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Assessing the life-cycle sustainability of algae and bacteria-based wastewater treatment systems: high-rate algae pond and sequencing batch reactor

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Abstract

High Rate Algae Ponds (HRAPs) are a promising technology for the treatment of municipal wastewater in locations with sufficient space and solar radiation. Algae-based processes do not require aeration, and thus have the potential to be less energy-intensive than activated sludge processes.

We used a combination of LCA and LCCA analysis to evaluate the sustainability of HRAP systems, using data from the construction and operation of two demonstration-scale systems in Almería and Cádiz, Spain. As a reference for comparison, we used data from an activated sludge-based Sequencing Batch Reactor (SBR) treatment system in operation in Leppersdorf, Germany, which has comparable removal rates for a similar inflow. We focused solely on the actual wastewater treatment aspect of these technologies, excluding sludge treatment from this analysis.

Based on our analysis, the current HRAP technology is more energy-efficient than activated sludge-based SBRs and requires only 22% of its electricity consumption. In addition, HRAP is

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more advantageous both economically (0.18 €/m^3 versus 0.26 €/m^3) and environmentally, with both lower global warming and eutrophication potentials (146.27 *vs*. 458.27 x 10^{-3} kg CO_2 equiv./m^3; 126.14 *vs*. 158.01 x 10^{-6} kg PO_4 equiv./m^3). However, the Net Environmental Benefit of SBR was lightly more favorable than of HRAP because of the higher removal rate for nutrients of SBR.

1. Introduction

Ensuring safe sanitation and protection of precious water resources for the world's growing population requires the development and implementation of decentralized solutions and sustainable wastewater treatment, especially in rural and suburban areas (Capodaglio, 2017; Eggimann et al., 2018; van Afferden et al., 2015).

At the moment, bacteria-based biological processes are the most common form of wastewater (WW) treatment at all scales. In activated sludge-based systems, an aerated phase is used for the removal of organic matter (measured as Chemical Oxygen Demand or COD) and nitrification, and an anoxic phase for denitrification. Phosphorus can be removed by means of chemical dosing or the implementation of an anaerobic step for enhanced biological phosphorus removal (or EBPR). Although efficient and robust, the activated sludge process in all technical configurations – carrousel, Modified Ludzack-Ettinger (MLE), Sequence Batch Reactor (SBR), etc. – is energy-intensive, primarily due to aeration requirements (Zhang et al., 2018). The electricity consumption of bacteria-based systems varies between 0.36 and 1.26 kWh/m^3 treated wastewater (see Table 1.) according to size and technology (Garfí et al., 2017; Lorenzo-Toja et al., 2016).

Source	Scope	Technology	Original functional unit*	Electricity consumption, kWh/m^3	GWP, kg CO_2 equiv./m^3
Garfí et al. (2017)	WT and ST with direct emissions C+O	Algae-based HRAP Activated sludge system Constructed wetland	1 m^3 WW treated	0.25 1.26 0.22	0.57 1.27 0.69
Maga (2017)	WT and ST with direct emissions and sludge disposal C+O	Algae-based HRAP	1 m^3 WW treated	-	0.280
Bao et al. (2016)	WT with direct emissions Size 0.23-1.2 MPE* O	SBR Anoxic/oxic process	1 m^3 WW treated	-	0.865 0.405
Lorenzo-Toja et al. (2016)	WT and ST Size 5000-1M PE O	Pre-treatment, bioreactor, secondary and tertiary settling, dewatering	1 m^3 WW treated	0.360**	0.345-0.378
Cornejo et al. (2013)	WT and ST C+O 727 PE	Facultative pond with two maturation ponds with water reuse	1 m^3 WW treated	1.069	0.500
	WT and ST C+O 1471PE Both with direct emissions	UASB with two maturation ponds with water reuse and energy recovery		0.986	1.510

Table 1. Environmental impact of WW treatment technologies

Abbreviations: C-Construction; O-Operation; WT-water treatment; ST-sludge treatment; PE-population equivalent; MPE – million population equivalent; UASB-upflow anaerobic sludge blanket reactor; WW-wastewater; SBR-sequencing batch reactor. Clarifications: *To convert FU from PE to m^3, 200 L/person*day WW production was assumed. **Average value for 22 WWTPs in Spain [+]The ratio of m^3 potable water and WW was set to 1 [++]from Amores et al. (2013).

Algae systems, referred to as High Rate Algae Ponds (HRAPs, Vikrant et al. (2018)), have been receiving increasing attention as a promising alternative to activated sludge systems for the treatment of municipal WW, particularly for small- to medium-scale treatment plants (WWTPs) serving between 200 and 15,000 population equivalents (PE) (Annavajhala et al., 2018). Algae utilize solar energy for growth, assimilating nitrogen and phosphorus from wastewater, and produce O_2 by photosynthesis, thus making mechanical aeration unnecessary for aerobic bacterial activity. When grown in a mixed culture, heterotrophic bacterial respiration provides CO_2 which serves as a carbon source for the algae while removing COD, thus avoiding the extra CO_2 supply that is required for cultivating algae in pure cultures (Posadas *et al*. 2017). Consequently, HRAP systems require considerably less electricity (0.25 kWh/m^3 treated wastewater (Garfí et al., 2017)), which has an advantage especially in small-scale systems. On the other hand, algal-ponds require much more space than bacteria-based systems (30 and 0.18-3 m^2/m^3 WW treated, respectively

(Bao et al., 2016; Garfí et al., 2017)), which results in high material inputs and expensive investment. HRAPs are therefore an attractive option for smaller systems in locations with ample space, frost-free temperatures and year-round solar radiation, conditions needed for algae growth. Life-cycle assessment (LCA) and life-cycle cost assessment (LCCA) are tools that can be used to assess the sustainability of the HRAP process in economic and environmental terms and to compare it to the more established activated sludge process. Previous studies have assessed the life-cycle sustainability of both algae cultivation and activated sludge systems (Bao et al., 2016; Cornejo et al., 2013; Garfí et al., 2017; Lorenzo-Toja et al., 2016; Maga, 2017). For the bacteria-based technology, these studies calculated Global Warming Potentials between 0.345 and 1.51 kg CO_2-equiv/m^3 WW treated (see Table 1). The range of reported values is broad, likely due to differences in life-cycle length, scope and technical details of WWTP operation. Direct emissions, such as N_2O, CH_4 and NH_3, and chemical additives, e.g. poly-aluminium chloride (PAC) and poly-acrylamide (PAM), were often considered important factors influencing the environmental impact of WWT systems.

However, the material and energy inputs used for the analysis in these studies often stem from models and hypothetical planning calculations rather than real data. Additionally, in recent years both HRAP and activated sludge technologies have advanced substantially (e.g. new mixing technology (Annavajhala et al., 2018)) with positive effects on their ecological and economic impacts, which so far have not been subject of sustainability analysis in a peer-reviewed journal. Consequently, a comparison of advanced HRAP and SBR technologies based on real planning data and empirical operational experience focusing on their treatment performance has been absent from the literature.

To address this gap, we based our calculations on empirical data for the two compared systems, HRAP and SBR.

2. **Materials and methods**

A cradle-to-gate LCA and LCCA of a demonstration scale HRAP-based wastewater treatment plant (WWTP) treating municipal wastewater in Almería and Cádiz, Spain, was carried out assuming a 40 year total lifespan (a 20-40 year period is a common value used to assess the life cycle of a wastewater treatment plant (Corominas et al., 2013; Langeveld, 2015; van Afferden et al., 2015) and a treatment capacity of 300 m^3 wastewater/day. An LCA and LCCA of a SBR-based wastewater treatment plant in Leppersdorf, Germany, that treats the same quality and amount of wastewater (WW) as the HRAP was performed in parallel as a reference for conventional activated sludge treatment technology. Both wastewater treatment plants were open air functioning under the climatic conditions of their location.

Data from the construction and operation was used of two demonstration-scale HRAP systems in Almería and Cádiz, Spain. We chose to compare this system with an activated sludge system in a SBR configuration, because SBRs are widely implemented, flexible and increasingly used wastewater treatment technology at small scale (up to 5000 population equivalent) in densely populated regions (Dutta and Sarkar, 2015; Fernandes et al., 2013). For this, data from a SBR in Leppersdorf, Germany, with a population equivalent range comparable to that of the HRAP was collected.

In our analysis, we only focused on the wastewater treatment of both technologies. Neither the potential for biomass production (algal or activated sludge) as a source for low- and high-value products nor the sludge treatment were considered due to the complexity of these assessment options. When assessing the sustainability and economic performance of HRAP, we compared data from ponds with a novel type of submerged mixing system (as opposed to the more common paddle wheel).

The goal and scope, inventory development, and impact assessment of the LCA were defined and carried out according to the ISO 14040:2006 standard, using GaBi8 LCA software and the GaBi databases Professional, Construction materials, Food&Feed, and the ecoinvent3 database. Distinct modules of the wastewater treatment process (e.g. pretreatment, raceway, separator) and corresponding sub-modules (e.g. “agitator”, “separator drum”) were modeled individually and then integrated into a comprehensive LCA model.

In GaBi8, the software tool for creating and calculating life-cycle assessment models, parameter tables were used for data input in a form of diagonal matrix. These parameter tables enabled us to gain separate results for the different sections of the wastewater treatment process and identify environmental hot spots along the technology.

The most important environmental impact caused by WW is the eutrophication potential (EP) (Lorenzo-Toja et al. 2016). The concept of Net Environmental Benefit (NEB) (Godin et al. 2012) considers EP and captures the environmental impact of outflow differences of WWT technologies. We used this concept for our sensitivity analysis. The concept distinguishes between the EP of untreated water and treated water and the difference of them gives the environmental benefit. When the EP of the wastewater treatment plant is subtracted from the environmental benefit the net environmental benefit is gained.

For the LCCA, the investment, operation, and maintenance costs for the entire life cycle of both technologies were calculated from data of the HRAP demonstration sites in Almería and Cádiz, and from the planning and operational data of the SBR plant in Leppersdorf, Germany. In addition, our inquiry also focused on the role of different cost categories, such as chemicals and electricity. Consequently, the contribution of these categories to life-cycle costs and environmental impacts was also scrutinized.

2.1. Demonstration-scale HRAP and conventional SBR

The HRAP water line consists of (i) a pretreatment step for solids removal – including a storage tank and rotary drum filter, (ii) a raceway algae pond and (iii) a separator, a conical drum in which the algae sludge is separated from the treated water by flotation (Figure 1). The raceway ponds have an active surface of 3000 m^2, a volume of 900 m^3 each, and are designed to operate with a 36-hour hydraulic retention time (HRT). The HRAP was calculated with two alternative mixing constructions: the conventional paddle wheel and a submersed mixing system patented as the "Low Energy Algae Raceway (LEAR)". This submersed mixing system consist of a flow booster with propeller and motor, and a built channel for mixing. The treated wastewater from the HRAP was led to the separator, where algae sludge was flocculated and separated from cleaned water. At this stage chemicals, such as poly-aluminium chloride (PAC 18%) and polyacrylamide (PAM), were added. The sludge concentration leaving the system after separation was 4%.

The reference SBR system was set to treat the same inflow and achieve similar removal rates of biological and chemical oxygen demand (BOD, COD), total suspended solids (TSS), total Kjeldahl nitrogen (TKN), and total phosphorus (TP) as the HRAP. The SBR plant consists of a pretreatment unit with a filter and sand trap, a buffer tank, an SBR tank, and a sludge tank (Figure 1).

The SBR tank was modelled to operate in 8h cycles with the following schedule: filling 57.47 min., anaerobic mixing 120 min., react – aerobic mixing 120 min., anoxic mixing 30 min, settle

90 min., decant 57.47 min., idle 4.8 min. This timing achieves the elimination rates of COD, TN, and TP that are indicated in Table 2.

		Inflow WW	Outflow HRAP	Outflow SBR	EU requirements
BOD_5	mg O_2/L	350	9	6.75	25
COD	mg O_2/L	800	80	28.68	125
TSS	mg TSS/L	500	20	10	60
TKN	mg N/L	67	15	(TN) 12,35	(TN) 15
TP	mg P/L	10	1	1.43	2

Table 2. Typical values for BOD, COD, TSS, TKN, and TP in wastewater and in HRAP or SBR outflow.
The inflow and outflow values of HRAP were empirically defined by FCC AQUALIA, Spain
The inflow and outflow values of SBR were calculated as an average value of the years 2014-17 from the reporting protocol provided by the SBR plant in Leppersdorf, Germany
EU requirements are specified in the European Directive (91/271/EEC)
Conventional wastewater parameters including biochemical oxygen demand over five days (BOD_5), chemical oxygen demand (COD), total suspended solids (TSS), total Kjeldahl nitrogen (TKN), total nitrogen (TN) and total phosphorus (TP) were analyzed by approved wastewater laboratories in Spain and Germany that are accredited according to DIN EN ISO / IEC 17025.

Downstream of the SBR tank, a sludge tank is provided where the settled sludge is treated with the addition of PAM (Praestol) to thicken it and to obtain a dry matter content of the sludge comparable to the dry matter content of the algal biomass after separation (i.e. approx. 3.5-4.8%).

The two alternative routes of wastewater treatment and the relevant flows per day are presented in Figure 1. The outflow parameters – obtained from operation of the HRAP in Cádiz and from the SBR, fulfilling the EU requirements for both systems – are presented in Table 1. For more information on the WWTPs, see SI-1 in the supplementary information.

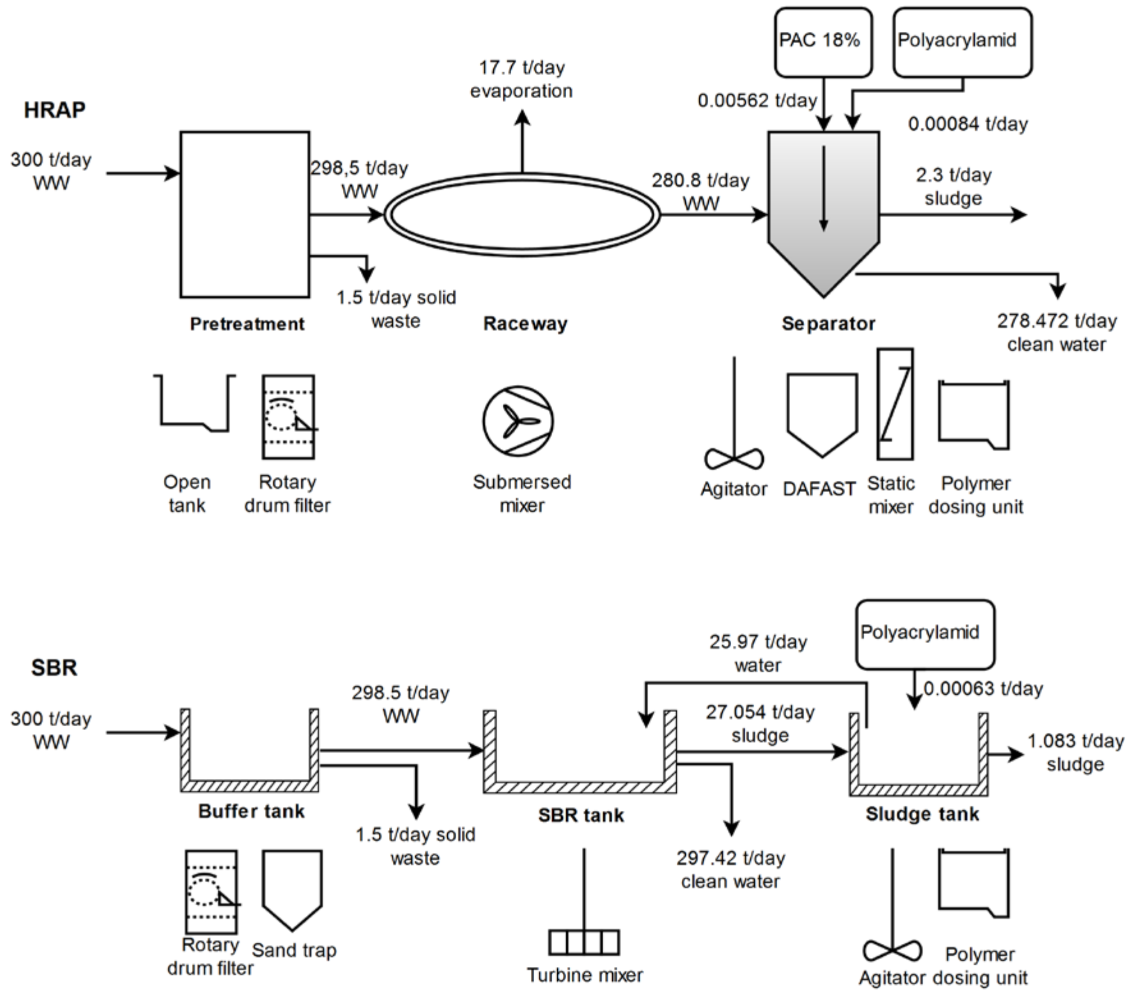

Figure 1. Defined system boundaries of the HRAP system and the reference system (SBR)

The functional unit (FU) was set to “1 m^3 of treated wastewater”. The shorter lifespan of some of the equipment (e.g. 20 years for drum filters, 10 years for pumps) was taken into consideration, as shown in the LCA inventory (Table SI-1).

2.2. Inventories

The data necessary for the LCA was taken from the construction and operational data of the demonstration HRAP plants in Almería and Cádiz, Spain, and of the SBR WWTP in Leppersdorf, Germany. Whenever suitable unit processes were available in the databases, these were included in the LCA model.

Otherwise, proxies were calculated based on material composition, weight, and type and amount of energy consumption. Table SI-1 presents all the relevant material and energy inputs used in modeling the wastewater treatment process.

The LCCA data also covered a 40 year lifespan of the WWTP. Capital expenditures (CAPEX) included every investment necessary for implementing the infrastructure, including foundations, structural work, land, and equipment. Additional investments were added to the initial CAPEX for replacement of equipment. Prices for materials, land, transport, and electricity were taken from the Spanish case in order to avoid price differences and make the two cases as comparable as possible. Please see Table SI-2 for details in the supplementary information.

Operating expenditures (OPEX) per year of WWTP operation included the cost of personnel, electricity, spare parts and materials necessary for maintenance, as well chemicals, including iron-chloride sulfate, polyacrylamide (PAM) and poly-aluminum chloride (PAC18%) – with costs of 154.7 €/t, 2,413 €/t, and 241.3 €/t, respectively (http1, http2).

The present value CAPEX and OPEX dependency on (i) discount rate, (ii) land cost, and (iii) personnel workload was assessed using the same criteria for both the HRAP and SBR technologies. Two extreme scenarios were considered for the discount rate: 0.25% (the typical interest rate of the European Central Bank (European Central Bank – http3 in 2017) and 3% (a typical risk-free interest rate in the Eurozone – and close to the interest rate of new loans up to 250,000€, 2.43% (European Central Bank – http4). These numbers provided a wide enough range to incorporate the opportunity costs of low-risk investments. We used the average land cost value in Spain in 2016: 1.05 €/m^2 (http5). The cost of personnel for the HRAP was estimated by assuming a need of 43 working h/month (Pogade et al., 2015), which corresponds to 0.29 of the total working hours of a full-time job in Spain (gobex, -) and a salary of 3,161 €/month (gobex, -). This results in personnel costs of 917 €/month for the HRAP. The personnel cost of the SBR was set to 1,418 €/month. A price of 0.1 €/kWh was assumed for the cost of electricity.

2.3. Impact assessment

The environmental impact of the inventory data was calculated using the characterization model[1] CML2001 - Jan. 2016 (Hischier et al., 2010) to assess the global warming potential (GWP) and eutrophication potential (EP) (Lorenzo-Toja et al., 2016). Direct emissions of the greenhouse gases N_2O and CH_4 were estimated based on values found in literature for N_2O (1.8 kg CO_2 equiv./m^2 yr in HRAP; 0.5% of the N removed in SBRs) and CH_4 (0.85% of COD treated in SBRs, negligible in HRAP) (Béchet et al., 2017; Campos et al., 2016).

To assess the economic impact of the HRAP and SBR WWT plants, a dynamic cost comparison with net present value calculation for the entire lifetime (40 years) was carried out. The discounted costs are summed and expressed per m^3 treated WW, giving the unit production costs of the wastewater treatment technology.

3. Results

3.1. Economic impact of HRAP versus SBR: CAPEX and OPEX

Based on our analysis, the HRAP operating costs are nearly half of those of the SBR to treat 1 m^3 of wastewater (Table 3). The high electricity consumption required to operate a sequencing batch reactor, including filling, mixing, aeration, and emptying, compared to the low energy

[1] A characterization model consist of characterization factors that transform the value of the different flows into and from the environment to environmental impacts. Characterization factors for this model can be downloaded from https://www.universiteitleiden.nl/en/research/research-output/science/cml-ia-characterisation-factors.

requirements of the HRAP, accounts for most of the difference. Since HRAP does not require aeration, this system saves a lot of energy, which makes it more cost-effective than the SBR.

Costs	CAPEX without cost of land, €	Land, €	CAPEX total, €	OPEX without personal cost, €/year	Personal costs, €/year	OPEX total, €/year
HRAP	291,407	3,392	294,799	2,369	11,000	13,369
SBR	208,772	3,150	211,922	6,455	17,773	24,229

Table 3. Cost categories and total cost of HRAP and SBR

3.2. Additional cost saving potential

Additionally, we identified two major areas with potential to decrease HRAP operating costs even further: (i) the chemical additives used for coagulation and flocculation during the algae harvesting step (i.e. PAC18% and PAM) – which made up 52% of the operating costs when personnel costs are not considered – and (ii) personnel costs, which accounted for 82% of the total operating costs.

The total CAPEX for the HRAP was more similar to the SBR than the OPEX (Tables 2 and SI-2). The CAPEX of HRAP was only 82,876 € more expensive than the SBR plant. The temporal distribution of life cycle costs (CAPEX + OPEX) and the difference between the two technologies is presented in Figure 2, showing a considerable advantage of HRAP at discount rates of 0.25% (0.182 vs 0.258 € per m^3 of wastewater treated with the HRAP and SBR, respectively) and 3% (0.125 vs 0.167 € per m^3 of wastewater treated).

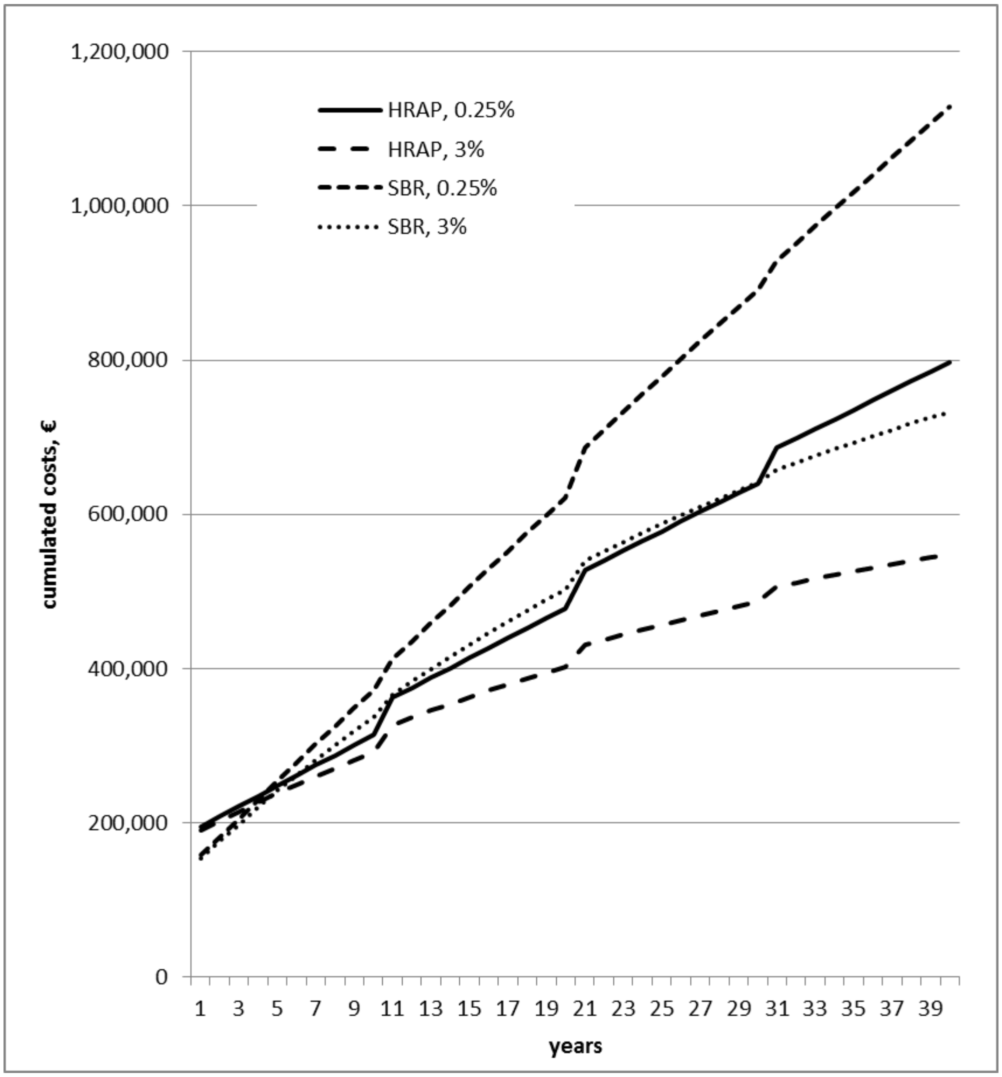

Figure 2. LCC of HRAP and SBR at 0.25% and 3% discount rates

According to these results, current HRAP technology – even before upscaling and optimization – is more cost-efficient than the referenced activated sludge-based SBR, especially in terms of operating costs.

3.3. Reducing power consumption of HRAP through efficient mixing systems

According to our data, HRAP systems can consume just 22% of the total energy needed by their SBR equivalent (0.10 vs. 0.45 kWh/m^3 WW treated, Figure 3) when using a novel type of submersed mixing technology, the “Low Energy Algae Raceway” (LEAR). With the conventional paddle wheel mixing it is 0.17 kWh/m^3 WW treated. For the SBR system, 74% of the electricity is consumed by aerating and mixing the SBR tank in its reaction phase.

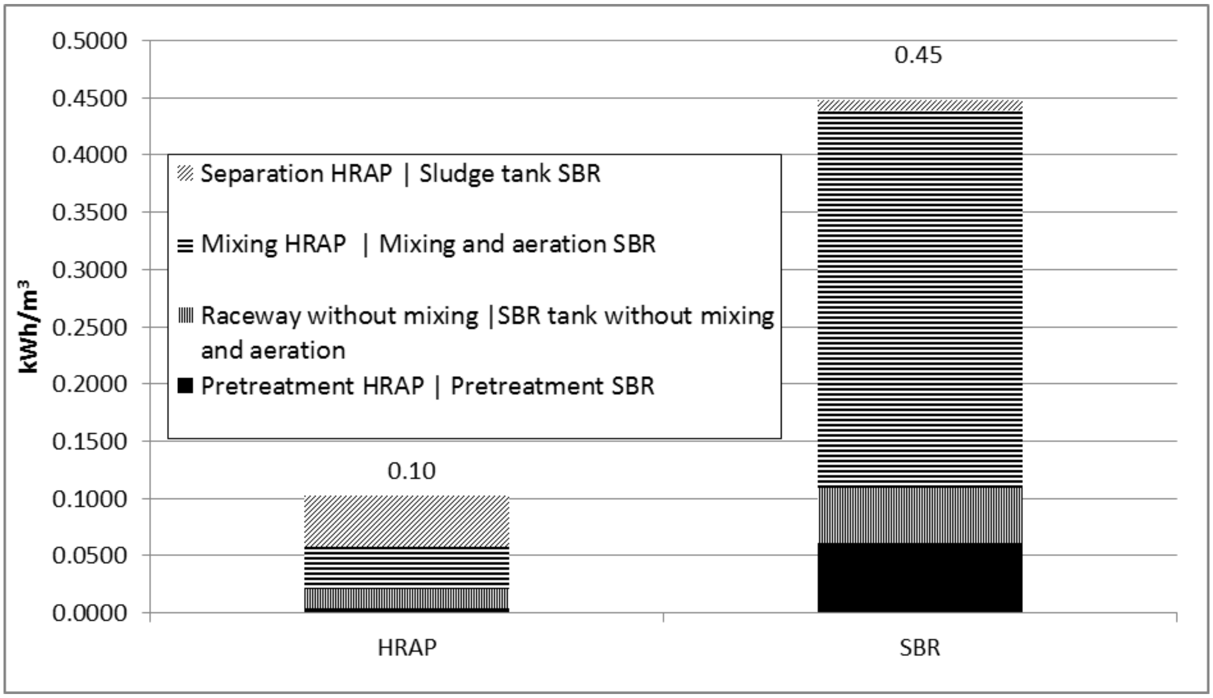

Figure 3. Electricity consumption

Mixing in algae ponds, typically by means of a simple paddle wheel mechanism, represents the second most power-consuming process of HRAP treatment systems, surpassed only by the algae harvesting step. However, we found that LEAR systems require less than half the power consumption for mixing of the paddle wheel equivalent (0.0375 vs. 0.103 kWh/m^3 WW treated, respectively).

3.4. Environmental impact of the HRAP vs. SBR: GWP and EP

According to our analysis, WW treatment in HRAP systems has a lower environmental impact than the SBR in terms of GWP and EP (146.27 vs. 458.27 x 10^{-3} kg CO_2 equiv./m^3 and 126.14 vs. 158.01 x 10^{-6} PO_4 equiv./m^3) (Table SI-2 and Figure 4-5). Electricity consumption accounts for more than 40% of the CO_2 equiv./m^3 and 27% of PO_4 equiv./m^3 in SBR operation. Furthermore, direct greenhouse gas emissions (primarily in the form of N_2O) are presumed to be higher in this type of bacterial nitrification-denitrification system than in an algae-dominated system.

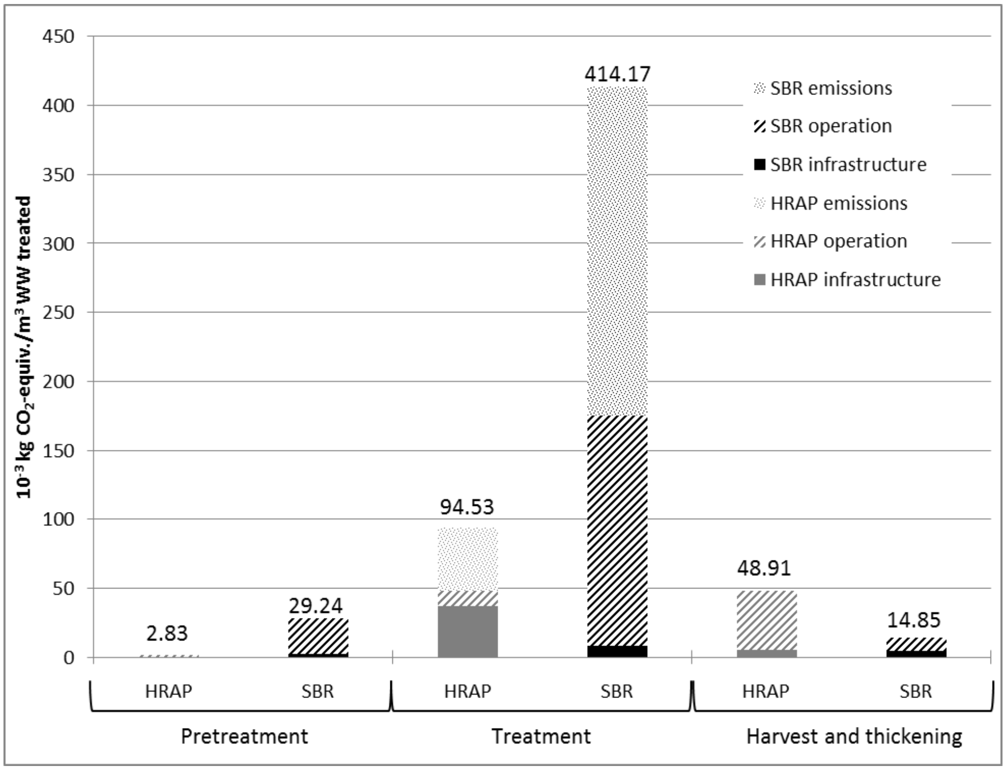

Figure 4. GWP of treatment phases in HRAP and SBR

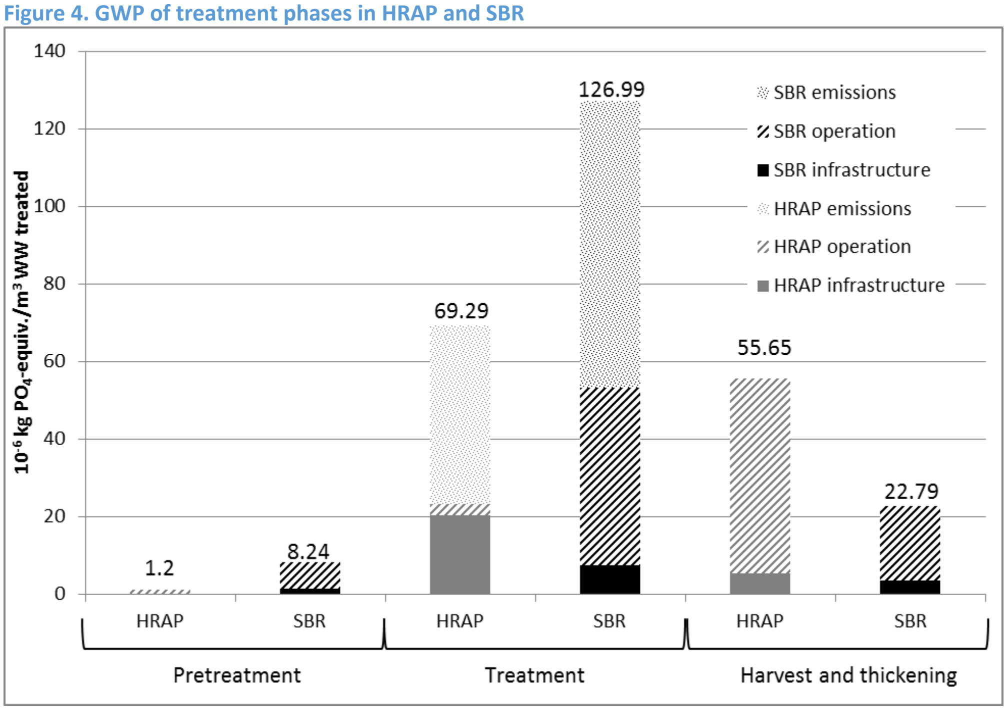

Figure 5. EP of treatment phases in HRAP and SBR

The relatively high contribution of infrastructure to the total GWP of HRAP systems (45.3 as compared to 55.7 x 10^{-3} kg CO_2 equiv./m^3 during operation) is unusual given that it typically accounts for less than 10% of the environmental impact of operation in other industrial processes (Choi et al., 2018). This is a characteristic of HRAP systems due to the large amounts of material needed to construct raceways with a large surface area. Direct greenhouse gas emissions from the raceway (in the form of N_2O and CH_4) relative to the indirect emissions of infrastructure are also notably high (45.3 x 10^{-3} kg CO_2 equiv./m^3) for these systems. Nonetheless, the relatively high GWP related to infrastructure (45.3 vs. 17.8 x 10^{-3} kg CO_2 equiv./m^3 in SBRs) is compensated in the long term by lower emissions (direct and indirect) over 40 years of operation in HRAP.

The algae separation step during WWT in HRAP systems accounts for 30% of the total environmental impact of the wastewater treatment process, and roughly 80% of the operation part in terms of GWP and even more of the EP. This is mainly due to power consumption and the use of the chemical additives, PAM and PAC18%, which are necessary for flocculation and coagulation of the biomass. In addition, the environmental impact of these additives may go beyond GWP and EP: PAM degradation in the environment, for example, can lead to emissions of hazardous compounds including acrylamide (Kay-Shoemake et al., 1998; Smith et al., 1996, 1997), which are not reflected in our model since they are not yet available in LCA databases. This highlights the necessity of further research to improve this part of operation, not only for economic purposes, i.e. the high cost of chemicals, but also to reduce environmental impacts.

3.5. Sensitivity analysis of nutrient removal potential in the LCA

The eutrophication potential (EP) values presented above reflect the environmental impact of construction and operation of the HRAP and SBR facilities, but neglects the environmental benefit that comes from removing nutrients (P, N) from wastewater before discharging it to the environment. We considered the Net Environmental Benefit (NEB) (Godin et al., 2012) – the difference in EP of treated and untreated water – to assess how fluctuations in nutrient removal performance may affect the environmental impact of HRAP and SBR technologies.

Assuming that both systems may change their performance to the same extents, we performed a sensitivity analysis that considered satisfactory and unsatisfactory nutrient removal rates for HRAP and SBR. Inflow and outflow values for the well-performing HRAP and SBR systems (referred to as "good") were taken from our data (Table 5), while effluent values of HRAP and SBR performing non-satisfactorily were calculated using values 20% higher than the values of good performance. The EP of the two scenarios in Figure 6 shows that the nutrient removal performance of WWT technologies plays a much more important role than the environmental impact of facility infrastructure and operation. An SBR performing satisfactorily has a slightly higher NEB (0.0459 kg phosphate equivalent/m^2 WW) than the good performing HRAP (0.0442 kg phosphate equivalent/m^2 WW) because of the slightly lower concentrations in the effluent of the well-performing SBR that can overcompensate the considerably higher adverse effects of the infrastructure and operation of SBR. Additionally, the 20% deterioration in performance decreased NEB only by 3% for the SBR and 4% for the HRAP which means SBR is slightly less sensitive to performance changes than HRAP.

mg/L	Inflow	Outflow HRAP good	Outflow HRAP bad	Outflow SBR good	Outflow SBR bad
COD	800	80	96	28.68	34.42
TKN	67	15	18	12.35	14.82
TP	10	1	1.2	1.43	1.72

Table 4. Effluent values for sensitivity analysis

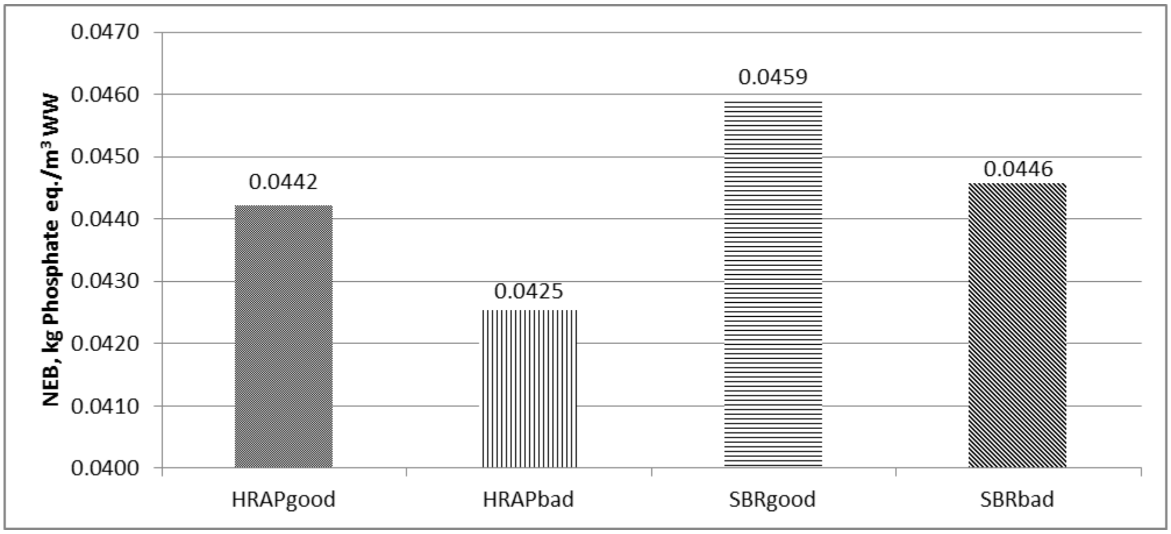

Figure 6. Results of EP for different effluent values

4. Discussion

Our study compared two rather small-scale WWT plants, a conventional bacteria-based SBR and an algae-based HRAP system. According to our results, the OPEX of HRAP is ca. half of the SBR's and the infrastructure of HRAP is slightly more expensive to be built than that of the SBR. These results are somewhat different from the numbers given in the literature. Meanwhile, our operation costs are 0.12 and 0.22 €/m^3 WW treated for HRAP and SBR, respectively, these values are in Garfí et al. (2017) much higher: 0.42 €/m^3 for HRAP and 0.79 €/m^3 for a conventional bacterial-based WWTP. This difference might be caused by the inclusion of maintenance costs, i.e. the replacement of worn out parts, as a reinvestment within CAPEX; and our calculations do not contain the personnel costs of building. Our CAPEX is also slightly higher than given in this literature: 294,799 and 246,225 €, respectively. In fact, the difference in operation costs is roughly 50% lower for HRAP than for conventional WWTPs, both according to the literature and our results. For these two systems, personnel costs of operation took about 50% of OPEX, which indicates the need for automatization, especially for small systems.

The difference in operation costs of SBR and HRAP is due to the higher energy efficiency of the HRAP. The power consumption values we obtained for the HRAP were slightly lower than for the modelled HRAP system in Garfí et al. (2017), 0.17 and 0.25 kWh/m^3, respectively, but considerably less than that of the conventional SBR (0.45 kWh/m^3). This is the other reason for higher OPEX in Garfí et al. (2017). While the largest share of the energy consumption in the SBR system relates to aeration and mixing (74%), the largest share of energy consumption in the HRAP relates to the requirements of the algae separation step followed by the mixing of the ponds.

The energy consumption of SBR can be changed, however, very quickly because the plant treats WW in a batch mode, i.e. the SBR tank has to be filled up with WW in an ordered sequence. In contrast, the HRAP works in a continuous mode and the influent WW is added to the raceway pond as it enters the plant. This mode of functioning makes the SBR plant more susceptible to changes in WW amounts and the scheduling of the reactor has to be adjusted, which may result in a higher energy consumption. If the inflow rate of WW does not allow the reactor to be run in the defined sequences, the time for filling the reactor and mixing of WW will increase significantly, while the time and energy required for aeration can be minimized. Although energy can be saved in this way, the longer mixing periods and the decreased WW amount compensate for the reduction in energy for aeration. In case of Leppersdorf, for example, the drop in flow rate from 300 to 192 m^3/day led to a proportional increase in electricity consumption from 0.45 to 0.70 kWh/m^3.

The introduction of a more effective mixing system (the LEAR) to the HRAP can further increase the difference of energy consumption between SBR and HRAP since it more than halves the energy demand for mixing with the paddle wheel. This results in a 22% lower energy consumption for the HRAP system. This difference has an important effect on GWP too: HRAP creates only one third of SBR's GWP (0.146 and 0.458 kg CO_2-equiv/m^3). Previous studies calculated higher values: 0.28-0.57 and 0.405-1.27 kg CO_2-equiv/m^3 for HRAP and for conventional systems, respectively partly because of less effective mixing and bigger systems. Especially in the case of electricity consumption, the size of the WWTP is critical: the bigger the plant, the smaller the electricity consumption (Lorenzo-Toja et al., 2015). This is why HRAP systems are particularly effective in small-scale (Garfí et al., 2017).

Meanwhile the environmental impact of infrastructure for SBR was only 5% of the impacts from operation, the impact of infrastructure for HRAP was 3.5 times bigger than that of the operation. This is because impacts from operation are almost negligible but the space requirement and the connected material input to establish the infrastructure are rather high for HRAP systems, e.g. concrete and plastic layers for the raceway. Consequently, the environmental and economic impact of building materials is considerable and the choice to select environmentally friendly and cheap construction alternatives is fundamental.

Another important source of environmental impacts was direct emissions. N_2O is a natural emission of algae metabolism and an important greenhouse gas. In addition, CH_4 and NH_3 are also emitted during WWT processes. 50% of GWP was created by direct N_2O and CH_4 emissions in SBR and 30% by direct N_2O emission in HRAP. Although direct emissions play a very important role in shaping environmental performance of WWT, their values are complicated to measure and are within wide ranges in the literature (Alcántara et al., 2015; Bao et al., 2016; Garfí et al., 2017). Thus, a reliable assessment of direct emissions requires much more detailed research.

Besides GWP, results of EP are also very important aspects of evaluating the performance of WWTPs. Our sensitivity analysis for calculating NEB highlighted the importance of cleaning performance. Our study proved that a higher removal rate of components bringing about eutrophication can in turn overcompensate less favorable results of infrastructure and operation, e.g. in the case of an effective SBR.

Finally, chemical additives, such as PAC and PAM, also result in environmental impacts, e.g. it was the second most important cost category for HRAP after personnel costs. Unfortunately, nature-based flocculants or coagulants are less effective and can be even more expensive than conventional chemicals. Consequently, research for finding effective but environmentally friendly and cheap chemicals for WWT is indispensable.

5. Conclusions

This study shows the advantages of a combined LCA and LCCA methodology to comparatively evaluate WWT technologies and identify their strengths and weaknesses.

Overall, the HRAP WWT technology proved to be more efficient both in economic and environmental terms than the SBR. In economic terms (CAPEX and OPEX) and in terms of energy balance:

- The large area requirement of algae-based systems is the greatest drawback of HRAP technology, as the economic viability/benefit of this process is dependent on land availability and cost.
- The relatively high cost and environmental impact of building HRAP infrastructures is compensated by the relatively low cost and environmental impact during operation of the wastewater treatment facility, primarily due to the higher power consumption required to operate in sequencing batch mode (and the environmental impact associated with this). The energy consumption of the HRAP system with a submerged mixing system is 22% of that of the SBR.

In terms of environmental impact (global warming and eutrophication potential):

- The GWP and EP of SBR is higher than the GWP of the HRAP. Indirect emissions linked to the higher power consumption contribute to the higher GWP of SBRs. Additionally, direct greenhouse gas emissions (primarily in the form of N_2O) are presumed to be higher in a bacteria-dominated activated sludge system than in an algae-dominated system.
- With regard to the net environmental benefit from the removal rate on EP, the HRAP was slightly less favorable than the SBR because of better removal rates of the latter.
- Just like any technology, algae-based wastewater treatment has its limitations (reviewed extensively in (Posadas et al., 2017)) and is most suitable for specific environmental conditions and WW characteristics: i.e. conditions optimal for algae growth: mild temperatures, large areas for harvesting of solar radiation, a specific range of C:N ratio, etc. Furthermore, HRAP systems have direct emissions of N_2O (Alcántara et al., 2015). Finally, harvesting algae from a highly diluted suspension (ca. 0.5 g/L) is costly and often involves the use of environmentally harmful chemicals (e.g. PAM) as discussed above (Béchet et al., 2017; Muylaert et al., 2017).

Further research will be required

- To optimize savings in material and energy flows in the building and operation of HRAPs
- To better evaluate direct emissions from both technologies.
- To include other forms of environmental impact (e.g. hazardous emissions that come from environmental degradation of chemical additives used, e.g. acrylamide).

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References

91/271/EEC, Directive of the Council of December 21, 1991 concerning urban waste water treatment, Brussels, p. 40.

Alcántara, C., Muñoz, R., Norvill, Z., Plouviez, M., Guieysse, B., 2015. Nitrous oxide emissions from high rate algal ponds treating domestic wastewater. Bioresource Technology 177, 110-117.

Amores, M.J., Meneses, M., Pasqualino, J., Anton, A., Castells, F., 2013. Environmental assessment of urban water cycle on Mediterranean conditions by LCA approach. Journal of Cleaner Production 43, 84-92.

Annavajhala, M.K., Kapoor, V., Santo-Domingo, J., Chandran, K., 2018. Comammox Functionality Identified in Diverse Engineered Biological Wastewater Treatment Systems. Environmental Science & Technology Letters 5, 110-116.

Bao, Z.Y., Sun, S.C., Sun, D.Z., 2016. Assessment of greenhouse gas emission from A/O and SBR wastewater treatment plants in Beijing, China. International Biodeterioration & Biodegradation 108, 108-114.

Béchet, Q., Plouviez, M., Chambonnière, P., Guieysse, B., 2017. 21 - Environmental impacts of full-scale algae cultivation A2 - Gonzalez-Fernandez, Cristina, in: Muñoz, R. (Ed.), Microalgae-Based Biofuels and Bioproducts. Woodhead Publishing, pp. 505-525.

Campos, J.L., Valenzuela-Heredia, D., Pedrouso, A., R, V.d., #xed, o, A., Belmonte, M., Mosquera-Corral, A., 2016. Greenhouse Gases Emissions from Wastewater Treatment Plants: Minimization, Treatment, and Prevention. Journal of Chemistry 2016, 12.

Capodaglio, A.G., 2017. Integrated, Decentralized Wastewater Management for Resource Recovery in Rural and Peri-Urban Areas. Resources 6.

Choi, S., Johnston, M., Wang, G.S., Huang, C.P., 2018. A seasonal observation on the distribution of engineered nanoparticles in municipal wastewater treatment systems exemplified by TiO2 and ZnO. Science of the Total Environment 625, 1321-1329.

Cornejo, P.K., Zhang, Q., Mihelcic, J.R., 2013. Quantifying benefits of resource recovery from sanitation provision in a developing world setting. Journal of Environmental Management 131, 7-15.

Corominas, L., Foley, J., Guest, J.S., Hospido, A., Larsen, H.F., Morera, S., Shaw, A., 2013. Life cycle assessment applied to wastewater treatment: State of the art. Water Res 47, 5480-5492.

Dutta, A., Sarkar, S., 2015. Sequencing Batch Reactor for Wastewater Treatment: Recent Advances. Current Pollution Reports 1, 177-190.

Eggimann, S., Truffer, B., Feldmann, U., Maurer, M., 2018. Screening European market potentials for small modular wastewater treatment systems - an inroad to sustainability transitions in urban water management? Land Use Policy 78, 711-725.

Fernandes, H., Jungles, M.K., Hoffmann, H., Antonio, R.V., Costa, R.H.R., 2013. Full-scale sequencing batch reactor (SBR) for domestic wastewater: Performance and diversity of microbial communities. Bioresource Technology 132, 262-268.

Garfí, M., Flores, L., Ferrer, I., 2017. Life Cycle Assessment of wastewater treatment systems for small communities: Activated sludge, constructed wetlands and high rate algal ponds. Journal of Cleaner Production 161, 211-219.

gobex, -. Estudio de Puesta en Servicio. E.D.A.R. Y Colectores en Segura de León (Badajoz). Anejo No. 12: Estudio de Explotación. Gobierno de Extramadura (gobex) and Inyges Consultores S.L., p. 23.

Godin, D., Bouchard, C., Vanrolleghem, P.A., 2012. Net environmental benefit: introducing a new LCA approach on wastewater treatment systems. Water Science and Technology 65, 1624-1631.

Hischier, R., Weidema, B., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., Frischknecht, R., Hellweg, S., Humbert, S., Jungbluth, N., Köllner, T., Loerincik, Y., Margni, M.a., Nemecek, T., 2010. Implementation of Life Cycle Impact Assessment Methods. ecoinvent report No. 3, in: Roland Hischier, Weidema, B. (Eds.). Swiss Centre for Life Cycle Inventories, St. Gallen, p. 176.

Kay-Shoemake, J.L., Watwood, M.E., Lentz, R.D., Sojka, R.E., 1998. Polyacrylamide as an organic nitrogen source for soil microorganisms with potential effects on inorganic soil nitrogen in agricultural soil. Soil Biology & Biochemistry 30, 1045-1052.

Langeveld, J., 2015. Comment on "Life cycle assessment of urban wastewater systems: Quantifying the relative contribution of sewer systems". Water Res 84, 375-377.

Lorenzo-Toja, Y., Vazquez-Rowe, I., Amores, M.J., Termes-Rife, M., Marin-Navarro, D., Moreira, M.T., Feijoo, G., 2016. Benchmarking wastewater treatment plants under an eco-efficiency perspective. Science of the Total Environment 566, 468-479.

Lorenzo-Toja, Y., Vazquez-Rowe, I., Chenel, S., Marin-Navarro, D., Moreira, M.T., Feijoo, G., 2015. Eco-efficiency analysis of Spanish WWTPs using the LCA plus DEA method. Water Res 68, 651-666.

Maga, D., 2017. Life cycle assessment of biomethane produced from microalgae grown in municipal waste water. Biomass Conversion and Biorefinery 7, 1-10.

Muylaert, K., Bastiaens, L., Vandamme, D., Gouveia, L., 2017. 5 - Harvesting of microalgae: Overview of process options and their strengths and drawbacks A2 - Gonzalez-Fernandez, Cristina, in: Muñoz, R. (Ed.), Microalgae-Based Biofuels and Bioproducts. Woodhead Publishing, pp. 113-132.

Pogade, F., Lee, M.-Y., van Afferden, M., Müller, R., 2015. O&M of Decentralized Wastewater Treatment Plants in Jordan based on International and German Standards and Practical Experiences. National Implementation Committee for Effective Decentralized Wastewater Management and Helmholtz Centre for Environmental Research, Germany, p. 46.

Posadas, E., Alcántara, C., García-Encina, P.A., Gouveia, L., Guieysse, B., Norvill, Z., Acién, F.G., Markou, G., Congestri, R., Koreiviene, J., Muñoz, R., 2017. 3 - Microalgae cultivation in wastewater, Microalgae-Based Biofuels and Bioproducts. Woodhead Publishing, pp. 67-91.

Smith, E.A., Prues, S.L., Oehme, F.W., 1996. Environmental degradation of polyacrylamides .1. Effects of artificial environmental conditions: Temperature, light, and pH. Ecotoxicology and Environmental Safety 35, 121-135.

Smith, E.A., Prues, S.L., Oehme, F.W., 1997. Environmental degradation of polyacrylamides .2. Effects of environmental (outdoor) exposure. Ecotoxicology and Environmental Safety 37, 76-91.

van Afferden, M., Cardona, J.A., Lee, M.-Y., Subah, A., Müller, R.A., 2015. A new approach to implementing decentralized wastewater treatment concepts. Water Sci Technol 72, 1923-1930.

Vikrant, K., Kim, K.H., Ok, Y.S., Tsang, D.C.W., Tsang, Y.F., Giri, B.S., Singh, R.S., 2018. Engineered/designer biochar for the removal of phosphate in water and wastewater. Science of the Total Environment 616, 1242-1260.

Zhang, Z.Z., Cheng, Y.F., Xu, L.Z.J., Bai, Y.H., Xu, J.J., Shi, Z.J., Zhang, Q.Q., Jin, R.C., 2018. Transient disturbance of engineered ZnO nanoparticles enhances the resistance and resilience of anammox process in wastewater treatment. Science of the Total Environment 622, 402-409.

Internet references

http 1: https://www.alibaba.com/product-detail/Polyacrylamide-price_60657407133.html?spm=a2700.7724857.main07.11.255a752eeZh7Di&s=p. Date of download: 05.11.2017 http 2: https://www.alibaba.com/product-detail/GB-15892-2009-water-treatment-chemicals_60669823722.html?spm=a2700.7724857.main07.12.1a214dd2rzKNY2&s=p. Date of download: 05.11.2017

http 3: http://sdw.ecb.europa.eu/browseTable.do?node=9691107. Date of download: 30.01.2018

http 4: Euro area bank interest rate statistics: November 2017. https://www.ecb.europa.eu/press/pdf/mfi/mir1711.pdf?75d979235949e5fef6754687f01b69bf. Date of download: 30.01.2018

http 5: http://www.mapama.gob.es/es/estadistica/temas/estadisticas-agrarias/economia/encuesta-precios-tierra/. Date of download: 05.11.2017