation, Methodology, Resources, Writing - Review & Editing. **A. Seco:** Investigation, Validation, Writing - Review & Editing, Supervision, Management and coordination responsibility for the research activity planning and execution. **J. Serralta:** Resources, Investigation, Management and coordination responsibility for the research activity planning and execution, Validation, Writing - Review & Editing. **F. Rogalla:** Investigation, Methodology, Resources. **Á. Robles:** Definition, Methodology, Formal analysis, Investigation, Validation, Writing - Review & Editing, Supervision, Management and coordination responsibility for the research activity planning and execution.

Declaration of Competing Interest

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

Data Availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jece.2023.111141.

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