# Environmental impact of submerged anaerobic MBR (SAnMBR) technology used to treat urban wastewater at different temperatures

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**Abstract**

The objective of this study was to assess the environmental impact of a submerged anaerobic MBR (SAnMBR) system in the treatment of urban wastewater at different temperatures: ambient temperature (20 and 33 ºC), and a controlled temperature (33 ºC). To this end, an overall energy balance (OEB) and life cycle assessment (LCA), both based on real process data, were carried out. Four factors were considered in this study: (1) energy consumption during wastewater treatment; (2) energy recovered from biogas capture; (3) potential recovery of nutrients from the final effluent; and (4) sludge disposal. The OEB and LCA showed SAnMBR to be a promising technology for treating urban wastewater at ambient temperature (OEB = 0.19 kWh·m$^{-3}$). LCA results reinforce the importance of maximising the recovery of nutrients (environmental impact in eutrophication can be reduced up to 45%) and dissolved methane (positive environmental impact can be obtained) from SAnMBR effluent.

**Keywords**

Energy balance; global warming potential; life cycle assessment; submerged anaerobic MBR (SAnMBR); environmental impact.

## 1. Introduction

Urban wastewater treatment (WWT) is an energy-intensive activity whose operating energy requirements vary considerably from one WWTP to another depending on the type of influent, treatment technology and required effluent quality. Hence, electricity consumption is a key element in the overall environmental performance of a WWTP (Gallego et al., 2008; Rodriguez-Garcia et al., 2011). Specifically, some studies indicate that bioreactor aeration could account for up to 60% of total WWTP energy consumption (Tchobanoglous, 2003; Hernandez-Sancho et al., 2011). In addition, from a sustainability viewpoint, aerobic urban WWT does not exploit the potential energy contained in the organic matter and the fertiliser value of nutrients.

It is, therefore, particularly important to implement new energy-saving technologies that reduce the overall WWTP carbon footprint and improve environmental sustainability. In recent years there has been increased interest in the feasibility of using submerged anaerobic MBRs (SAnMBRs) to treat urban wastewater. In this respect, SAnMBRs can provide the desired step towards sustainable wastewater treatment (Giménez et al., 2011, Robles et al., 2012 and Lin et al., 2013). This alternative WWT is more sustainable because it transforms wastewater into a renewable source of energy and nutrients, whilst providing a recyclable water resource. Biogas capture is a key operating opportunity of SAnMBR technology which further improves energy balance (Raskin, 2012) and thereby reduces operating costs.

Other aspects of sustainable urban WWT that must be taken into account are the quality and nutrient recovery potential of the effluent, the quantity and quality of the sludge generated, all of which are of vital importance when conducting an environmental assessment of a WWTP (Gallego et al., 2008).

Tools are needed to analyse the likely overall environmental burdens of any wastewater management system. Life cycle assessment (LCA) is a tool for measuring environmental impact that has been widely used in recent decades in the realm of WWT, and is useful for evaluating different WWT technologies (Gallego et al., 2008; Foley et al., 2010; Rodriguez-Garcia et al., 2011; Godin et al., 2012).

The aim of this study was to assess the environmental impact of SAnMBR technology in the treatment of urban wastewater at different temperatures: ambient temperature (20 and 33 ºC), and a controlled temperature (33 ºC) requiring energy input. To this aim, an overall energy balance (OEB) and an LCA, both based on real process data, were carried out. Four factors were considered in this study: (1) energy consumption during urban wastewater treatment; (2) energy recovered from biogas capture; (3) final effluent discharged, considering its nutrient recovery potential; and (4) sludge disposal. In order to obtain reliable results directly comparable to the results from existing full-scale plants, this study was carried out using data from a SAnMBR system featuring industrial-scale, hollow-fibre (HF) membrane units that was operated using effluent from the pre-treatment of the Carraixet WWTP (Valencia, Spain).

## 2. Materials and methods

### *2.1. Scenarios*

The environmental impact of a SAnMBR system to treat urban wastewater (i.e. reducing its organic load to comply with COD effluent standards), by applying OEB and LCA was evaluated. In this respect, since temperature is one of the key operating variables that determine the biological process performance in SAnMBR technology, the following three scenarios at three different operating temperatures were evaluated:

- Scenario 1a: SAnMBR operating at ambient temperature of 20 ºC (warm climate)
- Scenario 1b: SAnMBR operating at ambient temperature of 33 ºC (hot/tropical climate)
- Scenario 2: SAnMBR operating at 33 ºC when the ambient temperature is 20 ºC (controlled temperature requiring energy input).

In addition, within these three scenarios, working at ambient temperatures and controlled temperatures when an energy input is required was also assessed to evaluate the environmental impact of SAnMBRs treating urban wastewater.

The three scenarios were studied using the new version of the WWTP simulation software DESASS (Ferrer et al., 2008) which features the mathematical model BNRM2 (Barat et al., 2012) and a general tool enabling the OEB of the different units in a WWTP to be calculated.

In accordance with recent literature (Lassaux et al., 2007; Rodriguez-Garcia et al., 2011) in order to ensure comparable results, it is necessary to define the functional unit used (e.g. person equivalent, volume of treated wastewater, eutrophication associated with the effluent in terms of kg $PO_4^{3-}$ eq. removed, etc). The functional unit (FU) adopted in this study was the volume of treated wastewater ($m^3$). This approach may have the advantage of being based on physical data.

Four factors were considered when determining the environmental performance of the SAnMBR system being evaluated: (1) the energy consumption of the urban wastewater treatment; (2) energy from biogas capture; (3) the final discharge of effluent (including supernatant from sludge dewatering) taking its nutrient recovery potential into account; and (4) sludge disposal.

The SimaPro 7.3.3 programme was used to quantify the environmental impact of the SAnMBR system being evaluated in the above-mentioned scenarios. SimaPro is widely used in LCA studies and covers a large number of databases (Ecoinvent v.2.2, BUWAL 250, ETH-ESU 96, IDEMAT 2001...) and methodologies (Eco-Indicator 99, CML 2 baseline 2000, EPS 2000, IPCC Global warming potential (GWP) 100a…).

*2.2. System boundaries*

The following system boundaries were considered in this study:

- Wastewater treatment operations and the treated water discharge were considered to be the stages that significantly contribute to the total environmental impact (Lassaux et al., 2007).

- The operating phase was considered to have far more of an impact than the investment phase (Lundie et al., 2004; Lassaux et al., 2007) so the construction phase (including membrane investment cost) was not included in the LCA. Nevertheless, although recent advances in MBR technology have reduced significantly its capital cost, the impact related to this phase should be also considered to assess whether it is important or not.

- Pre-treatment processes (e.g. screening, degritting, and grease removal) were not included in this study because they were assumed to feature in all WWTPs.

- Final effluent was evaluated taking into account its possible re-use for irrigation purposes.

- Sludge transport was not contemplated in the calculations presented in the manuscript.

- The demolition phase was ignored in this study as it was identified to be relatively insignificant in others studies (Emmerson et al., 1995).

- $CO_2$ emissions due to sludge dewatering and biogas capture were not taken into account because $CO_2$ is classified as biogenic according to IPCC guidelines.

- GWP was defined as GWP100 (i.e. GWP with a 100 year horizon). Electricity consumption was considered to be the main contributor to greenhouse gases (Gallego et al., 2008).

- The thermal impact of the final effluent upon natural water courses (when operating at a controlled temperature) was not contemplated in this study.

*2.3. Description of SAnMBR plant*

This study was carried out using data obtained from a SAnMBR system featuring industrial-scale HF membrane units, which was fed with the effluent from a full-scale pre-treatment WWTP (screening, degritter, and grease removal). Table 1a shows the average characteristics of the urban wastewater influent at the SAnMBR plant.

The SAnMBR consists of an anaerobic reactor with a total volume of 1.3 $m^3$ connected to two membrane tanks (MT1 and MT2) each with a total volume of 0.8 $m^3$. Each membrane tank features an ultrafiltration HF membrane commercial system (PURON®, Koch Membrane Systems, 0.05 µm pore size, 30 $m^2$ total filtering area, and outside-in filtration). A rotofilter of 0.5 mm screen size has been installed as pre-treatment system. One equalisation tank (0.3 $m^3$) and one Clean-In-Place (CIP) tank (0.2 $m^3$) are also included as main elements of the pilot plant. In order to control the temperature when necessary, the anaerobic reactor is jacketed and connected to a water heating/cooling system.

The filtration process was studied using experimental data obtained from MT1 (operated whilst continuously recycling the permeate back into the system), whilst the biological process was studied using experimental data obtained from MT2 (operated without recycling the permeate). Hence, different transmembrane fluxes (J) were tested in MT1, without affecting the hydraulic retention time (HRT) of the process.
In addition to conventional membrane operating stages (filtration, relaxation and back-flushing), two additional stages were considered: degasification and ventilation.

Degasification consists of a period of high-flow filtration intended to improve filtration efficiency by removing the accumulated biogas from the top of the dead-end fibres. To capture the bubbles of biogas in the permeate leaving the membrane tank, two degasification vessels (DV) were installed, one between the respective MT and vacuum pump. The funnel-shaped section of conduit makes the biogas accumulate at the top of the DV. During ventilation, permeate is pumped into the membrane tank through the DV instead of through the membrane in order to recover the biogas accumulated in the DV.

Further details of this SAnMBR system can be found in Giménez et al. (2011) and Robles et al. (2012).

*2.4. SAnMBR plant operation*

The plant was operated with an SRT of 70 days at two different operating temperatures: 20 and 33 ºC. The treatment flow (set by MT2) was 2.12 $m^3\ d^{-1}$. The filtration process (studied in MT1) was conducted at sub-critical filtration conditions: the 20 ºC-normalised transmembrane flux ($J_{20}$) was set to 14.5 LMH; the membranes were operated at 13.5 g $L^{-1}$ of MLTS, and the specific gas demand per square meter of membrane area ($SGD_m$) was 0.1 $Nm^3 \cdot h^{-1} \cdot m^{-2}$. The resulting transmembrane pressure (TMP) was approximately 10, -10 and 20 kPa in filtration, back-flushing and degasification respectively. The sludge recycling flow in the anaerobic reactor and membrane tank was 0.4 and 2.1 $m^3 \cdot h^{-1}$, respectively.

*2.5. Analytical monitoring*

The following parameters were analysed according to Standard Methods (2005) in mixed liquor and effluent stream: total solids (TS); sulphate ($SO_4$-S); sulphide (HS-S); nutrients (ammonium ($NH_4$-N) and orthophosphate ($PO_4$-P)); and total chemical oxygen demand ($COD_T$). The methane fraction of the biogas was measured using a gas chromatograph equipped with a Flame Ionization Detector (GC-FID, Thermo Scientific) in accordance with Giménez et al. (2011).

*2.6. Overall energy balance description*

In this study, the SAnMBR plant was considered to be a continuous, steady-state reactor. The resulting OEB in this system is expressed by Equation 1 thus:

$$OEB = W + Q - E_{biogas} \qquad \text{(Equation 1)}$$

where OEB is net energy consumption, consisting of mechanical energy demand ($W$), heat energy ($Q$), and the energy from biogas capture ($E_{biogas}$).

*2.6.1. Mechanical Energy Demands (W)*

The equipment of the SAnMBR plant considered when calculating $W$ consists of the following: one anaerobic reactor feeding pump; one membrane tank sludge feeding pump; one anaerobic reactor sludge mixing pump; one permeate pump; one anaerobic reactor biogas recycling blower; one membrane tank biogas recycling blower; one rotofilter; and one sludge dewatering system.

As proposed by Judd and Judd (2011), the energy consumption of the blowers (adiabatic compression), the general pumps (feeding and recycling) and the permeate pump was calculated by applying Equations 2, 3 and 4, respectively.

$$P_B(\frac{J}{S}) = \frac{(M \cdot R \cdot T_{gas})}{(\alpha - 1) \cdot \eta_{blower}} \left[ \left( \frac{p_2}{p_1} \right)^{\frac{\alpha - 1}{\alpha}} - 1 \right] \quad \text{(Equation 2)}$$

where $P_B$ is the power requirement (adiabatic compression), $M$ (mol·s$^{-1}$) is the molar flow rate of biogas, $R$ (J·mol$^{-1}$·K$^{-1}$) is the gas constant for biogas, $P_1$ (atm) is the absolute inlet pressure, $P_2$ (atm) is the absolute outlet pressure, $T_{gas}$ (K) is the biogas temperature, α is the adiabatic index and $\eta_{blower}$ is the blower efficiency.

$$P_g(\frac{J}{S}) = q_{imp} \cdot \rho_{liquor} \cdot g \cdot \frac{\left\{ \left[ \left( \frac{(L + Leq) \cdot f \cdot V^2}{D \cdot 2 \cdot g} \right) asp. + \left( \frac{(L + Leq) \cdot f \cdot V^2}{D \cdot 2 \cdot g} \right) imp. \right] + [Z_1 - Z_2] \right\}}{\eta_{pump}} \quad \text{(Equation 3)}$$

where $P_g$ is the power requirement by the general pump, considering both pump aspiration and pump impulsion section, calculated from the impulsion volumetric flow rate ($q_{imp.}$ in m$^3$· s$^{-1}$), liquor density ($\rho_{liquor}$ in kg·m$^{-3}$), acceleration of gravity ($g$ in m· s$^{-1}$), pipe length ($L$ in m), equivalent pipe length of accidental pressure drops ($L_{eq}$ in m), the velocity ($V$ in m·s$^{-1}$), friction factor ($f$, dimensionless), diameter ($d$ in m), difference in height ($Z_1$-$Z_2$, in m) and pump efficiency ($\eta_{pump}$).

$$P_{stage(filtration, \deg asification\ or\ back-flushing)}(\frac{J}{S}) = \frac{q_{stage} \cdot TMP_{stage}}{\eta_{pump}} \quad \text{(Equation 4)}$$

where $P_{stage}$ is the power requirement during filtration, degasification or back-flushing calculated from transmembrane pressure ($TMP_{stage}$ in Pa), pump volumetric flow rate ($q_{stage}$ in m$^3$· s$^{-1}$) and pump efficiency ($\eta_{pump}$).

To calculate the net power required by the permeate pump ($P_{permeate}$), the sum of the power consumed in the following four membrane operating stages was considered: filtration ($P_{filtration}$), back-flushing ($P_{back\text{-}flushing}$), degasification ($P_{degasification}$) and ventilation ($P_{ventilation}$). Equation 4 was used to calculate the power in filtration, back-flushing and degasification. Equation 3 was used to calculate the power in ventilation since the fluid does not pass through the membrane.

The energy consumption of the rotofilter was obtained from a catalogue of full-scale equipment (Agua Técnica, 2012). When designing the sludge dewatering, a centrifuge with an average power consumption of 45 $kWh \cdot t^{-1}$ TSS was chosen.

### *2.6.2. Heat Energy Demands (Q)*

In scenarios 1a and 1b, $Q$ was not considered because the plant was operated at ambient temperatures of 20 and 33 ºC, respectively. In scenario 2 (operating at 33 ºC when the ambient temperature is 20 ºC), the intake temperature was increased by heating the system.

$Q$ was assumed to be the sum of the following: the energy required to heat the inflow if necessary ($Q_{REQUIRED}$, Equation 5); the heat dissipated through the walls of the reactor ($Q_{DISSIPATED}$, Equation 6); the heat generated or released in the gas decompression process ($Q_{DECOMPRESSION}$, Equations 7 and 8); and the heat generated or consumed by the biological reactions taking place in the wastewater treatment process ($Q_{ENTHALPY}$, Equation 9).

$$Q_{REQUIRED}\ (\frac{Kcal}{h}) = C_P \cdot q \cdot \rho \cdot \left(T_{fixed} - T_{\inf low}\right) \qquad \text{(Equation 5)}$$

where $C_P$ is the specific heat (1 Kcal·Kg$^{-1}$·K$^{-1}$ for water), $q$ (m$^3$·h$^{-1}$) is the inlet flow rate, $\rho$ (kg·m$^{-3}$) is the density of the sludge and $T_{fixed}$-$T_{inflow}$ (K) is the difference in temperature between the intake temperature and the temperature desired in the reactor.

$$Q_{DISSPATED}\ (\frac{Kcal}{h}) = \Sigma U \cdot S \cdot \Delta T \qquad \text{(Equation 6)}$$

where $U$ (Kcal·h$^{-1}$·m$^{-2}$·K$^{-1}$) is the overall heat transfer coefficient calculated by Equation 7, $S$ (m$^2$) is the surface of the reactor and $\Delta T$ *(K)* is the difference in temperature between the inside and the outside of the reactor.

$$U_{non-buried}(\frac{Kcal}{h \cdot m^2 \cdot K}) = \frac{1}{\Sigma \frac{\delta_{reactor}}{K_{reactor}} + \frac{1}{h_{air}}}; \quad U_{buried}\ (\frac{Kcal}{h \cdot m^2 \cdot K}) = \frac{1}{\Sigma \frac{\delta_{reactor}}{K_{reactor}} + \frac{\delta_{soil}}{K_{soil}}} \qquad \text{(Equation 7)}$$

where $U_{non\text{-}baried}$ and $U_{buried}$ are the heat transfer coefficient in the surface and buried sections of the reactor  respectively, $\delta_{reactor}$ (m) is the reactor thickness, $\delta_{soil}$ (m) is the thickness of the soil in contact with the reactor wall, $k_{reactor}$ (Kcal·h$^{-1}$·m$^{-1}$·K$^{-1}$) is the conductivity of the reactor material, $h_{air}$ (Kcal·h$^{-1}$·m$^{-2}$·K$^{-1}$) is the convective heat transfer coefficient of the air, and $k_{soil}$ (Kcal·h$^{-1}$·m$^{-1}$·K$^{-1}$) is the soil conductivity.

$$Q_{DECOMPRESSION}\ (\frac{Kcal}{h}) = \frac{R \cdot T_4}{\alpha - 1}\left[\left(\frac{P_1}{P_2}\right)^{\frac{\alpha - 1}{\alpha}} - 1\right] \cdot \frac{M}{\Sigma(MW \cdot \%)_i \cdot 4.187} \qquad \text{(Equation 8)}$$

where $P_1$ (atm) is the absolute head space pressure, $P_2$ (atm) is the absolute output blower pressure, $T_4$ (K) is the final temperature of the biogas, $\sum(MW\times\%)_i$ is the sum of the molecular weight of each gaseous component in g·mol$^{-1}$, $M$ is the mass flow rate of biogas in $Kg \cdot h^{-1}$, and $\alpha$ is the adiabatic index.

$$Q_{ENTHALPY} = \Delta H^{\circ}{}_{T}\ (\frac{Kcal}{mol}) = \left(\eta \Delta H^{\circ}{}_{F}\right)_{PRODUCTS} - \left(\eta \Delta H^{\circ}{}_{F}\right)_{REACTANTS} + \int_{298.15}^{T} \sum \eta \cdot C_{P} \quad \text{(Equation 9)}$$

where $\Delta H^{o}_{T}$ is the enthalpy of the reaction at a given temperature (T); $(\eta\ \Delta H^{o}_{F})_{PRODUCTS}$ is the enthalpy of the products; $(\eta\ \Delta H^{o}_{F})_{REACTANTS}$ is the enthalpy of the reactants; ç is the stoichiometric number; and $C_{P}$ ($Kcal \cdot mol^{-1} \cdot K^{-1}$ ) is the specific heat of each component of the reaction.

*2.6.3.* Energy from biogas capture

The CHP technology in this study uses microturbines because they can run on biogas. Although the electrical efficiency of microturbines is usually lower than other CHP systems, they operate adequately because of their simple design (EPA, 2012). Microturbine-based CHP technology has an overall efficiency of around 65.5%. Power and heat efficiency may be about 27.0 and 33.5%, respectively. Equations 10 and 11 show the energy from biogas capture in terms of heat ($Q_{biogas}$) and power ($W_{biogas}$), respectively.

$$Q_{biogas}\ (\frac{Kcal}{h}) = \frac{Vbiogas \cdot \left(\%CH_{4} \cdot CV_{CH4} + \%H_{2} \cdot CV_{H2}\right) \cdot \%_{heat\ effciency\ CHP}}{1000 \cdot 24 \cdot 4.187} \quad \text{(Equation 10)}$$

$$W_{biogas}\ (kW) = \frac{Vbiogas \cdot \left(\%CH_{4} \cdot CV_{CH4} + \%H_{2} \cdot CV_{H2}\right) \cdot \%_{power\ effciency\ CHP}}{1000 \cdot 24 \cdot 3600} \quad \text{(Equation 11)}$$

where $V_{biogas}$ ($l \cdot d^{-1}$) is the biogas volume; $\%CH_{4}$ is the methane percentage; $CV_{CH4}$ ($KJ \cdot m^{-3}$) is the methane calorific power; $\%H_{2}$ is the hydrogen percentage; and $CV_{H2}$

(KJ·$m^{-3}$) is the hydrogen calorific power.

*2.7. Life cycle inventory and life cycle impact assessment*

Life cycle inventory (LCI) methods are described in ISO 14041. The inventory analysis is a list of the volumes of the inflows that a system extracts from the natural environment and the outflows released into it. The energy consumed/generated and final matter discharged by the SAnMBR system were simulated using DESASS. The potential impact of these parameters was then assessed by applying SimaPro and its built-in Ecoinvent database. Simapro was chosen because it provides the most up-to-date and reliable LCI data worldwide (Frischknecht et al., 2004).

Life cycle impact assessment (LCIA) methods are described in ISO 14042. The methodology chosen to assess and evaluate the environmental impact of the system under study is the Centre of Environmental Science (CML) 2 baseline 2000. The impact categories considered in this study are as follows: eutrophication, GWP, acidification, abiotic depletion, ozone layer depletion (ODP), human toxicity, marine aquatic ecotoxicity, fresh water aquatic ecotoxicity, photochemical oxidation and land ecotoxicity.

Environmental loads are calculated by multiplying the amount of emission or consumption by a characterisation factor. Normalised results are calculated by taking into account the characterisation factor of total emissions and the depletion of resources caused by a benchmark system over a given period (in this instance, Europe 1995, the most recent figures available from SimaPro). The normalised value can then be used to calculate the environmental impact of the system under study.

*2.7.1 Electricity consumption data*

The data on the resources used to generate the electricity used to run the SAnMBR system were updated in this study according to data obtained from the Spanish electricity network (REE, 2010).

*2.7.2 Wastewater effluent data*

In this study, the impact of the effluent discharged into natural water courses was assessed after part of its nutrients was used for irrigating farmland (as fertiliser). Since fertiliser can be partially avoided, ammonium sulphate and diammonium phosphate were assumed to be generic N and P sources, which could substitute 50 and 70% respectively of the N and P provided by the effluent (Bengtsson et al., 1997).

*2.7.3 Sludge disposal data*

The stability of the sludge in the three scenarios was evaluated using % VSS (volatile suspended solids) and BVSS (biodegradable volatile suspended solids). The BVSS was calculated theoretically by the WWTP simulation software DESASS which features the mathematical model BNRM2 (Barat et al., 2012). The heavy metal content of the sludge in Spain proposed by Kidd et al. (2007) was adopted in this study.

As the sludge could be used as fertiliser on farmland, the synthetic fertiliser can be partially avoided, using the same percentages of N and P as the wastewater effluent (mentioned in section 2.7.2) according to Bengtsson et al (1997). In addition, nitrogen was emitted: 25.81% in the form of $NH_3$-N and 1.18% in the form of $N_2O$-N (Doka, 2009). On the other hand, heavy precipitation and erosion caused some phosphorus in the sludge spread on land to enter both surface and groundwater by filtering through the soil. The transfer coefficient of phosphorus from sludge into groundwater is 0.57% and into surface water is 2.005% (Doka, 2009).

## 3. Results and discussion

### *3.1. OEB results*

The OEB results of the three operating scenarios of the SAnMBR system evaluated, including energy consumption (mechanical and heat energy) and energy production (heat and power from biogas) (Table 3a). The possible energy obtained by capturing methane dissolved in the effluent was also evaluated (see Table 3b), although it is not included in the OEB results.

#### *3.1.1. Energy consumption and energy from biogas capture*

The mechanical energy was similar in all scenarios (around 0.22 kWh·m$^{-3}$) (see Table 3a). Nevertheless, considering the energy from biogas capture, the net energy requirements were 0.20 kWh·m$^{-3}$ (scenario 1a), 0.18 kWh·m$^{-3}$ (scenario 1b) and 36.71 kWh·m$^{-3}$ (scenario 2), since the high temperature (33 ºC in scenarios 1b and 2) increased the final biogas production. However, a considerable amount of heat energy was needed in the second scenario to maintain a temperature of 33 ºC (131649 kJ·m$^{-3}$, see Table 3). Therefore, increasing the operating temperature from 20 ºC (ambient temperature) to 33 ºC when using SAnMBR technology to treat urban wastewater may be assumed to be unsustainable because of the considerable heat energy needed. On the other hand, the low energy requirements recorded when operating at ambient temperature (scenario 1a and 1b) make SAnMBR a promising sustainable technology from an energy viewpoint. Moreover, when operating at hot/tropical ambient temperatures (e.g. 33 ºC) more biogas was captured than at warm ambient temperatures (e.g. 20 ºC), which slightly reduced overall energy consumption (from 0.20 to 0.18 $kWh \cdot m^{-3}$ in this scenario) when capturing biogas.

### *3.1.2. Impact of physical separation process*

As shown in Table 3a, the most important item contributing to the mechanical energy consumption in the three scenarios was the membrane tank biogas recycling blower, which accounts for some 45% of total mechanical energy requirements (some 0.10 $kWh \cdot m^{-3}$ in absolute terms). According to Lin et al. (2011) the energy consumed by gas scouring accounted for the largest percentage of operating costs, followed by the membrane tank sludge feed pump, which accounted for 43% (approx. 0.09 $kWh \cdot m^{-3}$ in absolute terms). The resulting weighted average distribution of mechanical energy consumption highlights the need to optimise filtration in all operating ranges to improve the feasibility of SAnMBR technology being used to treat urban wastewater. In this regard, operating at low $SGD_P$ (specific gas demand per $m^3$ of treated water) reduces net energy consumption considerably.

### *3.1.3. Impact of energy from capture of methane dissolved in effluent*

As shown in Table 3b, the theoretical amounts of energy from the capture of methane dissolved in effluent were 0.075, 0.083 and 0.152 $kWh \cdot m^{-3}$ in scenarios 1a, 1b and 2, respectively, assuming a methane capture efficiency of 100%.

It is important to emphasise that the energy from the methane dissolved in effluent is not contemplated in this study. If it was, it might reduce the energy consumed in scenarios 1a and 1b considerably (up to 57 and 47%, respectively). This highlights the need to develop technologies for the capture of methane dissolved in effluent not only in order to reduce the environmental impact (i.e. the release of dissolved methane into atmosphere) but also to enhance the OEB of SAnMBR technology.

### *3.1.4. Impact of sulphate content in influent*

Because of the significant sulphate content in the influent in this particular study, an important fraction of COD is consumed by sulphate-reducing bacteria (SRB). To be precise, sulphate content in the influent was approx. 97 mg $SO_4$-S $L^{-1}$, almost all of which was reduced to sulphide (approx. 98%). In this respect, 190 mg COD $L^{-1}$ were theoretically consumed by SRB (calculated using the stoichiometric ratio of kg of sulphate reduced to sulphide per kg of COD degraded).

Therefore, considerably far more power and heat could have been generated if low/non sulphate-loaded wastewaters had been used. If the sulphate content in the influent is considered to be zero, the amount of influent COD transformed into methane increases significantly (up to 37% of the influent COD). Therefore, the resulting methane generated will increase up to 141 $L_{CH4}$·$day^{-1}$ (calculated on the basis of the theoretical methane yield under standard temperature and pressure conditions: 350 $L_{CH4}$ $kg^{-1}$COD). Consequently, in absolute terms, the energy from methane capture (present in biogas and dissolved in the effluent assuming a capture efficiency of 100%) would increase to

0.19 kWh·m$^{-3}$ (power energy) and 592.17 KJ·m$^{-3}$ (heat energy), respectively.

*3.1.5. Comparison with other technologies*

According to Judd and Judd (2011), the full-scale aerobic MBR in Nordkanal (Germany) had a specific energy demand of 0.9 kWh·m$^{-3}$, which is low compared to the consumption (approx. 3.9 kWh·m$^{-3}$) at other full-scale municipal aerobic MBRs (e.g. Immingham Docks MBR WWTP, United Kingdom). On the other hand, conventional activated sludge (CAS) in Schilde (Belgium) consumed 0.19 kWh·m$^{-3}$ (Fenu et al., 2010). For this study, the energy consumption in scenarios 1a and 1b (operating at ambient temperatures of 20 and 33 ºC, respectively) is much lower (0.20 and 0.18 kWh·m$^{-3}$, respectively) than at Nordkanal MBR and similar to Schilde CAS. On the other hand, scenario 2 (operating at 33 ºC when the ambient temperature was 20 ºC) far exceeds the above-mentioned values. Hence, it can be concluded that from an energy perspective, SAnMBR operating at ambient temperatures is a promising sustainable wastewater technology in comparison with other existing urban WWT technologies. Nevertheless, it is important to note that SAnMBR energy demand does not include the energy needed to remove nutrients unlike at Nordkanal MBR, Immingham Docks MBR and Schilde CAS.

*3.2. LCA results*

As mentioned earlier, the SimaPro programme (using Ecoinvent data) was used to assess the potential impact of the SAnMBR system evaluated in this study (energy consumption and production, and matter discharged).

Table 4 shows the LCA results of each impact category (eutrophication, abiotic depletion, etc) in the three scenarios evaluated (1a, 1b and 2). This table is divided into five columns corresponding to the impact of: (1) the four factors of the inventory analysis considered in this study (total impact); (2) energy consumption; (3) energy from biogas capture; (4) sludge disposal; and (5) effluent discharge. The fourth column is divided into two columns to show the impact of the sludge, depending on the percentage considered: (1) for use as fertiliser on farmland (85%); and (2) sent to landfill (15%).

By way of example, Figure 1 shows the LCA of the inventory analysis of each impact category of the final effluents discharged after irrigation, taking into account whether the methane dissolved in the effluent is captured  (Figure 1b) or not (Figure 1a).

The impact of the factors contemplated in the inventory analysis are addressed below (on the basis of the results shown in Table 4 and Figure 1):

### *3.2.1. Impact of the final effluent discharge*

Table 1b shows the average SAnMBR effluent characteristics ($COD_T$, $NH_4$, $PO_4$, $SO_4$, $CH_4$ and $H_2S$). The nutrient content of the effluent shows how temperature affects the rate of hydrolysis: the nutrient content was slightly higher at 33ºC (scenarios 1b and 2). In accordance with Bengtsson et al. (1997), Table 1c shows the amount of nutrients that is not used by plants (i.e. the nutrients in effluents discharged into natural water courses).

The impact of reusing SAnMBR effluent for irrigation is positive because it avoids the direct discharge of nutrients into natural water courses and reduces the use of synthetic fertiliser containing nitrogen (N) and phosphorous (P) (Meneses et al., 2010). Table 4 shows that the effluent discharged after part of its nutrients is used for irrigating farmland, contributes to environmental impact by eutrophication, with environmental loads with normalised values of $153.5 \cdot 10^{-14}$ in scenario 1a, and $154.3 \cdot 10^{-14}$ in scenarios 1b and 2. A significant increase in the environmental impact of eutrophication occurs if the effluent is directly discharged into natural water courses, resulting in environmental loads with normalised values of $336.3 \cdot 10^{-14}$ and $341.8 \cdot 10^{-14}$, respectively. The other impact categories are not affected by the final destination of effluent nutrients (irrigation or discharge).

It is important to highlight that the nutrient discharge has an equal environmental impact in the two scenarios conducted at 33 ºC (scenarios 1b and 2). Scenario 1a (conducted at 20 ºC) has a slightly lower environmental impact than scenarios 2 and 1b, mainly due to the hydrolysis rate. In this respect, the nutrient discharge concentrations (shown in Table 1b and 1c) reveal that temperature seems to have little influence on the hydrolysis rate: similar effluent results were obtained in both scenarios. This is due to operating at 70 days of SRT. This SRT is enough to hydrolyse the main part of the particulate biodegradable organics ($XC_B$): 95% of the $XC_B$ is hydrolysed at 20ºC and 98% of the $XC_B$ is hydrolysed at 33ºC. Therefore, as shown in Table 1b, similar concentrations of $NH_4$ (0.0564, 0.0573 and 0.0573 kg $m^{-3}$) and $PO_4$ (0.0186, 0.0191 and 0.0192 kg $m^{-3}$) were observed in all scenarios (1a, 1b and 2, respectively).

Final effluent nutrient discharge after irrigating farmland has a slightly positive environmental impact (negative values) in all the evaluated impact categories (except eutrophication) due to partially replacing part of the required fertiliser (see Figure 1b). However, when the methane dissolved in the effluent is not captured, some of the impact categories are negatively affected (see Figure 1a or Table 4).

As shown in Figure 1a, different impact categories are affected by discharging the methane dissolved in the effluent, such as human toxicity (resulting in environmental loads with normalised values of $68.0\cdot10^{-14}$, $74.7\cdot10^{-14}$, $71.2\cdot10^{-14}$, in scenarios 1a, 1b and 2, respectively), fresh water aquatic ecotoxicity (resulting in environmental loads with normalised values of $65.4\cdot10^{-14}$, $71.9\cdot10^{-14}$, $68.4\cdot10^{-14}$, in scenarios 1a, 1b and 2, respectively) and to a lesser extent, terrestrial ecotoxicity and marine aquatic ecotoxicity.

### *3.2.2. Impact of energy consumption*

Electricity consumption affects all the impact categories assessed. As shown in Table 4, the main environmental impacts caused by electricity consumption are abiotic depletion (resulting in environmental loads with normalised values of $3.4\cdot10^{-14}$ in scenarios 1a and 1b, and $577.8\cdot10^{-14}$ in scenario 2), marine aquatic ecotoxicity (resulting in normalised environmental loads with normalised values of $2.1\cdot10^{-14}$ in scenarios 1a and 1b, and $354.0\cdot10^{-14}$ in scenario 2) followed to a lesser extent by GWP (resulting in environmental loads with normalised values of $1.4\cdot10^{-14}$, $1.3\cdot10^{-14}$ and $227.0\cdot10^{-14}$, in scenarios 1a, 1b and 2, respectively) and acidification (resulting in  environmental loads with normalised values of $1.1\cdot10^{-14}$ in scenarios 1a and 1b, and $178.6\cdot10^{-14}$ in scenario 2). Note that the environmental impact of electricity consumption on all the impact categories evaluated in this study is considerably higher in scenario 2 than in scenarios 1a and 1b due to the considerable amount of heat energy needed in scenario 2 to maintain an operating temperature of 33 ºC (131649 $kJ\cdot m^{-3}$, see Table 3). It must be said that ideally, this study should have contemplated the impact of discharging effluent to the natural water courses at 13 ºC above the ambient temperature. In this respect, this higher temperature would increase the adverse environmental impact even more in scenario 2.

### *3.2.3. Impact of energy from biogas capture*

Energy from biogas capture has a positive impact (shown in Table 4 as negative figures) on all the impact categories evaluated because it is considered to be an energy saving. As Table 4 shows, the main environmental benefits of energy from biogas capture are abiotic depletion (resulting in environmental loads with normalised values of $-0.3 \cdot 10^{-14}$, $-0.7 \cdot 10^{-14}$ and $-1.2 \cdot 10^{-14}$, in scenarios 1a, 1b and 2, respectively), marine aquatic ecotoxicity (resulting in environmental loads with normalised values of $-0.2 \cdot 10^{-14}$, $-0.4 \cdot 10^{-14}$ and $-0.8 \cdot 10^{-14}$, in scenarios 1a, 1b and 2, respectively) and GWP (resulting in environmental loads with normalised values of $-0.1 \cdot 10^{-14}$, $-0.3 \cdot 10^{-14}$ and $-0.5 \cdot 10^{-14}$, in scenarios 1a, 1b and 2, respectively). In this case, the environmental benefits when operating at 33 ºC (scenarios 2 and 1b) are greater than when operating at 20 ºC (scenario 1a) due to higher methane production. Although in scenario 2 the heat energy generated by captured biogas can be used for heating purposes, it is a very small amount in comparison with the total energy required to achieve the operating temperature.

### *3.2.4. Impact of sludge disposal*

Table 2 shows average sludge production and stability. Sludge production was 0.25, 0.23 and 0.23 kg TSS $kg^{-1}$ $COD_{REMOVED}$ in scenarios 1a, 1b and 2, respectively. Moreover, the produced sludge was stabilised, %BVSS below 20. This table shows the impact of temperature on both sludge production and stability: slightly lower sludge production and slightly higher sludge stability were obtained at 33 ºC (scenarios 1b and 2) than at 20 ºC (scenario 1a).

The main sustainable benefits of a SAnMBR is that lower volumes of sludge are generated and no further digestion is expected to be necessary to enable the sludge to be disposed of on farmland. According to Xing et al. (2003), sludge production in activated sludge processes is generally in the range of 0.3 - 0.5 kg TSS $kg^{-1}$ $COD_{REMOVED}$. As expected, low amounts of sludge were obtained in all scenarios. In addition, the sludge was already stabilised and could therefore be used directly as fertiliser on farmland or sent to a landfill.

As shown in Table 4, the main environmental impacts of sludge disposal on farmland are marine aquatic ecotoxicity (resulting in environmental loads with normalised values of $9.8\cdot10^{-14}$ and $9.3\cdot10^{-14}$, in scenarios 1a, 1b and 2, respectively), terrestrial ecotoxicity (resulting in environmental loads with normalised values of $6.9\cdot10^{-14}$ and $6.5\cdot10^{-14}$, in scenarios 1a, 1b and 2, respectively), acidification (resulting in environmental loads with normalised values of $6.7\cdot10^{-14}$ and $5.8\cdot10^{-14}$ in scenarios 1a, 1b and 2, respectively) and fresh water aquatic ecotoxicity (resulting in environmental loads with normalised values of $5.3\cdot10^{-14}$ and $5.0\cdot10^{-14}$, in scenarios 1a, 1b and 2, respectively). As mentioned earlier, one promising alternative for the disposal of sludge is to spread it on land – with the advantage of reusing the nutrient content in the sludge as fertiliser. Although the environmental impact of disposing of sludge in landfills is slightly lower (only 15 % of all sludge generated is disposed of in landfills), the environmental impact of major factors such as abiotic depletion, global warming and photochemical oxidation can be positive if sludge is used as a fertiliser.

### *3.2.5. Overall inventory results*

It must be said that heating the process from 20 to 33 ºC (see Table 4) increases the environmental impact caused by electricity consumption considerably (because it affects abiotic depletion, marine aquatic ecotoxicity, GWP and acidification categories). Electricity consumption is therefore the major contributor to overall environmental impact, and the most significant impact categories, in descending order, are: abiotic depletion, marine aquatic ecotoxicity, global warming and acidification. The environmental loads related to electricity consumption in scenario 1b are slightly lower than in scenario 1a because, as mentioned before, of the greater volume of biogas produced at higher temperatures. According to the IPCC method, greenhouse gas emissions are considerably higher in scenario 2 (10.98 kg $CO_2$ equivalents) than in scenarios 1a and 1b (0.13 and 0.12 kg $CO_2$ equivalents, respectively). Therefore, in order for SAnMBR technology to be feasible, it is important to operate at ambient temperature which, furthermore, avoids the heating impact caused by discharging effluent which is hotter than the temperature of natural water courses.

When operating at ambient temperature (scenario 1), the effluent treated (either reused for irrigation or discharged directly onto the natural water courses) is the main contributor to overall environmental impact through eutrophication. In addition, if the methane dissolved in the effluent is not captured, human toxicity and fresh water aquatic ecotoxicity are also significant (see Figure 1). This highlights the importance of maximising the recovery of nutrients (which mainly affects eutrophication) and dissolved methane (which mainly affects human toxicity and fresh water aquatic ecotoxicity, see Figure 1) from SAnMBR effluent.

Disposing of sludge upon farmland slightly affects marine aquatic ecotoxicity, terrestrial ecotoxicity, acidification and fresh water aquatic ecotoxicity (see Table 4). Disposing of sludge in landfills has barely any environmental impact on the system, in comparison with other factors.

Effluent discharge through eutrophication is the factor that affects the LCA results most. Nevertheless, the resulting overall environmental impact when operating at different ambient temperature (scenario 1a and 1b) is quite similar. These results reveal that the different operating temperatures seem to have little influence on the hydrolysis rate (due to operating at high SRT), and thus on effluent discharge. When an input energy is required, electricity consumption is the factor that affects the LCA results most, and significant differences in overall environmental impact among the compared scenarios (scenario 1 and 2) are obtained.

## 4. Conclusions

The environmental impact of a SAnMBR system treating urban wastewater at different operating temperatures was evaluated. OEB results highlight the importance of operating at ambient temperature and optimising membrane filtration (average 0.19 $kWh \cdot m^{-3}$). Moreover, maximising the capture of methane from both biogas streams and effluent enables considerable energy savings in SAnMBRs, which enhances the feasibility of this technology in comparison with others. Furthermore, LCA results revealed the importance of operating at ambient temperature, and maximising the recovery of nutrients (eutrophication can be reduced up to 50%) and dissolved methane (positive environmental impact can be achieved) from SAnMBR effluent.

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## Table and Figure captions

**Table 1. (a)** Average characteristics of SAnMBR influent. **(b)** Average characteristics of SAnMBR effluent in scenarios 1a, 1b and 2**. (c)** Effluent characteristics after irrigation of SAnMBR effluent on farmland. *Nomenclature: **OT**: Operating Temperature; and **AT**: Ambient Temperature.*

**Table 2.** Average characteristics of SAnMBR sludge production and stability in scenarios 1a, 1b and 2. *Nomenclature: **OT**: Operating Temperature; and **AT**: Ambient Temperature.*

**Table 3. (a)** OEB of scenarios 1a, 1b and 2 divided into mechanical and heat energy consumption; power energy heat energy fuelled by biogas; and net power and heat energy. **(b)** Energy from capture of methane dissolved in effluent considering an extraction efficiency of 100%. *Nomenclature: **OT**: Operating Temperature; and **AT**: Ambient Temperature. *N/A: not applicable*

**Table 4.** LCA of SAnMBR operating at: **(a)** ambient temperature of 20 ºC (scenario 1a); **(b)** ambient temperature of 33 ºC (scenarios 1b); and **(c)** at 33 ºC when the ambient temperature is 20 ºC (scenario 2). *Method: CML 2 baseline 2000 V2.05/ / West Europe, 1995 / Normalisation / Excluding infrastructure processes/ Excluding long-term emissions. Negative values correspond to a positive environmental impact.*

**Figure 1.** LCA of treated effluent discharge (in normalised values per $m^3$) considering: **(a)** non-capture of methane dissolved in effluent; and **(b)** capture of methane dissolved in effluent. Scenario 1a: operating at ambient temperature of 20 ºC; scenario 1b: operating at ambient temperature of 33 ºC); and scenario 2: operating at 33 ºC when the ambient temperature is 20 ºC. *Method: CML 2 baseline 2000 V2.05 / West Europe, 1995 / Normalisation / Excluding infrastructure processes / Excluding long-term emissions.*

**Table 1. (a)** Average characteristics of SAnMBR influent. **(b)** Average characteristics of SAnMBR effluent in scenarios 1a, 1b and 2**. (c)** Effluent characteristics after irrigation of SAnMBR effluent on farmland. *Nomenclature: **OT**: Operating Temperature; and **AT**: Ambient Temperature.*

| | |
|---|---|
| **COD, Kg·m$^{-3}$** | 0.518 |
| **BOD, Kg·m$^{-3}$** | 0.384 |
| **VFA, Kg·m$^{-3}$** | 0.009 |
| **$N_T$, Kg·m$^{-3}$** | 0.049 |
| **$NH_4$, Kg·m$^{-3}$** | 0.041 |
| **$P_T$, Kg·m$^{-3}$** | 0.008 |
| **$PO_4$, Kg·m$^{-3}$** | 0.009 |
| **$SO_4$, Kg·m$^{-3}$** | 0.285 |
| **SST, Kg·m$^{-3}$** | 0.267 |
| **SSNV, Kg·m$^{-3}$** | 0.056 |
| **Alkalinity, Kg·m$^{-3}$** | 0.351 |

**(a)**

| **Effluent discharge** | **Scenario 1a (OT=AT= 20ºC)** | **Scenario 1b (OT=AT=33ºC)** | **Scenario 2 (OT 33ºC, AT 20ºC)** |
|---|---|---|---|
| **COD, Kg·m$^{-3}$** | 0.1718 | 0.1656 | 0.1647 |
| **$NH_4$, Kg·m$^{-3}$** | 0.0564 | 0.0573 | 0.0573 |
| **$PO_4$, Kg·m$^{-3}$** | 0.0186 | 0.0191 | 0.0192 |
| **$SO_4$, Kg·m$^{-3}$** | 0.0002 | 0.0001 | 0.0001 |
| **CH4, Kg·m$^{-3}$** | 0.0173 | 0.0190 | 0.0181 |
| **$H_2S$, Kg·m$^{-3}$** | 0.1003 | 0.1001 | 0.0999 |

**(b)**

| **Effluent discharge** | **Scenario 1a (OT=AT= 20ºC)** | **Scenario 1b (OT=AT=33ºC)** | **Scenario 2 (OT 33ºC, AT 20ºC)** |
|---|---|---|---|
| **$NH_4$, Kg·m$^{-3}$** | 0.0282 | 0.0286 | 0.0286 |
| **$PO_4$, Kg·m$^{-3}$** | 0.0056 | 0.0057 | 0.0057 |

**(c)**

**Table 2.** Average characteristics of SAnMBR sludge production and stability in scenarios 1a, 1b and 2.

*Nomenclature: **OT**: Operating Temperature; and **AT**: Ambient Temperature.*

| **Sludge** | **Scenario 1a (OT=AT= 20ºC)** | **Scenario 1b (OT=AT=33ºC)** | **Scenario 2 (OT 33ºC, AT 20ºC)** |
|---|---|---|---|
| **Kg TSS· kg$^{-1}$ COD$_{REMOVED}$** | 0.25 | 0.23 | 0.23 |
| **VSS, %** | 56.3 | 53.8 | 53.8 |
| **BVSS, %** | 19.7 | 9.8 | 9.8 |

**Table 3. (a)** OEB of scenarios 1a, 1b and 2 divided into mechanical and heat energy consumption; power energy heat energy fuelled by biogas; and net power and heat energy. **(b)** Energy from capture of methane dissolved in effluent considering an extraction efficiency of 100%. *Nomenclature:* ***OT****: Operating Temperature; and* ***AT****: Ambient Temperature. *N/A: not applicable*

| | **Scenario 1a (OT=AT= 20ºC)** | **Scenario 1b (OT=AT=33ºC)** | **Scenario 2 (OT 33ºC, AT 20ºC)** |
|---|---|---|---|
| **Energy consumption** | | | |
| Mechanical energy consumption , $kWh\cdot m^{-3}$ | 0.219 | 0.218 | 0.218 |
| Anaerobic reactor sludge mixing pump | 0.0005 | 0.0004 | 0.0004 |
| Anaerobic reactor wastewater feeding pump | 0.0022 | 0.0022 | 0.0022 |
| Membrane tank sludge feeding pump | 0.0857 | 0.0853 | 0.0853 |
| Permeate Pump | 0.0052 | 0.0052 | 0.0052 |
| Anaerobic reactor biogas recycling blower | 0.0113 | 0.0113 | 0.0113 |
| Membrane tank biogas recycling blower | 0.1017 | 0.1019 | 0.1017 |
| Rotofilter | 0.0055 | 0.0055 | 0.0055 |
| Sludge dewatering | 0.0067 | 0.0064 | 0.0064 |
| Heat energy consumption, $KJ\cdot m^{-3}$ | 0.0000 | 0.0000 | 131649 |
| Heat required for heating inflow (Qrequired) | *N/A** | *N/A** | 54408 |
| Heat dissipated through reactor (Qdissipated) | *N/A** | *N/A** | 75428 |
| Heat in the gas decompression (Qdecompression) | *N/A** | *N/A** | -271 |
| Heat enthalpy of the biological reactions (Qenthalpy) | *N/A** | *N/A** | 2085 |
| **Energy from biogas capture** | | | |
| Power energy production , $kWh\cdot m^{-3}$ | 0.021 | 0.042 | 0.044 |
| Heat energy production , $KJ\cdot m^{-3}$ | 65.897 | 132.031 | 136.417 |
| **Net power energy, $kWh\cdot m^{-3}$** | 0.198 | 0.176 | 0.174 |
| **Net heat energy, KJ·** $m^{-3}$ | -65.897 | -132.031 | 131512 |
| **OEB, $kWh\cdot m^{-3}$** | 0.20 | 0.18 | 36.71 |

**(a)**

| | | | **Power energy generated** | **Heat energy generated** | **Total energy recovered** |
|---|---|---|---|---|---|
| Scenarios | $mg_{CH4}\cdot l^{-1}$ | $l_{CH4}\cdot dia^{-1}$ | $kWh\cdot m^{-3}$ | $KJ\cdot m^{-3}$ | $kWh\cdot m^{-3}$ |
| Scenario 1a (OT=AT= 20ºC) | 70.53 | 56.13 | 0.075 | 235.78 | 0.075 |
| Scenario 1b (OT=AT=33ºC) | 77.89 | 61.99 | 0.083 | 260.38 | 0.083 |
| Scenario 2 (OT 33ºC, AT 20ºC) | 76.13 | 60.589 | 0.081 | 254.50 | 0.152 |

**(b)**

**Table 4.** LCA of SAnMBR operating at: **(a)** ambient temperature of 20 ºC (scenario 1a); **(b)** ambient temperature of 33 ºC (scenario 1b); and **(c)** at 33 ºC when the ambient temperature is 20 ºC (scenario 2). *Method: CML 2 baseline 2000 V2.05/ / West Europe, 1995 / Normalisation / Excluding infrastructure processes/ Excluding long-term emissions. Negative values correspond to a positive environmental impact.*

| Impact category | Total ($\cdot 10^{-14}$) | Energy consumption ($\cdot 10^{-14}$) | Energy from biogas capture ($\cdot 10^{-14}$) | Sludge disposal | | Effluent discharge ($\cdot 10^{-14}$) |
|---|---|---|---|---|---|---|
| | | | | Farmland ($\cdot 10^{-14}$) | Landfill ($\cdot 10^{-14}$) | |
| Eutrophication | 158.8726 | 0.1958 | -0.0188 | 3.3025 | 1.9280 | 153.4651 |
| Marine aquatic ecotoxicity | 11.6750 | 2.1077 | -0.2031 | 9.8158 | 0.0247 | -0.0700 |
| Acidification | 7.7487 | 1.0630 | -0.1024 | 6.7452 | 0.0568 | -0.0140 |
| Terrestrial ecotoxicity | 7.4031 | 0.1481 | -0.0143 | 6.8798 | 0.0051 | 0.3843 |
| Fresh water aquatic ecotox. | 70.7456 | 0.0436 | -0.0042 | 5.2833 | 0.0013 | 65.4215 |
| Abiotic depletion | 3.2047 | 3.4399 | -0.3314 | -0.0047 | 0.1425 | -0.0415 |
| Global warming (GWP100) | 2.5455 | 1.3511 | -0.1302 | -0.0017 | 1.3403 | -0.0141 |
| Human toxicity | 69.7208 | 0.1389 | -0.0134 | 1.5487 | 0.0013 | 68.0453 |
| Photochemical oxidation | 0.3407 | 0.1426 | -0.0137 | -0.0003 | 0.2141 | -0.0019 |
| Ozone layer depletion (ODP) | 0.0061 | 0.0068 | -0.0007 | 0.0000 | 0.0001 | -0.0001 |

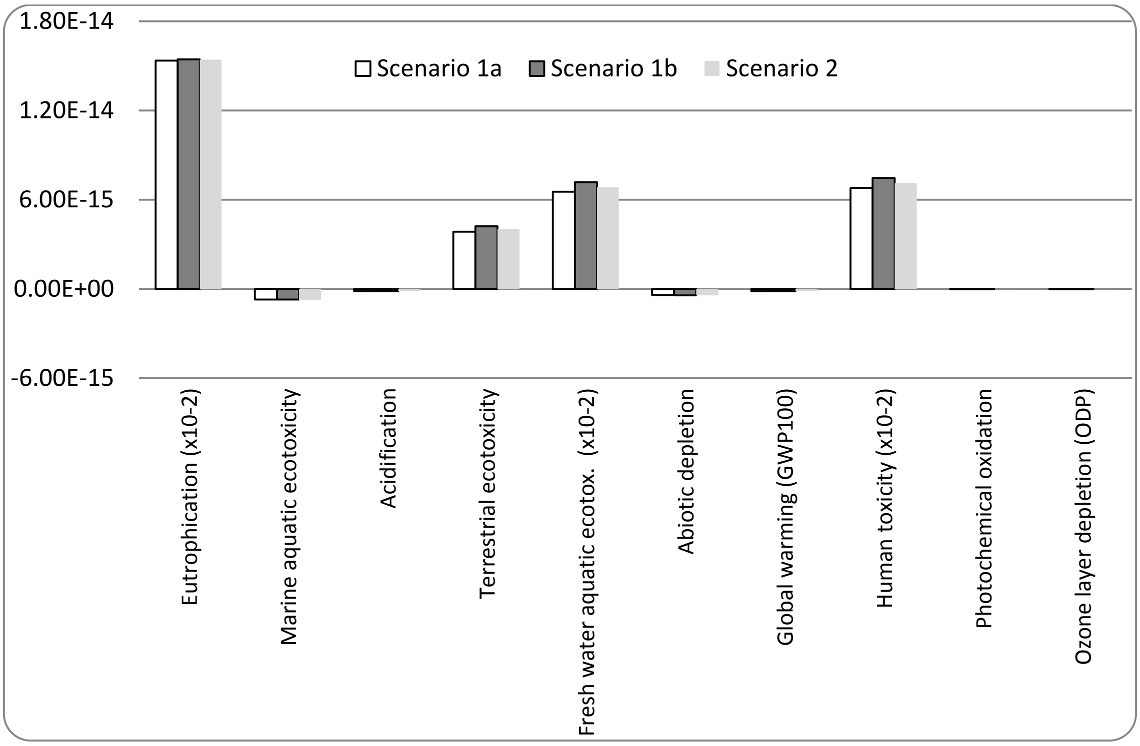
**(a)**

| Impact category | Total ($\cdot 10^{-14}$) | Energy consumption ($\cdot 10^{-14}$) | Energy from biogas capture ($\cdot 10^{-14}$) | Sludge disposal | | Effluent discharge ($\cdot 10^{-14}$) |
|---|---|---|---|---|---|---|
| | | | | Farmland ($\cdot 10^{-14}$) | Landfill deposition ($\cdot 10^{-14}$) | |
| Eutrophication | 159.1307 | 0.1949 | -0.0376 | 2.8386 | 1.8213 | 154.3135 |
| Marine aquatic ecotoxicity | 10.9076 | 2.0981 | -0.4051 | 9.2609 | 0.0233 | -0.0695 |
| Acidification | 6.6890 | 1.0582 | -0.2042 | 5.7957 | 0.0537 | -0.0143 |
| Terrestrial ecotoxicity | 7.0542 | 0.1474 | -0.0285 | 6.5077 | 0.0049 | 0.4227 |
| Fresh water aquatic ecotox. | 76.8873 | 0.0434 | -0.0084 | 5.0006 | 0.0013 | 71.8504 |
| Abiotic depletion | 2.8501 | 3.4241 | -0.6612 | -0.0041 | 0.1346 | -0.0433 |
| Global warming (GWP100) | 2.3352 | 1.3449 | -0.2597 | -0.0015 | 1.2661 | -0.0146 |
| Human toxicity | 76.3144 | 0.1383 | -0.0267 | 1.4693 | 0.0012 | 74.7322 |
| Photochemical oxidation | 0.3145 | 0.1419 | -0.0274 | -0.0003 | 0.2023 | -0.0020 |
| Ozone layer depletion (ODP) | 0.0055 | 0.0068 | -0.0013 | 0.0000 | 0.0001 | -0.0001 |

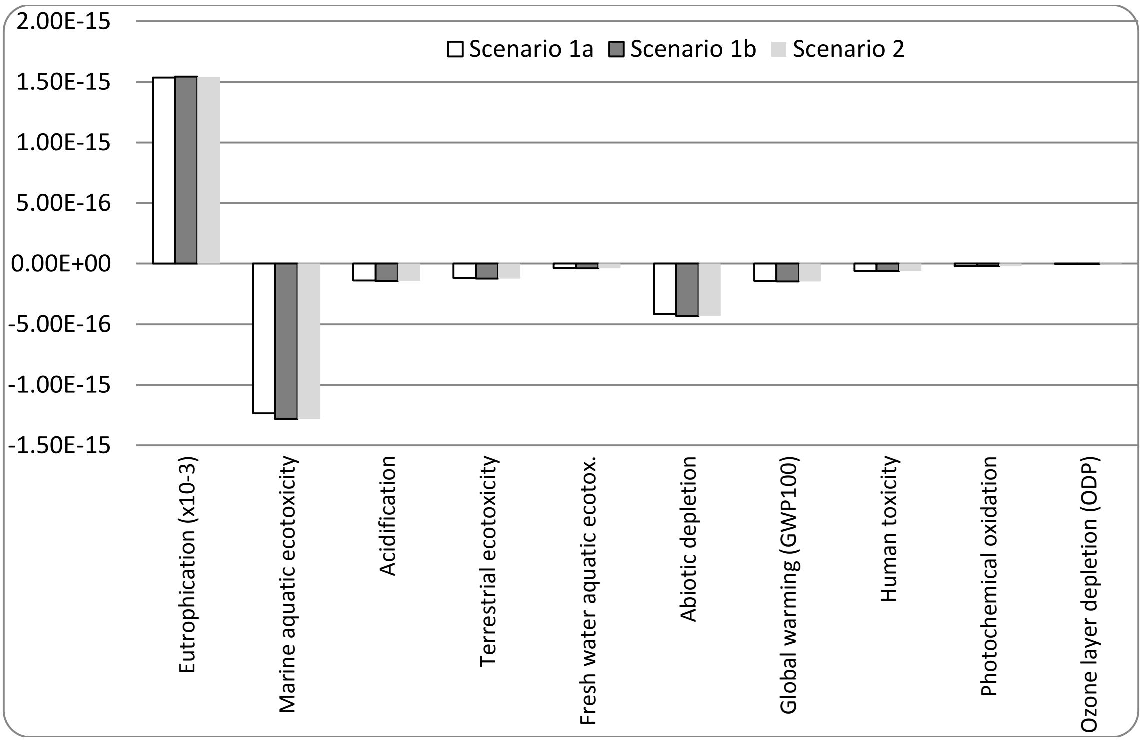
**(b)**

| Impact category | Total ($\cdot 10^{-14}$) | Energy consumption ($\cdot 10^{-14}$) | Energy from biogas capture ($\cdot 10^{-14}$) | Sludge disposal | | Effluent discharge ($\cdot 10^{-14}$) |
|---|---|---|---|---|---|---|
| | | | | Farmland ($\cdot 10^{-14}$) | Landfill deposition ($\cdot 10^{-14}$) | |
| Eutrophication | 191.6357 | 32.8911 | -0.0727 | 2.8414 | 1.8213 | 154.1547 |
| Marine aquatic ecotoxicity | 362.4733 | 354.0457 | -0.7843 | 9.2609 | 0.0233 | -0.0723 |
| Acidification | 184.0135 | 178.5680 | -0.3954 | 5.8015 | 0.0537 | -0.0143 |
| Terrestrial ecotoxicity | 31.7411 | 24.8815 | -0.0551 | 6.5077 | 0.0049 | 0.4021 |
| Fresh water aquatic ecotox. | 80.7569 | 7.3244 | -0.0162 | 5.0006 | 0.0013 | 68.4468 |
| Abiotic depletion | 576.6242 | 577.8171 | -1.2801 | -0.0041 | 0.1346 | -0.0433 |
| Global warming (GWP100) | 227.7044 | 226.9572 | -0.5028 | -0.0015 | 1.2661 | -0.0146 |
| Human toxicity | 95.9476 | 23.3368 | -0.0517 | 1.4693 | 0.0012 | 71.1920 |
| Photochemical oxidation | 24.0949 | 23.9479 | -0.0530 | -0.0003 | 0.2023 | -0.0020 |
| Ozone layer depletion (ODP) | 1.1397 | 1.1422 | -0.0025 | 0.0000 | 0.0001 | -0.0001 |

**(c)**




**(a)**




**(b)**

**Figure 1.** LCA of treated effluent discharge (in normalised values per $m^3$) considering: **(a)** non-capture of methane dissolved in effluent; and **(b)** capture of methane dissolved in effluent. Scenario 1a: operating at ambient temperature of 20 ºC; scenario 1b: operating at ambient temperature of 33 ºC); and scenario 2: operating at 33 ºC when the ambient temperature is 20 ºC. *Method: CML 2 baseline 2000 V2.05 / West Europe, 1995 / Normalisation / Excluding infrastructure processes / Excluding long-term emissions.*