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Unravelling the environmental and economic impacts of innovative technologies for the enhancement of biogas production and sludge management in wastewater systems

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Abstract

The retrofitting of wastewater treatment plants (WWTPs) should be addressed under sustainability criteria. It is well known that there are two elements that most penalize wastewater treatment: (i) energy requirements and (ii) sludge management. New technologies should reduce both of these drawbacks to address technical efficiency, carbon neutrality and reduced economic costs.

In this context, the main objective of this work was to evaluate two real plants of different size in which major modifications were considered: enhanced recovery of organic matter (OM) in the primary treatment and partial-anammox nitrification process in the secondary treatment. Plant-wide modelling provided an estimate of the input and output flows of each process unit as well as the diagnosis of the main performance indicators, which served as a basis for the calculation of environmental and economic indicators using the LCA methodology.

The combination of high-rate activated sludge (HRAS) + partial nitrification Anammox can decrease the environmental impacts by about 70% in the climate change (CC) category and 50% in the eutrophication potential (EP) category. Moreover, costs can be reduced by 35-45% depending on the size of the plant. In addition, the enhanced rotating belt filter (ERBF) can also improve the environmental profile, but to a lesser extent than the previous scenario, only up to 10% for CC and 15% for EP. These positive results are only possible considering the production of energy through biogas valorization according to the waste-to-energy scheme.

Keywords: high rate activated sludge (HRAS), enhanced rotating belt filter (ERBF) integrated fixed film activated sludge (IFAS), life cycle assessment (LCA), scale-up analysis, wastewater treatment modelling.

Nomenclature

AD	Anaerobic Digestion
AS	Activated Sludge
CC	Climate Change
CHP	Combined Heat and Power
COD	Chemical Oxygen Demand
EP	Eutrophication Potential
ERBF	Enhanced Rotating Belt Filter
EROI	Energy return on investment
FD	Fossil Depletion
FET	Freshwater EcoToxicity
FU	Functional Unit
GHG	Greenhouse Gas
HRAS	High Rate Activated Sludge
HT	Human Toxicity
IFAS	Integrated Fixed Film Activated Sludge
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MET	Marine EcoToxicity
NEB	Net Environmental Benefit
OD	Ozone Depletion
OM	Organic Matter
PC	Primary Clarifier
PMF	Particulate Matter Formation
PN-AMX	Partial Nitritation-AnaMmoX
TA	Terrestrial Acidification
TET	Terrestrial EcoToxicity
UASB	Upflow Anaerobic Sludge Blanket
WD	Water Depletion
WWTPs	WasteWater Treatment Plants

1. Introduction

Traditionally, wastewater treatment plants (WWTPs) have been considered as end-of-line elements. Their main objective was to discharge a treated effluent into the aquatic environment and avoid problems related to eutrophication and ecotoxicity (Cieślik and Konieczka, 2017). In this context, conventional nitrification-denitrification has been applied to remove nutrients, mainly nitrogenous compounds, through the oxidation of organic matter (OM) provided that a suitable C/N ratio between 5 and 20 is maintained in the influent (Wu et al., 2009). In last years, increasingly strict environmental regulations introduce other aspects to be taken into account such as the removal of organic micro-pollutants, gas emissions and the efficient management of sludge, which is generally considered as an environmental and economic burden (Kamali et al., 2019). The conventional nitrification-denitrification consumes a greater amount of energy in aeration to convert ammoniacal nitrogen into nitrite and its subsequent oxidation into nitrate (Gude, 2015). The electricity demand of these conventional technologies can vary from 0.3 kWh/m^3 to 0.6 kWh/m^3 (Wan et al., 2016). In addition, if it is necessary to add an external source of OM, the operational costs can increase considerably (Morales et al., 2015).

With the main objective of improving the removal of nutrients, new processes and technologies have recently been exploited. One of the most important is the autotrophic nitrogen removal (Anammox) process, which is characterized by lower temperature, between 10-20 ^0C (Tao et al., 2014), low carbon-to-nitrogen ratio (Lackner et al., 2014) and a low sludge production (Pedrouso et al., 2018). Different alternatives such as SHARON-Anammox (Single-reactor High Actively ammonia Removal Over Nitrate)

(Kampschreur et al., 2008), CANON (Complete Autotrophic Nitrogen-removal Over Nitrate) (Vázquez-Padín et al., 2010) or IFAS (Integrated Fixed Film Activated Sludge) (Malovanyy et al., 2015) have been considered. However, this new process cannot work with a high percentage of solids or COD (Lackner et al., 2014). It is for this reason that the highest possible OM fraction should be recovered in the primary treatment, as a preliminary stage of the partial nitrification-Anammox process. For this purpose, enhanced rotating belt filters (ERBFs) or high rate activated sludge (HRAS) (Jimenez et al., 2015; Ruiken et al., 2013) and other more conventional technologies such as upflow anaerobic sludge blanket (UASB) (Kujawa-Roeleveld et al., 2006) could be implemented.

In this regard, innovative wastewater treatment schemes have been proposed to address more complex challenges. These modifications range from retrofitting stages, encompassing novel units in conventional processes, to substantial modifications of the WWTP configurations (Grobelak et al., 2019; Rajasekhar et al., 2020). However, it is difficult to assess how these new technologies (many of them still at a pilot level) work in a real facility. Therefore, it is necessary to use tools to model, optimize and select the most appropriate plant layout for each particular scenario. It is not possible to undertake the construction of new facilities unless the previous techno-economic and environmental studies have been rigorously conducted.

In this context, a plant-wide modelling and simulation study of the different innovative configurations may provide additional insight on the compatibility of the above discussed technologies. Several plant-wide studies have been conducted to evaluate treatment schemes, technology retrofitting or control strategies (Flores-Alsina et al., 2008; Gernaey et al., 2014). In addition, modelling studies of innovative technologies have been conducted to evaluate their implementation in conventional plants (Behera et al., 2019; Boiocchi et al., 2019; Wang et al., 2017). Many of these studies focused only on techno-economic feasibility lacking environmental aspects of such technologies which is the primary focus of this manuscript.

Today, thanks to advances and availability of computational power, highly demanding computational tasks such as plant-wide simulation can be performed. To evaluate the environmental profile of these new configurations, the life cycle assessment (LCA) methodology has proven to be a good alternative because it allows the calculation of environmental impacts over the entire life of a product or a process (ISO 14040, 2006). The LCA methodology has been applied not only to the evaluation of conventional WWTPs and alternative wastewater and sludge technologies (Dong et al., 2014; Singh et al., 2018); but also the use of reclaimed water from WWTP (Kamble et al., 2017; Opher and Friedler, 2016). In addition, the combination of environmental and economic indicators in WWTPs has also been considered for the definition of sustainability criteria in the selection of treatment technologies (Lorenzo-Toja et al., 2016; Resende et al., 2019).

The main objective of this study is to combine the approach of OM recovery to maximize biogas production and a partial nitrification-Anammox to remove nitrogen in the treated effluent as the scenario to be implemented in two real WWTPs of different sizes (medium and large) in different European countries (Spain and Denmark). With the outcomes of the modelling stage, an environmental and economic analysis was conducted to assess whether the wastewater treatment schemes based on this perspective are better than conventional wastewater treatment strategies.

2. Materials and methods

2.1. Methodology

The IWA task group has developed new models and tools for the evaluation of WWTPs such as the Benchmark Simulation Model No.2 (BSM2), which is being widely used as a framework for plant-wide analysis (Gernaey et al., 2014; Saagi et al., 2017). This study addresses several models developed from BSM2 and its interfaces. Table 1 summarizes the modelling approach used for both conventional and emerging technologies. As part of the simulation strategy, the plant-wide model is initialized using a sequential approach to avoid model convergence issues (Behera et al., 2019; Solon et al., 2017). A closed loop stable state simulation is then performed using stiff differential solver like *ode15s* in MATLAB-Simulink software (2016a). In addition, further information (related to the underlying concepts of the primary technologies and the different plant-wide layouts representing novel retrofitting solutions considered in the model) can be found in the Supplementary Material. For the calculation of environmental impacts, the Life Cycle Analysis (LCA) methodology was applied according to the standardized method defined by ISO 14040 (2006).

>TABLE 1<

2.2. Goal and scope definition

In this study, two real WWTPs (one medium and one large) were considered as the basic configuration for implementing the technologies previously proposed. It is important to know how improve biogas production and the efficiency of the wastewater schemes depending on the technology considered. One WWTP is located in Denmark (Avedøre) and is designed for 265,000 equivalent inhabitants with a flow of about 72,000 m^3/d. The second plant is located in Valladolid (Spain). The flow is 213,000 m^3/d with a population of 1,000,000 equivalent inhabitants. All the flows of energy and materials, as well as the emissions associated with the operation of the WWTPs, were considered and quantified in detail.

2.3. Functional unit (FU)

The functional unit (FU) is defined as the quantification of all inputs and outputs of the product or system under evaluation. The FU should be consistent with the main goal of the study (ISO 14040, 2006). The main function of a WWTP is treated wastewater and 1 m^3 of treated wastewater is the most common FU (Resende et al., 2019; Schaubroeck et al., 2015). However, in this case, electricity production is the main reason for implementing these innovative technologies. In this context, 1 kWh of energy produced in a combined heat and power unit (CHP) was defined as FU (Singh et al., 2020).

2.4. System boundaries and definition of the system under assessment

The system boundaries were approached from a gate-to-gate perspective (ISO 14040, 2006). In a WWTP, the main environmental impacts correspond to the operation phase (Shiu et al., 2017; Tabesh et al., 2019) while those of the decommissioning and construction phases can be considered minor. The main reason is that the lifetime of a wastewater treatment plant can vary between 25 and 50 years, so while the impact of the operational phase is added up year by year, the impact of the construction will be divided by the number of years that the WWTP is in operation (Buonocore et al., 2018; Lorenzo-Toja et al., 2016). Further details on the operational conditions and input parameters of the different WWTPs can be found in the Supplementary Material (Table S1 and S2).

>FIGURE 1<

In this case, the base scenario (Scenario 0) is the conventional scheme of a WWTP. Both WWTPs consist of a pre-treatment followed by a primary clarifier (PC) and an activated sludge (AS) process with nitrogen removal in the water line. The sludge line consists of a thickener, an anaerobic digestion (AD) unit and a dewatering system. In addition, biogas is transformed into electricity and heat in a CPH unit. The main difference between the two plants is sludge disposal. In the case of the Valladolid plant, the sludge is applied on agricultural land, while at the Avedøre plant, it is incinerated. Therefore, these alternatives are considered for the environmental profile. These conventional technologies are replaced by innovative technologies. Thus, two scenarios were studied and compared with the base case:

Scenario 1 consists of the combination of ERBF and IFAS in the water line. Scenario 2 is based on HRAS followed by an IFAS unit in the water line. The sludge line is the same for all scenarios and does not change. As mentioned above, the only change is the final disposal of the sludge (incineration or land application).

2.5. Inventory data acquisition

Data collection is the most time-consuming stage and is linked to the quality of the results obtained in the environmental analysis (Nguyen et al., 2020). In this case, the inventory considered real data from the WWTPs (primary data), while secondary data are obtained from the modelling results (Niero et al., 2014). Accordingly, data associated with COD, nitrogen, phosphorus or heavy metals were obtained from the available information reported by the managers of Valladolid and Avedøre facilities (Aguas de Valladolid, 2017; BIOFOS, 2017). More information on these data can be found in Table S1 of the supplementary material.

These influent parameters were implemented in the model to obtain data related to methane production, energy consumption or effluent parameters. The data obtained for the simulation of the model (secondary data) are presented for each scenario (Table S3 to Table S8). Finally, the primary and secondary data were completed with the Ecoinvent v3.5 database (Wernet et al., 2016). Moreover, several simplifications were considered for the life cycle inventory data, especially those of foreground processes, as detailed below:

Two different electricity country mix were selected due to the different location of the WWTPs. Spanish and Denmark country mix were updated and the medium-voltage electricity used in WWTPs was modelled, including transport losses (Dones et al., 2007).

Euro 4 trucks with a capacity between 12-32 t were selected to transport chemicals and sludge. In addition, an average distance of 50 km was considered for the transport of chemicals and sludge (Morera et al., 2020). For the sludge application into the soil, emissions to air (N_2O and NO_3) and water (NO_3^- and PO_4^{-3}) were calculated and taken into account in the final environmental profile (Bruun et al., 2006). Finally, for biogas leaks associated with CH_4, CO_2 and H_2S emissions, a value of 1.5% was considered to calculate emissions to air (Lijó et al., 2017).

The inventories (main inputs and outputs) are shown in Tables 2 and 3 per functional unit considered (1 kWh of energy produced). That is, the results of the model simulation were referred to the FU and implemented in the environmental software to calculate the environmental impacts.

>TABLE 2<

>TABLE 3<

2.6. Environmental and economic indicators

In this study, only the classification and characterization steps within the LCA methodology were taken into account. Two methods were selected to calculate the most representative impacts of a WWTP. Eutrophication potential (EP) was calculated with CML 2001 method (Guinée, 2002) whereas climate change (CC), ozone depletion (OD), terrestrial acidification (TA), particulate matter formation (PMF), human toxicity (HT), marine ecotoxicity (MET), terrestrial ecotoxicity (TET), freshwater ecotoxicity (FET), water depletion (WD) and fossil depletion (FD) were calculated the ReCiPe Midpoint (H) method (Huijbregts et al., 2017). The software SimaPro 9.0 was used for the implementation of the inventories.

The reason for choosing two methods is how to assess the impact linked to the COD concentration. In the ReCiPe method there is no characterization factor, whereas in the CML 2001 method there is a characterization factor for this parameter. Since COD is a discharge limiting parameter, it is important to consider its contribution in WWTP profile.

As for the economic indicators, only the costs associated to the operational phase was considered. These economic indicators are related to sludge management, electricity and chemical consumption. Biogas is considered as a benefit; thus, it is computed for the calculation of revenues (Li et al., 2019). As in the environmental analysis, construction costs were not considered because they represent a minor contribution to the total costs (Termes-Rifé et al., 2013).

3. Results

3.1. Life cycle environmental profile for each new wastewater treatment scheme

Firstly, the environmental results are presented according to the size of the plant, observing the contribution that each subsystem makes to the total environmental profile. Figure 2 shows the contribution per subsystem for the Avedøre case. In the conventional case (Scenario 0), the main contributor to the impact in all environmental categories except EP, TET and HT is the CAS with nitrogen removal unit followed by dewatering. The negative effect is associated with high electricity consumption and direct emissions from the treated effluent as it presents residual concentrations of nitrogen, phosphorus and heavy metals. The discharge of these pollutants in large quantities can cause mortality of aquatic species. In the case of heavy metals and metalloids, there are compounds that are more toxic, and their discharge is more limited such as arsenic or mercury. For this reason, the impacts of these substances are studied in more detail in the Supplementary Material (Tables S9 and S10).

In the dewatering unit, emissions are associated with the consumption of electricity and polyelectrolyte that is used as an additive to improve the dewatering of the sludge. Depending on the impact category, the negative effect related to polyelectrolyte can vary from 83% in the TET category to 52% in the WD category (Figure S2 of the Supplementary Material).

The incineration unit is the unit causing the major impact in the EP and HT categories due to the disposal of ashes that may contain hazardous contaminants such as heavy metals. Other units such as PC or thickener have a negligible impact (Figure 2a).

In Scenario 1 (incorporation of ERBF technology), the main impact is distributed as in Scenario 0 (CAS unit) and the reasons for the negative contribution are the same. However, the incorporation of this type of treatment cannot be considered irrelevant and has a contribution of between 2% in the FET or MET categories and 11% in the TET category. This negative effect is related to indirect emissions from chemical production (Figure 2b and Figure S3).

Finally, for Scenario 2 (integration of HRAS technology), the environmental profile changes. In this case, the dewatering unit is the main responsible of the impacts in all categories except FET and MET. In these categories, the IFAS unit is responsible for the impact due to direct emissions associated with the impact of nutrients present in the effluent discharged to the environment. Finally, as in Scenario 0, the impact caused by HRAS can be considered negligible in all categories.

>FIGURE 2<

Figure 3 shows the environmental profile of the large plant for each scenario considered. In this case, the main difference is the incorporation of a composting unit followed by land application. This final disposal has a negative effect on the PMF and TA categories due to air emissions (Table 3a). In addition, as in the medium plant, the main factor contributing to the impact is the CAS unit, as electricity consumption in energy-dependent categories such as CC, WD, OD, HT and FD. In categories that do not depend on energy consumption (MET, FET, EP and TET), the negative effect is caused by the discharge of the effluent into the environment. This effluent may contain hazardous substances such as heavy metals that may be harmful to aquatic or terrestrial species (Table S10). In other units such as cogeneration, PC or thickening, the impacts are very small (Figure 3a). For Scenario 1 (ERBF), the impacts are very similar to Scenario 0 (conventional scenario) (Figure 3b). The main difference is that the impact on this new unit cannot be considered negligible and ranges from 3% in the FET or MET categories to 11% in FD (Figure S7 of the Supplementary Material). Finally, for Scenario 3 (HRAS), as in a medium plant, the impact of HRAS can be considered negligible. Therefore, in this case, IFAS consumes less energy due to the recovery of OM. This implies that the impact on energy dependence decreases, while in categories that do not depend on energy consumption, the negative effect is the same as in Scenario 1. In this Scenario, the dewatering unit becomes more important in terms of impact in energy-dependent categories due to the electricity and polyelectrolyte consumption (Figure 3c).

>FIGURE 3<

The first environmental analysis (Figures 2 and 3) provides an insight into the impact that new technologies have on the profile of the WWTP. A priori, the worst environmental profile would correspond to the combination of ERBF, while the impact associated with the HRAS unit can be considered not significant in the impact categories evaluated. An interesting step forward to make a conclusive decision on the selection of the most suitable configuration is to compare the different configurations with each other.

3.2. Environmental comparison for different scenarios in both WWTP analysed

In this analysis, only the categories of CC and EP were evaluated due to their special relevance in the environmental profile of WWTPs (Rodriguez-Garcia et al., 2011).

CC is related to the energy production and consumption, while EP is related to the quality of the effluent discharged into water courses.

When comparing both plants in terms of these impact categories (Figures 4 and 5), the environmental profile decreases when the HRAS unit is incorporated followed by the IFAS reactor. The reduction for the CC category, if the values are compared with the conventional scenario, is approximately 68% for the medium plant (Figure 4a) and 51% for the large plant (Figure 5a). The main reason for the reduction in the CC category is the increase in biogas production and the reduction in energy consumption.

In addition, for the EP category, the impacts can also decrease by incorporating Scenario 2 (HRAS + IFAS unit): 48% for the small plant (Figure 4b) and 30% for the large plant (Figure 5b). The main difference between the environmental profiles for the different scales is that plants have different energy consumption, production or consumption of chemicals. Moreover, the wastewater composition (COD, nutrients or heavy metals) that are treated in each WWTP are different (Table S1 of the Supplementary Material). However, in spite of the variability found for both plants, the new schemes appear to have a better environmental profile regardless the size of the plant.

>FIGURE 4<

>FIGURE 5<

3.3. Influence of the plant size on environmental impacts

Finally, the incorporation of these new schemes for both plant sizes was compared. As seen above, the best impacts are presented for Scenario 2 in both plant sizes. Although, a priori, the reduction of impacts is more noticeable in the large plant than in the medium scheme, if the categories are analysed, for the CC category, Scenario 2 has more impact reduction in the medium plant than in the large plant. On the contrary, in the large plant, the incorporation of the ERBF scheme has better results than in the medium plant (Figure 6a). For the EP category, good effluent quality is achieved at the large plant with the incorporation of these technologies. However, although in the medium plant, the reduction is not as great as in large plants, better effluent quality is also obtained (Figure 6b). Therefore, although the plants are located in different countries, Scenario 2 (HRAS + IFAS unit) showed a reduction in the CC and EP categories (Figure 6). Therefore, the scale or location of the plant does not influence the incorporation of these new technologies in a WWTP, since, in both cases, these technologies could considerably reduce the environmental impacts.

>FIGURE 6<

3.4. Economic results

Table 4 shows the economic results for each scenario considered. As in the environmental results, Scenario 2 (HRAS followed by IFAS technology) shows the best economic results for both plant sizes. The implementation of this configuration can reduce the cost between 70% for large plants and 45% for medium plants. Biogas production increases in this scheme and the incorporation of IFAS technology can reduce aeration requirements by 38%. In addition, in Scenario 1 (ERBF followed by IFAS configuration), costs also decrease compared to Scenario 0 (conventional case) by about 19% for the large plant and 23% for the medium plant. In this case, the costs associated with the consumption of chemicals are higher than in the other scenarios. However, aeration electricity may decrease due to the incorporation of the IFAS unit.

As for the final disposal of sludge, incineration has more costs related to electricity consumption than the composting unit (Kelessidis and Stasinakis, 2012). However, the amount of sludge in medium and large plants is not the same; therefore, the costs related to sludge management are higher in the large plant than in the medium plant. But when these technologies are incorporated, the costs associated with sludge disposal can be reduced by 15% for Scenario 1 to 32% for Scenario 2. In general, the incorporation of Scenario 2 can lower all operational costs and Scenario 1 can reduce the cost associated with final sludge disposal and electricity consumption. Although chemical costs will increase, this increase is not reflected in total operating costs.

>TABLE 4<

4. Discussion

4.1. Evaluation of the efficiency of different schemes

In WWTPs, it is important to consider the water-energy nexus to make systems more efficient. The main objective of incorporating these innovative technologies is to improve electricity production because biogas is considered a green energy and emissions to the atmosphere are lower than in non-renewable energies (Nair et al., 2014). In this regard, an indicator called energy return on investment (EROI) was calculated. This indicator relates the energy produced in a cogeneration unit between the energy consumed to treat wastewater and sludge and can be calculated by Equation [1]. However, if this indicator is less than 1, WWTPs are not self-sufficient in terms of energy production (Colosi et al., 2015). Therefore, it must be ensured that the incorporation of innovative technologies is more efficient than conventional systems.

$$\mathrm{EROI} = \frac{\text{Electricity produced}}{\text{Electricity consumed}} \qquad [1]$$

Large plants perform worse in terms of electricity production than medium-sized plants. In large plants, Scenario 0 (conventional scenario) has an efficiency of around 0.33, with the incorporation of innovative schemes, the EROI can be increased by 0.38 for Scenario 1 (ERBF + IFAS) and by 0.69 for Scenario 2 (HRAS + IFAS). The EROI values for medium plant are better even becoming self-sufficient in the Scenario 2. The values are 0.72 for Scenario 0, 0.79 for Scenario 1 and 3.75 for Scenario 2. This is to say, if the medium plant introduces HRAS followed by the IFAS configuration, it could not depend on the grid electricity. In addition, in large plants, this scheme only needs about 31% of the energy from the grid; thus, fossil CO_2 emissions can be reduced.

Energy reduction associated with the Anammox process or enhanced biogas production has been reported at laboratory scale (Cao et al., 2020; Sancho et al., 2019). A similar configuration of the RBF + PC+ denitrification process was evaluated by Gikas (2017), who reported a reduction in electricity consumption of about 85%. This value is close to the scheme of the ERBF + IFAS reactors in the medium plant. It is true that these new schemes to reduce energy consumption and enhance biogas production are still being implemented and, time is needed to assess their performance at full scale.

Finally, it is important to note that this energy benefit can reduce indirect energy-related emissions. However, this does not mean that the impacts on the CC category will be zero. In this category, direct air emissions from other units, such as IFAS or AD units, should be considered.

4.2. Trade-off analysis of eutrophication impact category

As mentioned before, eutrophication is one of the most representative impact categories in WWTPs. Eutrophication can generate toxicity problems and even mortality of different aquatic species due to the amount of nutrients (nitrogen and phosphorus) present in wastewater (Zang et al., 2015).

It is estimated that the implementation of the conventional nitrification-denitrification process decreases potential eutrophication by 54-58% (Larsen et al., 2007). To evaluate the IFAS technology, an indicator called Net Environmental Benefit (NEB) was estimated (Godin et al., 2012) according to Eq. 2. The NEB indicator relates the discharge of wastewater into the environment without any type of treatment (PI_{NO}) with the treatment scenario (PI_{TW}) and, finally, the impacts linked to the technology or WWTP during its useful life (PI_{SLC})

$$\mathrm{NEB} = [\mathrm{PI_{NO}} - \mathrm{PI_{TW}}] - \mathrm{PI_{SLC}} \qquad [2]$$

When the PN-Anammox process is included in the WWTPs, the results of nitrogen removal increase in comparison with the conventional case. These removal percentages range from 70% for large plants to 86% for medium plants, which leads to an improvement of between 10 and 20%.

It is important to note that several technologies have been developed to apply the partial nitrification-anammox process for the treatment of domestic wastewater. Ji et al. (2020) reported a nitrogen removal of about 89% using a novel simultaneous nitrogen and phosphorus process consisting of an anammox, endogenous partial-denitrification and denitrifying phosphate removal in an SBR. Gu et al. (2018) studied the feasibility of incorporating an Anammox process in a conventional WWTP and reported a nitrate removal of 87%. For the treatment of the effluent from the AD unit, this process showed better results and slow growth of biomass, so the amount of sludge can be considered not significant (Morales et al., 2015). Therefore, a priori, the partial nitrification-Anammox can replace the conventional nitrification-denitrification according to the efficiency of nitrogen removal and energy consumption.

4.3. Mapping the environmental impact of electricity from WWTPs

When analyzing the issue of the water-energy nexus in a WWTP under the LCA approach, it can be observed that the energy produced in the cogeneration unit is used in the plant itself. This energy can replace electricity from the grid and is considered as green energy. The use of fossil energy implies an unsustainable source of electricity and heat for wastewater treatment. Combining the fact that WWTPs may not be energy self-sufficient with the importance of energy source in terms of energy footprint, the most natural step would be to assess how the electricity mix affects sustainability when assessed through the LCA method (Barragán-Escandón et al., 2017).

In this study, the WWTPs are located in different European Countries, so it is interesting to observe how the environmental profile of 1 kWh of energy produced in Spain or Denmark changes. Only, the energy-dependent categories (CC, OD, PMF, TA, HT and FD) were evaluated in this case (Figure 7). Denmark has better environmental profile in terms of energy production than Spain. This is because in Denmark about 73% of energy comes from renewable wind and biomass (Danish Energy Agency, 2018). However, in Spain, renewable energy is only 44% (REE, 2018), which means that emissions related to fossil CO_2 are higher in Spain than in Denmark. Thus, wastewater treatment in Spain pollutes more than in Denmark. For this reason, it is very important to have new wastewater treatment systems that consume less energy from the grid and produces more green energy.

>FIGURE 7<

4.4. Study of sludge management

In this study, the final disposal of sludge varies according to the country selected. In Spain, the most common method is land application, while in Denmark, the most common disposal is incineration technology. It is therefore important to know how to change the environmental and economic impacts if one or the other alternative is selected. Incineration is a more expensive alternative to land application due to electricity consumption (Tomei et al., 2016).

However, incineration is not considered environmentally friendly due to the fossil CO_2 emissions in the energy-dependent categories, while composting followed by land application is considered the worst option in the categories that depend on soil emissions associated with heavy metals (Yoshida et al., 2018). But, as mentioned before, in Denmark these emissions are lower than in Spain. In addition, the composting process have air emissions considered GHG emissions such as N_2O or CH_4 (Table 2b and 3b). Some studies show that direct N_2O emissions can be even more harmful than indirect fossil CO_2 emissions (Rodriguez-Caballero et al., 2014).

To make this comparison reliable, as there are different plant sizes, incineration was considered in the large plant and composting followed by land application was included in the medium plant. Only the CC category was evaluated because it is the category most affected by GHG emissions and electricity consumption. In Denmark, incineration is the best option because land application can increase GHG emissions by 65% associated with N_2O, CH_4 and CO_2 emissions. However, at the plant in Spain, the situation is the opposite. This does not mean that incineration and land application are the best alternatives for treating sludge. Beyond these options, ongoing research is devoted to improve the final management of the sludge such as hydrothermal-pyrolysis

(Lishan et al., 2018) or addition of biopolymers for sludge dewatering (Guo and Wen, 2020).

5. Conclusions

The retrofitting of wastewater treatment plants (WWTPs) should be addressed under sustainability criteria. It is well known that there are two elements that most penalize wastewater treatment: (i) energy requirements and (ii) sludge management. New technologies should reduce both drawbacks to address technical efficiency, carbon neutrality and reduced economic costs. In this study, a number of technologies were modelled, two based on OM recovery (HRAS and ERBF) to improve biogas production and another aiming at nitrogen removal (IFAS). Economic and environmental indicators of different plant sizes (one medium and one large) were evaluated and these new schemes: (i) ERBF + IFAS and (ii) HRAS + IFAS, were compared with a conventional scheme (PC + CAS with nitrogen removal).

These schemes based on OM recovery followed by partial nitrification-Anammox showed better environmental and economic results than conventional schemes due to higher biogas production and lower energy consumption. Furthermore, the incorporation of the IFAS unit improved the quality of the effluent in terms of nutrient removal. Although these technologies are more complex than conventional ones, they also showed a better economic profile despite the size of the plant. These positive results are only possible considering the production of energy through biogas valorization according to the waste-to-energy scheme.

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References

Aguas de Valladolid, 2017. Estación depuradora de aguas residuales.

Barragán-Escandón, A., Terrados-Cepeda, J., Zalamea-León, E., 2017. The role of renewable energy in the promotion of circular urban metabolism. Sustain. 9, 1–29. https://doi.org/10.3390/su9122341

Batstone, D.J., Keller, J., Angelidaki, I., Kalyuzhnyi, S.V., Pavlostathis, S.G., Rozzi, A., Sanders, W.T., Siegrist, H., Vavilin, V.A., 2002. The IWA Anaerobic Digestion Model No 1 (ADM1). Water Sci. Technol. 45, 65–73. https://doi.org/https://doi.org/10.2166/wst.2002.0292

Behera, C.R., Al, R., Gernaey, K. V., Sin, G., 2019. A process synthesis tool for WWTP – Applying to preliminary stage design and evaluation of plant layouts. Submitt. to Chem. Eng. Res. Des.

Behera, C.R., Santoro, D., Gernaey, K.V., Sin, G., 2018. Organic carbon recovery modeling for a rotating belt filter and its impact assessment on a plant-wide scale. Chem. Eng. J. 334. https://doi.org/10.1016/j.cej.2017.11.091

BIOFOS, 2017. Environmental Review.

Boiocchi, R., Behera, C.R., Sherrat, A., DeGroot, C.T., Gernaey, K. V., Sin, G., Santoro, D., 2019. Dynamic model validation and advanced polymer control for rotating belt filtration as primary treatment of domestic wastewaters. Submitt. to Chem. Eng.

Sci.

Bruun, S., Hansen, T.L., Christensen, T.H., Magid, J., Jensen, L.S., 2006. Application of processed organic municipal solid waste on agricultural land - A scenario analysis. Environ. Model. Assess. 11, 251–265. https://doi.org/10.1007/s10666-005-9028-0

Buonocore, E., Mellino, S., De Angelis, G., Liu, G., Ulgiati, S., 2018. Life cycle assessment indicators of urban wastewater and sewage sludge treatment. Ecol. Indic. 94, 13–23. https://doi.org/10.1016/j.ecolind.2016.04.047

Cao, S., Lu, D., Phua, K.M., Yan, W., Le, C., Tao, G., Zhou, Y., 2020. Organics transformation and energy production potential in a high rate A-stage system: A demo-scale study. Bioresour. Technol. 295, 122300. https://doi.org/10.1016/j.biortech.2019.122300

Cieślik, B., Konieczka, P., 2017. A review of phosphorus recovery methods at various steps of wastewater treatment and sewage sludge management. The concept of “no solid waste generation” and analytical methods. J. Clean. Prod. 142, 1728–1740. https://doi.org/10.1016/j.jclepro.2016.11.116

Colosi, L.M., Resurreccion, E.P., Zhang, Y., 2015. Assessing the energy and environmental performance of algae-mediated tertiary treatment of estrogenic compounds. Environ. Sci. Process. Impacts 17, 421–428. https://doi.org/10.1039/c4em00541d

Danish Energy Agency, 2018. Denmark’s Energy and Climate Outlook 2018: Baseline Scenario Projection Towards 2030 With Existing Measures (Frozen Policy).

Dones R., Bauer C., Bolliger R., Burger B., Faist Emmenegger M., Frischknecht R., Heck

T., Jungbluth N., Röder A., T.M., 2007. Life cycle inventories of energy systems: results of current systems in Switzerland and other UCTE countries. Ecoinvent Rep.5. https://doi.org/10.1007/s11367-014-0838-7

Dong, J., Chi, Y., Tang, Y., Wang, F., Huang, Q., 2014. Combined life cycle environmental and exergetic assessment of four typical sewage sludge treatment techniques in China. Energy and Fuels 28, 2114–2122. https://doi.org/10.1021/ef4024146

Flores-Alsina, X., Rodríguez-Roda, I., Sin, G., Gernaey, K. V., 2008. Multi-criteria evaluation of wastewater treatment plant control strategies under uncertainty. Water Res. 42, 4485–4497. https://doi.org/10.1016/j.watres.2008.05.029

Gernaey, K. V., Jeppsson, U., Vanrolleghem, P.A., Copp, J.B., 2014. Benchmarking of Control Strategies for Wastewater Treatment Plants, Scientific and Technical Report No. 23. IWA Publishing. https://doi.org/10.2166/9781780401171

Gikas, P., 2017. Towards energy positive wastewater treatment plants. J. Environ. Manage. 203, 621–629. https://doi.org/10.1016/j.jenvman.2016.05.061

Godin, D., Bouchard, C., Vanrolleghem, P.A., 2012. Net environmental benefit: Introducing a new LCA approach on wastewater treatment systems. Water Sci. Technol. 65, 1624–1631. https://doi.org/10.2166/wst.2012.056

Grobelak, A., Grosser, A., Kacprzak, M., Kamizela, T., 2019. Sewage sludge processing and management in small and medium-sized municipal wastewater treatment plant-new technical solution. J. Environ. Manage. 234, 90–96. https://doi.org/10.1016/j.jenvman.2018.12.111

Gu, J., Yang, Q., Liu, Y., 2018. Mainstream anammox in a novel A-2B process for energy-efficient municipal wastewater treatment with minimized sludge production. Water Res. 138, 1–6. https://doi.org/10.1016/j.watres.2018.02.051

Gude, V.G., 2015. Energy and water autarky of wastewater treatment and power generation systems. Renew. Sustain. Energy Rev. 45, 52–68. https://doi.org/10.1016/j.rser.2015.01.055

Guinée, J., 2002. Handbook on Life Cycle Assessment Operational Guide to the ISO Standards. Environ. Impact Assess. Rev. 23, 129–130. https://doi.org/10.1016/S0195-9255(02)00101-4

Guo, J., Wen, X., 2020. Performances and mechanisms of sludge dewatering by a biopolymer from piggery wastewater and application of the dewatered sludge in remediation of Cr(VI)-contaminated soil. J. Environ. Manage. 259, 109678. https://doi.org/10.1016/j.jenvman.2019.109678

Guo, L., Vanrolleghem, P.A., 2014. Calibration and validation of an activated sludge model for greenhouse gases no. 1 (ASMG1): Prediction of temperature-dependent N2O emission dynamics. Bioprocess Biosyst. Eng. 37, 151–163. https://doi.org/10.1007/s00449-013-0978-3

Henze, M., Gujer, W., Mino, T., van Loosdrecht, M.C.M., 2000. Activated sludge models ASM1, ASM2, ASM2d and ASM3. IWA publishing.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D.M., Hollander, A., Zijp, M., van Zelm, R., 2017. ReCiPe 2016 v1.1.

ISO 14040, 2006. International Organization for Standardization, ISO 14040. Environmental Management-Life Cycle Assessment - Principles and Framework, Geneve.

Ji, J., Peng, Y., Wang, B., Li, X., Zhang, Q., 2020. A novel SNPR process for advanced nitrogen and phosphorus removal from mainstream wastewater based on anammox, endogenous partial-denitrification and denitrifying dephosphatation. Water Res. 170, 115363. https://doi.org/10.1016/j.watres.2019.115363

Jimenez, J., Miller, M., Bott, C., Murthy, S., De Clippeleir, H., Wett, B., 2015. High-rate activated sludge system for carbon management - Evaluation of crucial process mechanisms and design parameters. Water Res. 87, 476–482. https://doi.org/10.1016/j.watres.2015.07.032

Kamali, M., Suhas, D.P., Costa, M.E., Capela, I., Aminabhavi, T.M., 2019. Sustainability considerations in membrane-based technologies for industrial effluents treatment. Chem. Eng. J. 368, 474–494. https://doi.org/10.1016/j.cej.2019.02.075

Kamble, S.J., Chakravarthy, Y., Singh, A., Chubilleau, C., Starkl, M., Bawa, I., 2017. A soil biotechnology system for wastewater treatment: technical, hygiene, environmental LCA and economic aspects. Environ. Sci. Pollut. Res. 24, 13315–13334. https://doi.org/10.1007/s11356-017-8819-6

Kampschreur, M.J., van der Star, W.R.L., Wielders, H.A., Mulder, J.W., Jetten, M.S.M., van Loosdrecht, M.C.M., 2008. Dynamics of nitric oxide and nitrous oxide emission during full-scale reject water treatment. Water Res. 42, 812–826. https://doi.org/10.1016/j.watres.2007.08.022

Kelessidis, A., Stasinakis, A.S., 2012. Comparative study of the methods used for treatment and final disposal of sewage sludge in European countries. Waste

Manag. 32, 1186–1195. https://doi.org/10.1016/j.wasman.2012.01.012

Kujawa-Roeleveld, K., Elmitwalli, T., Zeeman, G., 2006. Enhanced primary treatment of concentrated black water and kitchen residues within DESAR concept using two types of anaerobic digesters. Water Sci. Technol. 53, 159–168. https://doi.org/10.2166/wst.2006.265

Lackner, S., Gilbert, E.M., Vlaeminck, S.E., Joss, A., Horn, H., van Loosdrecht, M.C.M., 2014. Full-scale partial nitritation/anammox experiences - An application survey. Water Res. 55, 292–303. https://doi.org/10.1016/j.watres.2014.02.032

Larsen, H.F., Hauschild, M., Wenzel, H., Almemark, M., 2007. Homogeneous LCA methodology agreed by NEPTUNE and INNOWATECH. Denmark 1–34.

Li, Y., Zhang, S., Zhang, W., Xiong, W., Ye, Q., Hou, X., Wang, C., Wang, P., 2019. Life cycle assessment of advanced wastewater treatment processes: Involving 126 pharmaceuticals and personal care products in life cycle inventory. J. Environ. Manage. 238, 442–450. https://doi.org/10.1016/j.jenvman.2019.01.118

Lijó, L., Malamis, S., González-García, S., Moreira, M.T., Fatone, F., Katsou, E., 2017. Decentralised schemes for integrated management of wastewater and domestic organic waste: the case of a small community. J. Environ. Manage. 203, 732–740. https://doi.org/10.1016/j.jenvman.2016.11.053

Lishan, X., Tao, L., Yin, W., Zhilong, Y., Jiangfu, L., 2018. Comparative life cycle assessment of sludge management: A case study of Xiamen, China. J. Clean. Prod. 192, 354–363. https://doi.org/10.1016/j.jclepro.2018.04.171

Lorenzo-Toja, Y., Vázquez-Rowe, I., Amores, M.J., Termes-Rifé, M., Marín-Navarro, D., Moreira, M.T., Feijoo, G., 2016. Benchmarking wastewater treatment plants under an eco-efficiency perspective. Sci. Total Environ. 566–567, 468–479. https://doi.org/10.1016/j.scitotenv.2016.05.110

Malovanyy, A., Trela, J., Plaza, E., 2015. Mainstream wastewater treatment in integrated fixed film activated sludge (IFAS) reactor by partial nitritation/anammox process. Bioresour. Technol. 198, 478–487. https://doi.org/10.1016/j.biortech.2015.08.123

Mills, N., Pearce, P., Farrow, J., Thorpe, R.B., Kirkby, N.F., 2014. Environmental & economic life cycle assessment of current & future sewage sludge to energy technologies. Waste Manag. 34, 185–195. https://doi.org/10.1016/j.wasman.2013.08.024

Morales, N., Val del Río, Á., Vázquez-Padín, J.R., Méndez, R., Mosquera-Corral, A., Campos, J.L., 2015. Integration of the Anammox process to the rejection water and main stream lines of WWTPs. Chemosphere 140, 99–105. https://doi.org/10.1016/j.chemosphere.2015.03.058

Morera, S., Santana, M.V.E., Comas, J., Rigola, M., Corominas, L., 2020. Evaluation of different practices to estimate construction inventories for life cycle assessment of small to medium wastewater treatment plants. J. Clean. Prod. 245, 118768. https://doi.org/10.1016/j.jclepro.2019.118768

Nair, S., George, B., Malano, H.M., Arora, M., Nawarathna, B., 2014. Water-energy-greenhouse gas nexus of urban water systems: Review of concepts, state-of-art and methods. Resour. Conserv. Recycl. 89, 1–10. https://doi.org/10.1016/j.resconrec.2014.05.007

Nguyen, T.K.L., Ngo, H.H., Guo, W.S., Chang, S.W., Nguyen, D.D., Nghiem, L.D., Nguyen, T. V., 2020. A critical review on life cycle assessment and plant-wide models towards emission control strategies for greenhouse gas from wastewater treatment plants. J. Environ. Manage. 264, 110440. https://doi.org/10.1016/j.jenvman.2020.110440

Niero, M., Pizzol, M., Bruun, H.G., Thomsen, M., 2014. Comparative life cycle assessment of wastewater treatment in Denmark including sensitivity and uncertainty analysis. J. Clean. Prod. 68, 25–35. https://doi.org/10.1016/j.jclepro.2013.12.051

Opher, T., Friedler, E., 2016. Comparative LCA of decentralized wastewater treatment alternatives for non-potable urban reuse. J. Environ. Manage. 182, 464–476. https://doi.org/10.1016/j.jenvman.2016.07.080

Otterpohl, R., Raak, M., Rolfs, T., 1994. A mathematical model for the efficiency of the primary clarification, in: Proceedings of IAWQ 17th Biennial International Conference.

Pedrouso, A., Aiartza, I., Morales, N., Vázquez-Padín, J.R., Rogalla, F., Campos, J.L., Mosquera-Corral, A., Val del Rio, A., 2018. Pilot-scale ELAN® process applied to treat primary settled urban wastewater at low temperature via partial nitritation-anammox processes. Sep. Purif. Technol. 200, 94–101. https://doi.org/10.1016/j.seppur.2018.02.017

Rajasekhar, B., Venkateshwaran, U., Durairaj, N., Divyapriya, G., Nambi, I.M., Joseph, A., 2020. Comprehensive treatment of urban wastewaters using electrochemical advanced oxidation process. J. Environ. Manage. 266, 110469.

https://doi.org/10.1016/j.jenvman.2020.110469

REE, 2018. Spanish Power Systems 2017.

Resende, J.D., Nolasco, M.A., Pacca, S.A., 2019. Life cycle assessment and costing of wastewater treatment systems coupled to constructed wetlands. Resour. Conserv. Recycl. 148, 170–177. https://doi.org/10.1016/j.resconrec.2019.04.034

Rodriguez-Caballero, A., Aymerich, I., Poch, M., Pijuan, M., 2014. Evaluation of process conditions triggering emissions of green-house gases from a biological wastewater treatment system. Sci. Total Environ. 493, 384–391. https://doi.org/10.1016/j.scitotenv.2014.06.015

Rodriguez-Garcia, G., Molinos-Senante, M., Hospido, A., Hernández-Sancho, F., Moreira, M.T., Feijoo, G., 2011. Environmental and economic profile of six typologies of wastewater treatment plants. Water Res. 45, 5997–6010. https://doi.org/10.1016/j.watres.2011.08.053

Ruiken, C.J., Breuer, G., Klaversma, E., Santiago, T., van Loosdrecht, M.C.M., 2013. Sieving wastewater - Cellulose recovery, economic and energy evaluation. Water Res. 47, 43–48. https://doi.org/10.1016/j.watres.2012.08.023

Saagi, R., Flores-Alsina, X., Kroll, S., Gernaey, K. V., Jeppsson, U., 2017. A model library for simulation and benchmarking of integrated urban wastewater systems. Environ. Model. Softw. 93, 282–295. https://doi.org/10.1016/j.envsoft.2017.03.026

Sancho, I., Lopez-Palau, S., Arespacochaga, N., Cortina, J.L., 2019. New concepts on carbon redirection in wastewater treatment plants: A review. Sci. Total Environ.

647, 1373–1384. https://doi.org/10.1016/j.scitotenv.2018.08.070

Schaubroeck, T., De Clippeleir, H., Weissenbacher, N., Dewulf, J., Boeckx, P., Vlaeminck, S.E., Wett, B., 2015. Environmental sustainability of an energy self-sufficient sewage treatment plant: Improvements through DEMON and co-digestion. Water Res. 74, 166–179. https://doi.org/10.1016/j.watres.2015.02.013

Shiu, H.Y., Lee, M., Chiueh, P. Te, 2017. Water reclamation and sludge recycling scenarios for sustainable resource management in a wastewater treatment plant in Kinmen islands, Taiwan. J. Clean. Prod. 152, 369–378. https://doi.org/10.1016/j.jclepro.2017.03.110

Singh, A., Kamble, S.J., Sawant, M., Chakravarthy, Y., Kazmi, A., Aymerich, E., Starkl, M., Ghangrekar, M., Philip, L., 2018. Technical, hygiene, economic, and life cycle assessment of full-scale moving bed biofilm reactors for wastewater treatment in India. Environ. Sci. Pollut. Res. 25, 2552–2569. https://doi.org/10.1007/s11356-017-0605-y

Singh, A.D., Upadhyay, A., Shrivastava, S., Vivekanand, V., 2020. Bioresource Technology Life-cycle assessment of sewage sludge-based large-scale biogas plant. Bioresour. Technol. 309, 123373. https://doi.org/10.1016/j.biortech.2020.123373

Smitshuijzen, J., Pérez, J., Duin, O., Loosdrecht, M.C.M. van, 2016. A simple model to describe the performance of highly-loaded aerobic COD removal reactors. Biochem. Eng. J. https://doi.org/10.1016/j.bej.2016.04.004

Solon, K., Flores-Alsina, X., Kazadi Mbamba, C., Ikumi, D., Volcke, E.I.P., Vaneeckhaute, C., Ekama, G., Vanrolleghem, P.A., Batstone, D.J., Gernaey, K. V., Jeppsson, U., 2017. Plant-wide modelling of phosphorus transformations in wastewater treatment systems: Impacts of control and operational strategies. Water Res. 113, 97–110. https://doi.org/10.1016/j.watres.2017.02.007

Tabesh, M., Feizee Masooleh, M., Roghani, B., Motevallian, S.S., 2019. Life-Cycle Assessment (LCA) of Wastewater Treatment Plants: A Case Study of Tehran, Iran. Int. J. Civ. Eng. 17, 1155–1169. https://doi.org/10.1007/s40999-018-0375-z

Takács, I., Patry, G.G., Nolasco, D., 1991. A dynamic model of the clarification-thickening process. Water Res. 25, 1263–1271. https://doi.org/10.1016/0043-1354(91)90066-Y

Tao, L., Tan, E.C.D., Mccormick, R., Zhang, M., Aden, A., He, X., Zigler, B.T., 2014. Techno-economic analysis and life-cycle assessment of cellulosic isobutanol and comparison with cellulosic ethanol and n-butanol 30–48. https://doi.org/10.1002/bbb

Termes-Rifé, M., Molinos-Senante, M., Hernández-Sancho, F., Sala-Garrido, R., 2013. Life cycle costing: A tool to manage the urban water cycle. J. Water Supply Res. Technol. - AQUA 62, 468–476. https://doi.org/10.2166/aqua.2013.110

Tomei, M.C., Bertanza, G., Canato, M., Heimersson, S., Laera, G., Svanström, M., 2016. Techno-economic and environmental assessment of upgrading alternatives for sludge stabilization in municipal wastewater treatment plants. J. Clean. Prod. 112, 3106–3115. https://doi.org/10.1016/j.jclepro.2015.10.017

Vangsgaard, A.K., Mutlu, A.G., Gernaey, K. V, Smets, B.F., Sin, G., 2013. Calibration and validation of a model describing complete autotrophic nitrogen removal in a granular SBR system. J. Chem. Technol. Biotechnol. 88, 2007–2015. https://doi.org/https://doi.org/10.1002/jctb.4060

Vázquez-Padín, J., Mosquera-Corral, A., Campos, J.L., Méndez, R., Revsbech, N.P., 2010. Microbial community distribution and activity dynamics of granular biomass in a CANON reactor. Water Res. 44, 4359–4370. https://doi.org/10.1016/j.watres.2010.05.041

Wan, J., Gu, J., Zhao, Q., Liu, Y., 2016. OPEN COD capture : a feasible option towards energy self-sufficient domestic wastewater treatment. Nat. Publ. Gr. 1–9. https://doi.org/10.1038/srep25054

Wang, X., Ratnaweera, H., Holm, J.A., Olsbu, V., 2017. Statistical monitoring and dynamic simulation of a wastewater treatment plant: A combined approach to achieve model predictive control. J. Environ. Manage. 193, 1–7. https://doi.org/10.1016/j.jenvman.2017.01.079

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Wu, J., Zhang, J., Jia, W., Xie, H., Gu, R.R., Li, C., Gao, B., 2009. Impact of COD/N ratio on nitrous oxide emission from microcosm wetlands and their performance in removing nitrogen from wastewater. Bioresour. Technol. 100, 2910–2917. https://doi.org/10.1016/j.biortech.2009.01.056

Xu, X., Ma, B., Lu, W., Feng, D., Wei, Y., Ge, C., Peng, Y., 2020. Effective nitrogen removal in a granule-based partial-denitrification/anammox reactor treating low C/N sewage. Bioresour. Technol. 297, 122467. https://doi.org/10.1016/j.biortech.2019.122467

Yoshida, H., ten Hoeve, M., Christensen, T.H., Bruun, S., Jensen, L.S., Scheutz, C., 2018. Life cycle assessment of sewage sludge management options including long-term impacts after land application. J. Clean. Prod. 174, 538–547. https://doi.org/10.1016/j.jclepro.2017.10.175

Zang, Y., Li, Y., Wang, C., Zhang, W., Xiong, W., 2015. Towards more accurate life cycle assessment of biological wastewater treatment plants: A review. J. Clean. Prod. 107, 676–692. https://doi.org/10.1016/j.jclepro.2015.05.060

Figure Captions

Figure 1. System boundaries of the different wastewater schemes considered.

Figure 2. Environmental impacts for each scenario considered in Avedøre WWTP (FU: 1 kWh of energy produced). (a) Scenario 0 (conventional case) (b) Scenario 1 (ERBF) (c) Scenario 2 (HRAS). Acronyms: CC: climate change; OD: ozone depletion; TA: terrestrial acidification; EP: eutrophication potential; HT: human toxicity; PMF: particulate matter formation; TET: terrestrial ecotoxicity: FET: freshwater ecotoxicity; MET: marine ecotoxicity; WD: water depletion; FD: fossil depletion

Figure 3. Environmental impacts for each scenario considered in Valladolid WWTP (FU: 1 kWh of energy produced). (a) Scenario 0 (conventional case) (b) Scenario 1 (ERBF) (c) Scenario 2 (HRAS). Acronyms: CC: climate change; OD: ozone depletion; TA: terrestrial acidification; EP: eutrophication potential; HT: human toxicity; PMF: particulate matter formation; TET: terrestrial ecotoxicity: FET: freshwater ecotoxicity; MET: marine ecotoxicity; WD: water depletion; FD: fossil depletion

Figure 4. Comparison between the different scenarios considered for Avedøre plant (a) CC category (b) EP category. Acronyms: CC: climate change; EP: eutrophication potential

Figure 5. Comparison between the different scenarios considered for the Valladolid plant (a) climate change (CC) (b) eutrophication potential (EP)

Figure 6. Comparison between the different plant sizes (FU: 1 kWh of energy produced). a) Climate change (CC) category; b) eutrophication potential (EP) category. Acronyms: ○ large plant Δ medium plant.

Figure 7. Comparison between Spanish and Danish electricity country mix production. Acronyms: CC: climate change; OD: ozone depletion; PMF: particulate matter formation; TA: terrestrial acidification; HT: human toxicity; FD: fossil depletion

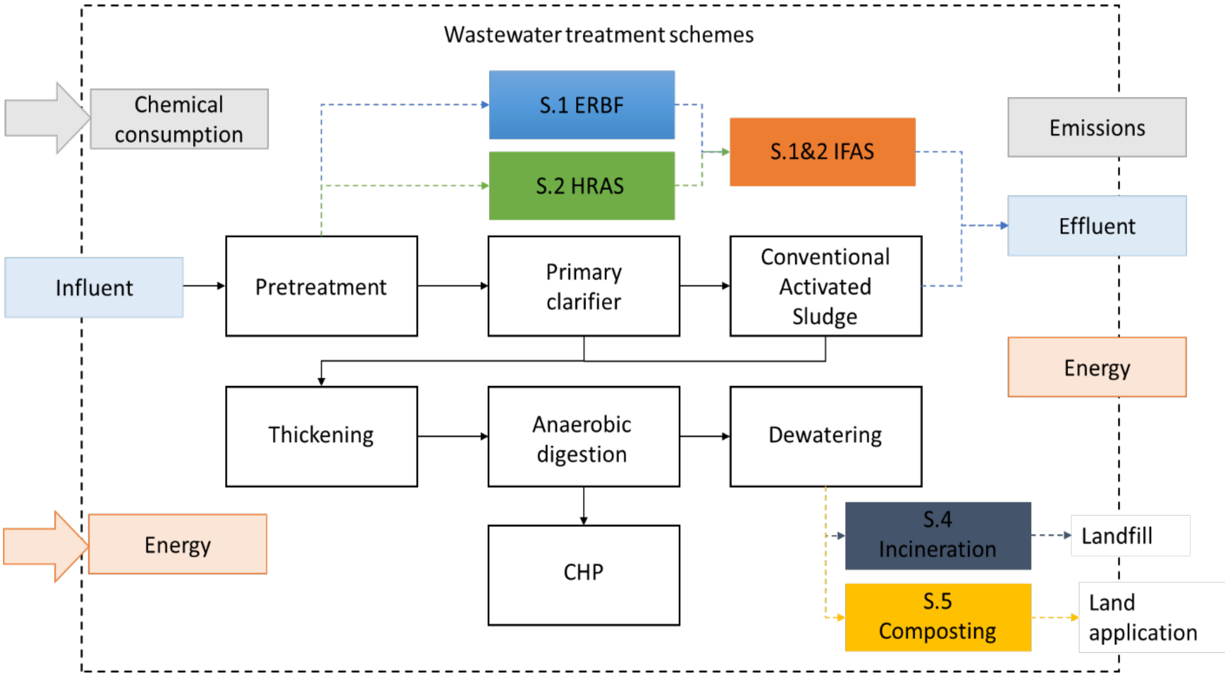
Wastewater treatment schemes
Chemical consumption
S.1 ERBF
S.2 HRAS
S.1&2 IFAS
Emissions
Effluent
Influent
Pretreatment
Primary clarifier
Conventional Activated Sludge
Energy
Thickening
Anaerobic digestion
Dewatering
Energy
CHP
S.4 Incineration
Landfill
S.5 Composting
Land application

Figure 1

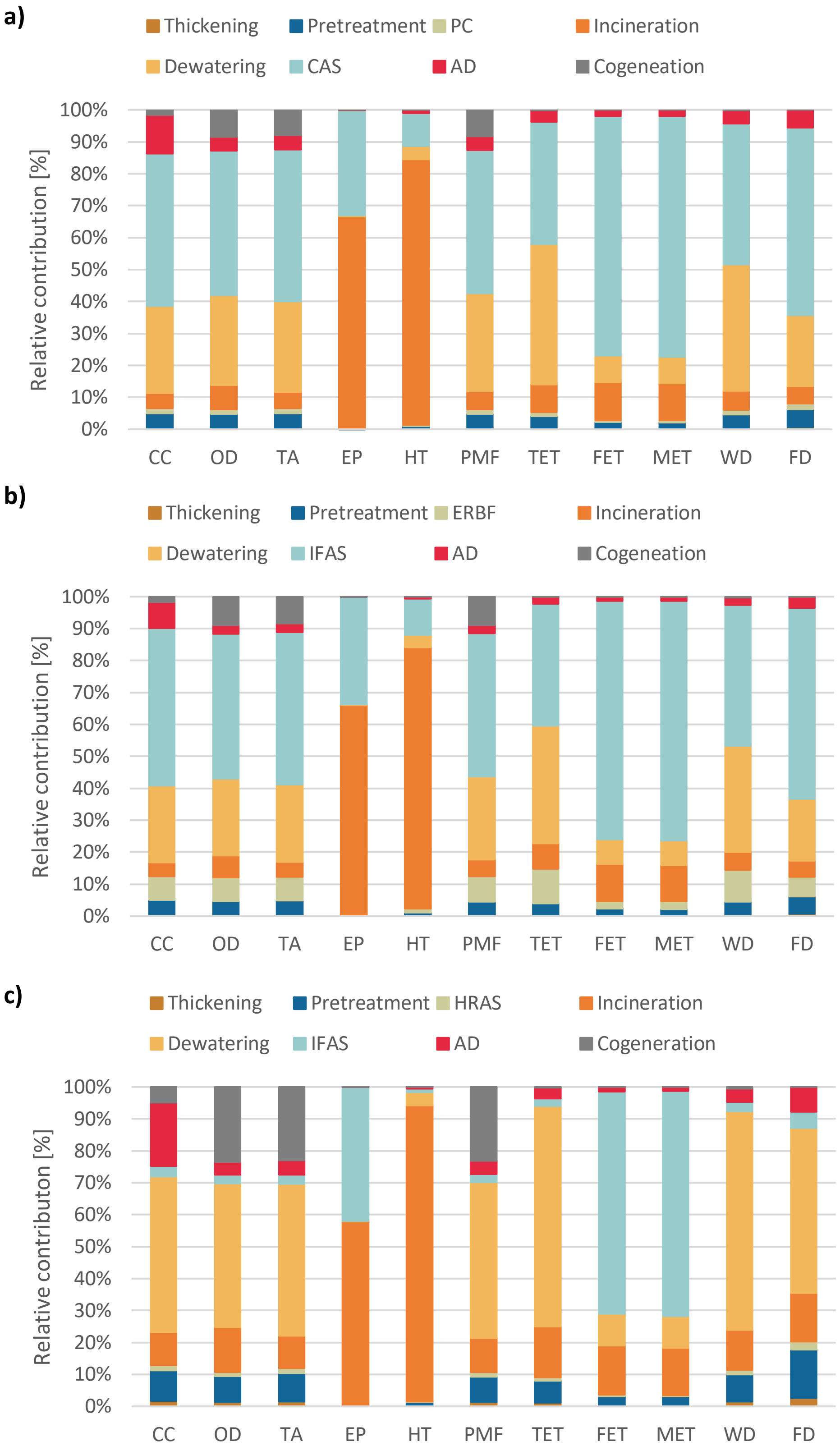
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Figure 2


a)
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FD

Figure 3

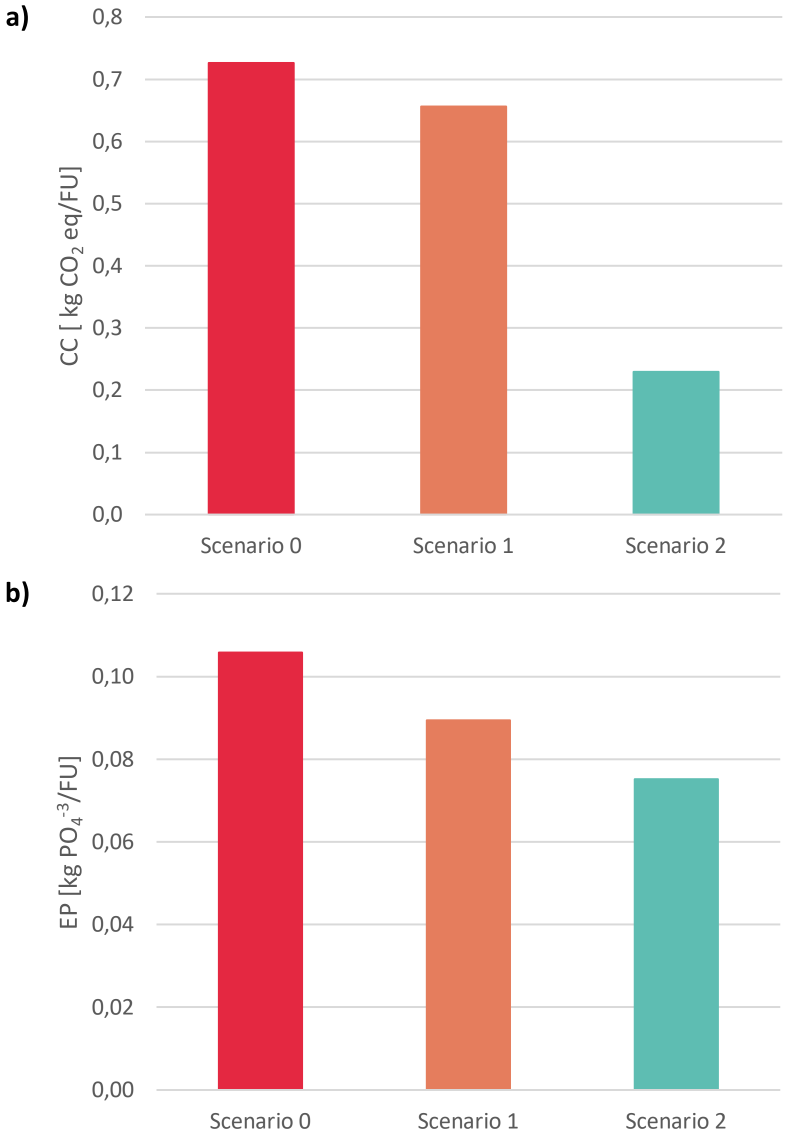
a)
0,8
0,7
0,6
0,5
0,4
0,3
0,2
0,1
0,0
CC [kg CO_2 eq/FU]
Scenario 0
Scenario 1
Scenario 2
b)
0,12
0,10
0,08
0,06
0,04
0,02
0,00
EP [kg PO_4^{-3}/FU]
Scenario 0
Scenario 1
Scenario 2

Figure 4

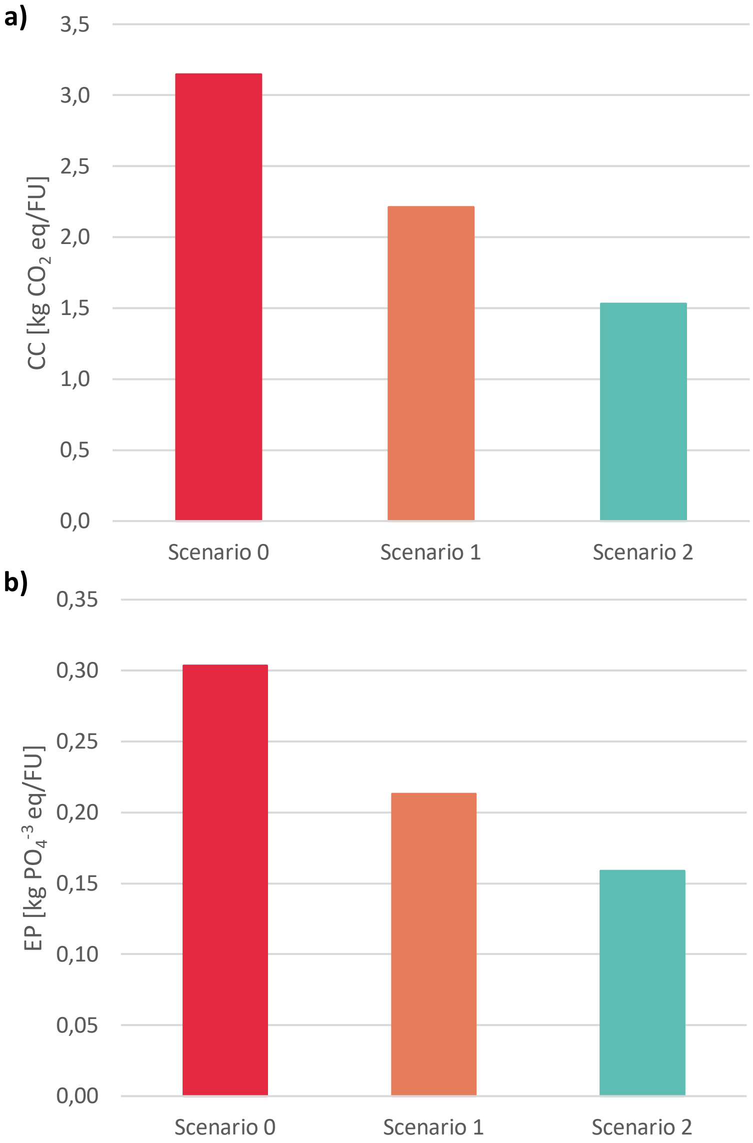
a)
CC [kg CO2 eq/FU]
3,5
3,0
2,5
2,0
1,5
1,0
0,5
0,0
Scenario 0
Scenario 1
Scenario 2
b)
EP [kg PO4-3 eq/FU]
0,35
0,30
0,25
0,20
0,15
0,10
0,05
0,00
Scenario 0
Scenario 1
Scenario 2

Figure 5

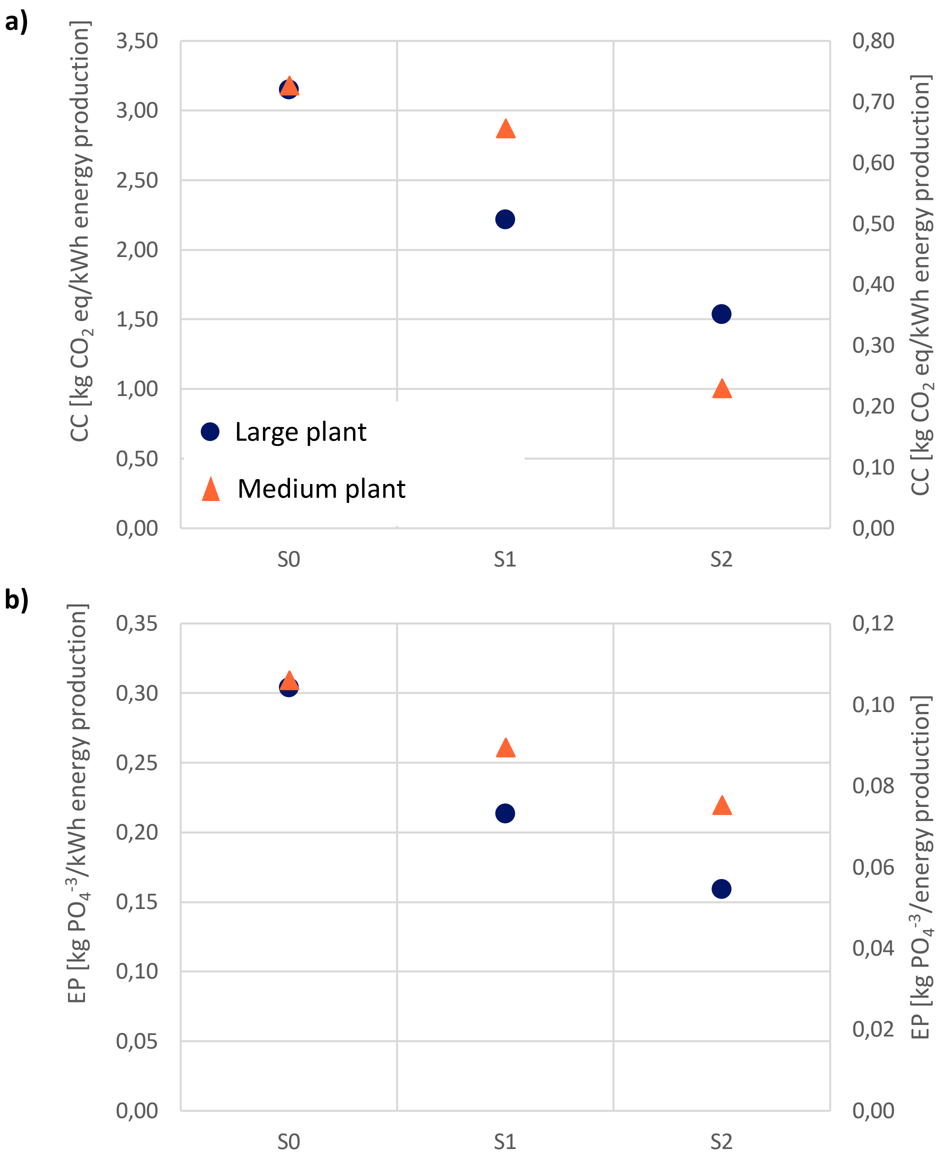
a)
CC [kg CO2 eq/kWh energy production]
3,50
3,00
2,50
2,00
1,50
1,00
0,50
0,00
0,80
0,70
0,60
0,50
0,40
0,30
0,20
0,10
0,00
CC [kg CO2 eq/kWh energy production]
S0
S1
S2
● Large plant
▲ Medium plant
b)
EP [kg PO4-3/kWh energy production]
0,35
0,30
0,25
0,20
0,15
0,10
0,05
0,00
0,12
0,10
0,08
0,06
0,04
0,02
0,00
EP [kg PO4-3/energy production]
S0
S1
S2

Figure 6

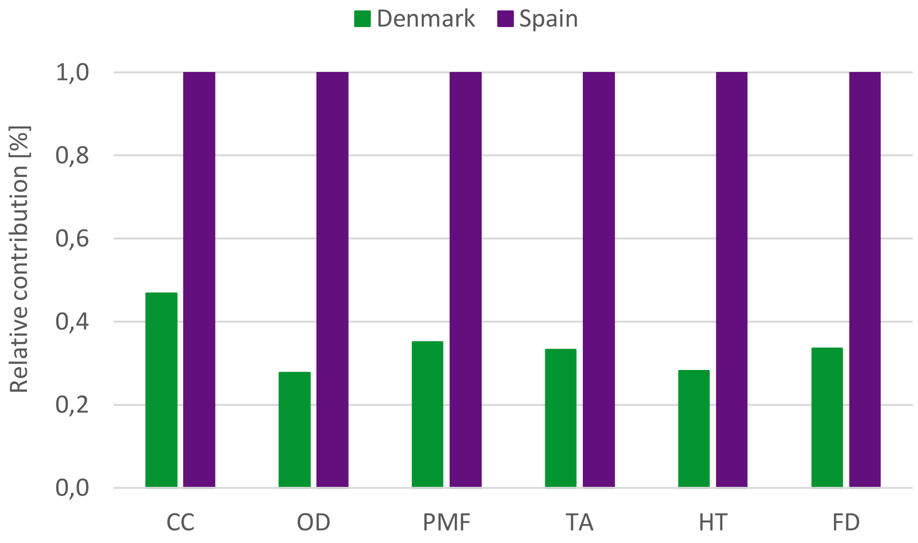
Denmark
Spain
Relative contribution [%]
1,0
0,8
0,6
0,4
0,2
0,0
CC
OD
PMF
TA
HT
FD

Figure 7

Table 1. Modelling approach used for conventional and emerging technologies

Technology Name	**Mechanism**	**Modeling approach reference**
Primary clarifier (PC)	Gravitational settling	(Gernaey et al., 2014; Otterpohl et al., 1994)
Enhanced rotating belt filter (ERBF)	Coagulation, flocculation, sieving and cake filtration	(Behera et al., 2018; Boiocchi et al., 2019)
High rate activated sludge (HRAS)	Bio-sorption	(Smitshuijzen et al., 2016)
Modified Ludzack-Ettinger (MLE)	COD oxidation, nitrification, and pre-denitrification	(Guo and Vanrolleghem, 2014; Henze et al., 2000)
Secondary clarifier (SC)	Gravitational settling	(Takács et al., 1991)
Integrated fixed-film activated sludge (IFAS)	Partial nitrification/anammox, COD oxidation, conventional denitrification	(Behera et al., 2019; Vangsgaard et al., 2013)
Thickener and Dewatering	Gravitational settling	(Gernaey et al., 2014)
Anaerobic digester (AD)	Hydrolysis, acidogenesis, acetogenesis, methanogenesis	(Batstone et al., 2002)

Table 2a. Main inputs to the different wastewater schemes considered in the Averdøre plant (FU: 1 kWh energy produced)

	Scenario 0	**Scenario 1**	**Scenario 2**
Inputs from the technosphere			
Materials and fuel			
Influent			
COD (kg)	2.93	2.93	2.93
TN (kg)	0.30	0.30	0.30
TP (kg)	$3.28{\cdot}10^{-3}$	$3.28{\cdot}10^{-3}$	$3.28{\cdot}10^{-3}$
Electricity consumption			
Pretreatment (kWh)	0.09	0.08	0.06
PC (kWh)	0.03	–	–
ERBF (kWh)	–	0.05	–
HRAS (kWh)	–	–	0.01
Activated sludge (kWh)	0.94	–	–
IFAS (KWh)	–	0.88	0.02
Thickening (kWh)	0.01	$9.82{\cdot}10^{-3}$	$7.24{\cdot}10^{-3}$
AD (kWh)	0.09	0.05	0.03
Dewatering (kWh)	0.17	0.14	0.10
Incineration (kWh)	0.07	0.06	0.05
Chemical consumption			
Primary treatment			
Polyelectrolyte (kg)	–	0.02	–
Dewatering			
Polyelectrolyte (kg)	0.09	0.07	0.05
Transport			
Polyelectrolyte (kg·km)	2.18	2.34	1.35
Ashes (kg·km)	17.09	14.30	10.55
Amount of ashes (kg)	0.68	0.50	0.42

Table 2b. Main outputs to the different wastewater schemes considered in the Averdøre plant (FU: 1 kWh energy produced)

	Scenario 0	**Scenario 1**	**Scenario 2**
Outputs to the environment			
Emissions to air			
AD			
CH_4 (kg)	$1.63 \cdot 10^{-2}$	$1.02 \cdot 10^{-2}$	$1.02 \cdot 10^{-2}$
CO_2 (kg)	$3.28 \cdot 10^{-2}$	$2.05 \cdot 10^{-2}$	$2.05 \cdot 10^{-2}$
H_2S (kg)	$5.64 \cdot 10^{-4}$	$3.52 \cdot 10^{-4}$	$3.52 \cdot 10^{-4}$
Emissions to water			
Effluent			
COD (kg)	0.27	0.24	0.15
TN (kg)	0.04	0.03	0.03
TP (g)	3.69	3.69	3.69
Pb (mg)	3.28	2.74	2.02
Cd (mg)	0.23	0.20	0.14
Cu (mg)	9.60	8.03	5.92
Cr (mg)	9.12	7.64	5.63
Hg (mg)	0.47	0.39	0.29
As (mg)	3.75	3.13	2.31
Ni (mg)	32.77	27.41	30.21
Zn (mg)	139.03	116.32	85.79

Table 3a. Main inputs to the different wastewater schemes considered in the Valladolid plant (FU: 1 kWh energy produced)

	Scenario 0	**Scenario 1**	**Scenario 2**
Inputs from the technosphere			
Materials and fuel			
Influent			
COD (kg)	7.31	4.64	2.46
TN (kg)	0.64	0.41	0.22
TP (kg)	7.68	4.88	2.58
Electricity consumption			
Pre-treatment (kWh)	0.31	0.20	0.11
PC (kWh)	0.09	–	–
ERBF (kWh)	–	0.13	–
HRAS (kWh)	–	–	0.01
Activated sludge (kWh)	3.52	–	–
IFAS (KWh)	–	2.03	1.02
Thickening (kWh)	0.04	0.02	0.01
AD (kWh)	0.29	0.09	0.04
Dewatering (kWh)	0.55	0.03	0.18
Composting (kWh)	0.20	0.13	0.07
Chemical consumption			
Primary treatment			
Polyelectrolyte (kg)	–	0.05	–
Dewatering			
Polyelectrolyte (kg)	0.85	0.54	0.28
Transport			
Polyelectrolyte (kg·km)	21.16	14.72	7.11
Sludge (kg·km)	72.27	50.77	24.26
Spreading (kg)	2.89	2.03	0.97

Table 3b. Main outputs to the different wastewater schemes considered in the Valladolid plant (FU: 1 kWh energy produced)

	Scenario 0	**Scenario 1**	**Scenario 2**
Outputs to the environment			
Emissions to air			
AD			
CH_4 (kg)	$1.02 \cdot 10^{-2}$	$9.96 \cdot 10^{-3}$	$1.51 \cdot 10^{-2}$
CO_2 (kg)	$8.06 \cdot 10^{-2}$	$1.95 \cdot 10^{-2}$	$4.20 \cdot 10^{-2}$
H_2S (kg)	$3.52 \cdot 10^{-4}$	$3.44 \cdot 10^{-4}$	$5.21 \cdot 10^{-4}$
Composting			
CH_4 (kg)	$1.84 \cdot 10^{-2}$	$1.29 \cdot 10^{-2}$	$6.17 \cdot 10^{-3}$
CO_2 (kg)	1.69	2.75	1.31
N_2O (kg)	$9.82 \cdot 10^{-2}$	$1.85 \cdot 10^{-4}$	$9.73 \cdot 10^{-5}$
NH_3 (kg)	$5.06 \cdot 10^{-2}$	$5.61 \cdot 10^{-2}$	$2.95 \cdot 10^{-2}$
Land application			
N_2O (kg)	$1.47 \cdot 10^{-3}$	$1.09 \cdot 10^{-3}$	$5.74 \cdot 10^{-4}$
NH_3 (kg)	$1.21 \cdot 10^{-3}$	$9.00 \cdot 10^{-4}$	$4.73 \cdot 10^{-4}$
Emissions to water			
Effluent			
COD (kg)	0.74	0.45	0.26
TN (kg)	0.19	0.03	0.02
TP (g)	39.70	39.70	39.70
Pb (mg)	11.69	7.42	3.92
Cd (mg)	1.30	0.83	0.44
Cu (mg)	127.26	80.82	42.73
Cr (mg)	6.65	4.22	2.23
Hg (mg)	8.36	5.31	2.80
As (mg)	63.98	40.63	21.50
Ni (mg)	39.29	24.96	13.20
Zn (mg)	381.98	242.58	128.29
Land application			
NO_3^- (kg)	$9.82 \cdot 10^{-2}$	$1.85 \cdot 10^{-4}$	$5.74 \cdot 10^{-4}$
PO_4^{-3} (kg)	$5.06 \cdot 10^{-2}$	$5.61 \cdot 10^{-2}$	$4.73 \cdot 10^{-4}$
Emissions to soil			
TP (g)	1.65	1.05	0.55
Pb (mg)	122.14	77.57	41.01
Cd (mg)	1.27	0.80	0.43
Cu (mg)	382.96	242.20	128.61
Cr (mg)	61.36	38.97	20.61
Hg (mg)	1.20	0.76	0.40
As (mg)	25.58	16.24	8.59
Ni (mg)	46.17	29.32	15.50
Zn (mg)	826.70	524.99	277.63

Table 4. Economic results for the different plant sizes and scenarios considered (FU: 1 kWh of energy produced)

	Electricity consumption ($€·kWh^{-1}$)	**Chemical consumption ($€·kg^{-1}$)**	**Sludge management ($€·kg^{-1}$)**	**Total ($€·kWh^{-1}$)**
Avedøre				
Case 0	0.25	0.02	0.23	0.51
Case 1	0.22	0.02	0.05	0.30
Case 2	0.04	0.01	0.04	0.10
Valladolid				
Case 0	0.59	0.19	0.26	1.04
Case 1	0.31	0.13	0.18	0.62
Case 2	0.17	0.06	0.09	0.32