Unravelling the economics behind mainstream anaerobic membrane bioreactor application under different plant layouts

Sergi Vinardell[a,*], Joan Dosta[a], Joan Mata-Alvarez[a], Sergi Astals[a]

[a] Department of Chemical Engineering and Analytical Chemistry, University of Barcelona, 08028, Barcelona, Spain

*Corresponding author (e-mail: svinardell@ub.edu)

ABSTRACT

This research evaluated the economic feasibility of anaerobic membrane bioreactor (AnMBR) as a mainstream technology for municipal sewage treatment. To this end, different wastewater treatment plant (WWTP) layouts were considered, including primary settler, AnMBR, degassing membrane, partial nitritation-Anammox, phosphorus precipitation and sidestream anaerobic digestion. The net treatment cost of an AnMBR-WWTP decreased from 0.42 to 0.35 € m^{-3} as the sewage COD concentration increased from 100 to 1100 mg COD L^{-1} due to revenue from electricity production. However, the net treatment cost increased above 0.51 € m^{-3} when nutrient removal technologies were included. The AnMBR and partial nitritation-Anammox were the costliest processes representing a 57.6 and 30.3% of the treatment cost, respectively. Energy self-sufficiency was achieved for high-strength municipal sewage treatment (1000 mg COD L^{-1}) and a COD:SO_4^{2-}-S ratio above 40. Overall, the results showed that mainstream AnMBR has potential to be an economically competitive option for full-scale implementation.

KEYWORDS: Anaerobic digestion; Anaerobic membrane bioreactor (AnMBR); Water resource recovery factory (WRRF); Plant-wide assessment; Techno-economic analysis

1. Introduction

Most wastewater treatment plants (WWTPs) were designed and constructed decades ago when sewage was considered a source of pollution rather than a source of resources (Sheik et al., 2014). These WWTPs, based on the conventional activated sludge (CAS) process, do not make an efficient use of the energy, water and nutrients contained in municipal sewage. The development and implementation of novel technologies able to maximise resource recovery while obtaining high-quality effluents is crucial to transform WWTPs into water resource recovery factories (WRRF) (Guest et al., 2009).

Anaerobic membrane bioreactor (AnMBR) is a promising technology for mainstream municipal sewage treatment (Vinardell et al., 2020a). In contrast to the CAS process, AnMBR converts sewage organic matter into renewable biogas energy with no oxygen requirements and low sludge production. Additionally, the membrane system allows producing high-quality effluents and providing an excellent decoupling of the hydraulic retention time (HRT) from the solids retention time (SRT) (Stuckey, 2012).

The application of mainstream AnMBR technology represents an opportunity for WWTPs to become energy neutral, reduce treatment costs and produce high-quality effluents. Several publications have demonstrated that AnMBRs can achieve energy self-sufficiency at net treatment costs between 0.1 and 0.4 € m^{-3} (Batstone et al., 2015; Cogert et al., 2019; Evans et al., 2019; Lim et al., 2019; Pretel et al., 2015b; Shoener et al., 2016; Smith et al., 2014). However, some of these studies limited their analysis to the AnMBR unit, omitting the implications and impact that AnMBR implementation has on sewage primary treatment, AnMBR post-treatment and sludge management. The incorporation of all these factors in the AnMBR-WWTP economic evaluation is paramount to obtain a realistic picture since the feasibility of AnMBR for mainstream

WWTP application goes beyond its capacity to achieve high COD removal efficiencies and produce biogas.

The sensitivity of the AnMBR process to the sewage chemical oxygen demand (COD) and sulphate concentrations is critical for the AnMBR profitability (Batstone et al., 2015; Song et al., 2018). On the one hand, the higher the sewage COD concentration, the higher the amount of COD available for methane production (Shin and Bae, 2018). On the other hand, the presence of sulphate makes sulphate reducing bacteria (SRB) compete with anaerobic microorganisms for easily biodegradable substrate while reducing sulphate to sulphide (Serrano et al., 2019). Additionally, the presence of sulphide partially inhibits methanogenic archaea activity and makes necessary the use of equipment and instrumentation resistant to corrosion (Madden et al., 2014). Some previous publications have considered the impact of sewage COD concentration (Batstone et al., 2015; Cogert et al., 2019; Smith et al., 2014) and sewage sulphate concentration (Pretel et al., 2015a) on AnMBR-WWTP feasibility. However, little attention has been given to the relative and combined impact of sewage COD and sulphate concentrations on AnMBR-WWTP costs.

The presence of dissolved methane and nutrients (i.e. N, P) in the permeate (AnMBR effluent) is a major bottleneck for AnMBR application (Vinardell et al., 2020a). Dissolved methane can account for 50% of the total methane produced at psychrophilic conditions (ca. 15 ºC) and its recovery is important to maximise energy production and minimise greenhouse gas emissions (Sanchis-Perucho et al., 2020; Smith et al., 2013). Furthermore, the mobilisation of nutrients in the AnMBR as a result of organic matter degradation makes necessary the implementation of post-treatments able to recover or remove nitrogen and phosphorus to fulfil the discharge requirements and reduce the environmental impact on aquatic systems (Robles et al., 2020). Accordingly, the inclusion of all treatment units (i.e. dissolved methane recovery, nutrients recovery/removal, and sludge management) in the economic evaluation is necessary to obtain a reliable estimation of the AnMBR-WWTP costs.

The selection of suitable plant layouts able to solve the aforementioned challenges is critical to support AnMBR full-scale implementation. Several plant layouts have been proposed for AnMBR-WWTPs, including (i) downstream processes for the recovery/removal of nutrients (Ab Hamid et al., 2020; Batstone et al., 2015; Cogert et al., 2019; Harclerode et al., 2020; Lim et al., 2019), (ii) downstream processes for the recovery/removal of dissolved methane (Cogert et al., 2019; Harclerode et al., 2020; Lim et al., 2019; Pretel et al., 2015a), and (iii) sludge treatment processes for sludge hygienisation and conditioning (Ab Hamid et al., 2020; Batstone et al., 2015; Harclerode et al., 2020; Pretel et al., 2015a). Although some of these studies considered similar operational conditions and sewage characteristics, they differed in the AnMBR-WWTP layout. These different selection criteria show that AnMBR-WWTP layout does not have yet a common baseline framework. Consequently, the plant layout selection is sometimes made intuitively and subjectively omitting important factors such as the plant cost or the plant energy consumption. However, energy and economic aspects should be particularly considered for the selection of the AnMBR-WWTP layout.

The goal of this study is to evaluate the economic feasibility of WWTPs based on mainstream AnMBR technology under different plant layouts. To this end, this research evaluated the impact of sewage COD and sulphate concentrations on the AnMBR-WWTP energy and economic balances, as well as their impact on the plant layout selection. The AnMBR-WWTP layout comprises a combination of primary settler, AnMBR, dissolved methane recovery, nutrients removal and sidestream anaerobic digestion. The ultimate goal is to provide a comprehensive understanding of the implications that different factors and their combination have on the AnMBR-WWTP economic feasibility.

2. Methodology

2.1. AnMBR-WWTP scenarios definition

Figure 1 illustrates the three scenarios considered for the energy-economic evaluation, while Table 1 shows the treatments included in each scenario. The different scenarios were evaluated for an AnMBR-WWTP treating 100,000 m^3 d^{-1} of municipal sewage with different COD and sulphate concentrations. The lifetime of the AnMBR-WWTP was 20 years. The three scenarios conceived in this publication are summarised below.

Scenario 1 represents the implementation of an AnMBR and a downstream dissolved methane recovery unit (see Figure 1A). Degassing membrane was selected for dissolved methane recovery since this technology achieves relatively high recovery efficiencies (ca. 70%) at a relatively low energy input (0.01 kWh m^{-3}) (Cookney et al., 2016; Lim et al., 2019). In Scenario 1, primary settler, sidestream anaerobic digestion (AD) and nutrients treatment were not included.

Scenario 2 was an extension of Scenario 1 and included three different plant layouts integrating primary settler and/or sidestream AD (see Figure 1B and Table 1). The primary settler controls the amount of COD fed to the AnMBR, which affects (i) the amount of methane produced in the AnMBR unit and (ii) the energy consumption for membrane fouling control. The sidestream AD maximises biogas energy production and further stabilises the WWTP sludge. The three plant layouts were: (i) Scenario 2A with a primary settler and sidestream AD, (ii) Scenario 2B with a primary settler and without sidestream AD, and (iii) Scenario 2C with sidestream AD and without primary settler.

Scenario 3 was an extension of the most favourable alternative in Scenario 2 (i.e. Scenario 2C) and included downstream treatments for nitrogen and phosphorus removal (see Figure 1C). Phosphorus precipitation with ferric chloride was used for mainstream phosphorus removal (Figure 1C) (Harclerode et al., 2020; Lim et al., 2019), while partial nitritation-Anammox (PN-Anammox) was selected for nitrogen removal since this is a suitable treatment for sewage with a low COD:N ratio (Batstone et al., 2015; Cogert et al., 2019). Specifically, a MBR PN-Anammox system was considered for nitrogen removal due to its capacity to achieve an effective retention of the slow-growing Anammox bacteria in the system at ambient temperature (Dai et al., 2015; Kwak et al., 2020). The PN-Anammox was placed before phosphorus precipitation unit: (i) to prevent phosphorus limitation in PN-Anammox and (ii) to allow a better control of phosphorus removal, which is important to meet the increasingly stringent regulations concerning phosphorus discharge.

2.2. Sewage composition and variability

Municipal sewage COD concentrations ranging from 100 to 1200 mg COD L^{-1} were considered. This interval is representative for municipal sewage and comprises typical concentrations for low-, medium-, and high-strength sewage. Sewage COD consisted of biodegradable soluble COD ($COD_{S,B}$), inert soluble COD ($COD_{S,I}$), biodegradable particulate COD ($COD_{X,B}$) and inert particulate COD ($COD_{X,I}$) representing individual fractions of 0.36, 0.04, 0.40 and 0.20, respectively (Henze et al., 2008). Total nitrogen and phosphorus concentrations were obtained through a lineal COD-dependent function adapted from data provided by Henze et al. (2008). Specifically, the ratios for nitrogen and phosphorus were 12.5 mg COD mg^{-1} N and 51 mg COD mg^{-1} P, respectively. Therefore, the sewage N and P concentrations increased as the sewage COD concentration increased. In Section 3.3.2, where the influence of PN-Anammox energy consumption was evaluated, the sewage nitrogen concentration ranged from 10 to 100 mg N L^{-1} for a fixed sewage COD concentration of 700 mg COD L^{-1}.

A COD:SO_4^{2-}-S ratio of 57 was considered in the scenarios where the influence of sulphate was not evaluated since this is a typical ratio for sewage with a low sulphate content (Ferrer et al., 2015). However, in Section 3.2 and 3.3.1, where the influence of sulphate concentration was evaluated, the COD:SO_4^{2-}-S ratio ranged from 2 to 100.

2.3. System design and costs

The AnMBR-WWTP was designed (i) using data reported from lab-, pilot- and full-scale applications and (ii) well-stablished model equations (ASCE et al., 1996; Cogert et al., 2019; Metcalf & Eddy, 2014; Pretel et al., 2015a; Qasim, 1999; Smith et al., 2014). This section summarises the main design and cost considerations for the different technologies considered in this study including AnMBR, PN-Anammox, phosphorus precipitation, primary settler, sludge treatment processes (i.e. sidestream AD, sludge thickener, and centrifuge), dissolved methane recovery and methane valorisation. Detailed information about the equations and parameters used for cost and energy calculations can be found in the electronic supplementary material.

2.3.1 AnMBR

The AnMBR was designed as a two-stage process operated at ambient temperature (20 ºC). The AnMBR system consisted of two tanks: (i) the anaerobic digester and (ii) a membrane tank equipped with submerged ultrafiltration membrane modules. A two-stage system was chosen since the maintenance procedures are simpler than for one stage-systems (Shin and Bae, 2018). The membrane area was calculated considering a net flux of 10 L m^{-2} h^{-1} (LMH) (Giménez et al., 2011; Smith et al., 2014). The membrane replacement cost was calculated considering a membrane lifetime of 10 years (Harclerode et al., 2020; Smith et al., 2014). Gas sparging was used to control long-term membrane fouling assuming a specific gas demand (SGD) of 0.23 Nm^3 m^{-2} h^{-1} (Giménez et al., 2011; Smith et al., 2014). An SRT of 60 days and an HRT of 1 day were considered (Lim et al., 2019; Vinardell et al., 2020b). The recirculation flow rate from the bioreactor to the membrane tank is an important parameter for two-stage AnMBRs since it allows controlling the mixed liquor suspended solids (MLSS) concentration in the membrane tank and reducing membrane fouling (Aslam et al., 2019; Ferrer et al., 2015). In this study, the recirculation flow rate was calculated considering a MLSS concentration in the membrane tank of 18 g L^{-1} (Shin and Bae, 2018). This approach allowed evaluating the influence of sewage strength on energy consumption for fouling control. The MLSS concentration and sludge production were calculated through steady-state equations (see electronic supplementary material).

AnMBR energy consumption accounted for pumping requirements (i.e. influent pump, recirculation pump and permeate/backwash pump), stirring requirements and gas sparging. The energy required to operate centrifugal pumps and gas blowers were calculated through theoretical equations (see electronic supplementary material). The other operating costs (i.e. membrane replacement, chemical reagents for membrane cleaning, labour and equipment maintenance) and capital costs (i.e. civil engineering, mechanical/electrical and equipment) were adapted from Vinardell et al (2020b).

2.3.2 PN-Anammox

The PN-Anammox process was designed for a nitrogen loading rate (NLR) of 0.3 kg N m^{-3} d^{-1}. This is a typical NLR for mainstream PN-Anammox applications (Batstone et al., 2015; Dai et al., 2015). It was assumed that 90% of sewage nitrogen remained in the AnMBR effluent mainly as ammonium ion (Bair et al., 2015), and that nitrogen removal efficiencies of 81% were achieved in the PN-Anammox process (Dai et al., 2015; Schaubroeck et al., 2015). PN-Anammox sludge production was calculated through a steady-state equation considering the growth rates of ammonia oxidising bacteria (AOB) and anammox bacteria (see electronic supplementary material). Theoretically, the PN-Anammox process can significantly reduce the energy requirements for nitrogen removal when compared with the conventional nitrification-denitrification process (Morales et al., 2015). However, the selective inhibition of nitrite oxidising bacteria (NOB), the retention of anammox bacteria at low temperatures, and the presence of residual organic matter in the anaerobic effluent increase PN-Anammox energy requirements (Cruz et al., 2019; Schaubroeck et al., 2015). An energy consumption of 5 kWh per kg of N removed was used for the economic evaluation of the PN-Anammox process (Schaubroeck et al., 2015). However, future technological advances could improve the PN-Anammox process and, subsequently, reduce its energy requirements. In Section 3.3.2, the impact of reducing PN-Anammox energy consumption on AnMBR-WWTP energy balance was evaluated through a sensitivity analysis.

2.3.3 Phosphorus precipitation

The chemical precipitation of phosphorus included a settler, a sludge thickener and a centrifuge. It was considered that 89% of sewage phosphorus remained in the AnMBR effluent mainly as phosphate (Bair et al., 2015) and that phosphorus removal efficiencies in the precipitation unit were 90% (Taboada-Santos et al., 2020). Ferric chloride ($FeCl_3$) was used for phosphorus precipitation considering a cost of 220 € t^{-1} (Taboada-Santos et al., 2020). Sludge thickening and sludge dewatering were designed considering that the sludge production differed for the different sewage P concentrations (see Section 2.3.5 for further details on thickener and centrifuge design).

2.3.4 Primary settler

The primary settler efficiency determines the amount of COD fed to the AnMBR, which has a direct impact on the AnMBR biogas production as well as on the MLSS concentration in the bioreactor. A 40% of the sewage COD was separated in the primary sludge in those scenarios with primary settler (Metcalf & Eddy, 2014). The primary sludge was composed of 6% of TSS and a COD biodegradable fraction of 0.66 (Andreoli et al., 2007). The COD biodegradable fraction was calculated considering that the amount of soluble COD contained in primary sludge is negligible.

2.3.5 Sludge treatment processes

Sludge thickening, sidestream AD and sludge dewatering processes were designed considering that the sludge production differed for the different sewage COD concentrations and for the different scenarios and layouts. It was assumed that the combined thickened sludge contained a 5% of TSS and that no solids were washed out in the thickener. The AD was designed to treat a VS loading rate of 1.6 kg VS m^{-3} d^{-1} at mesophilic conditions (Andreoli et al., 2007) with a VS removal ranging between 17-59% depending on the sludge biodegradability of each scenario. The biodegradability of the combined sludge (including primary and secondary) was calculated considering (i) the biodegradable particulate fraction of the sewage that is separated in the primary settler, and (ii) the amount of sludge that is biologically produced in the AnMBR and PN-Anammox processes. The energy consumption for sludge treatment accounted for sludge thickening, digester mixing and sludge dewatering. Polyelectrolyte was dosed at 6 kg t^{-1} TSS with a cost of 2.35 € kg^{-1} (Pretel et al., 2015b). It was considered that the biosolids (dewatered sludge) after AD were stable and thus suitable to be used as fertiliser with a cost of 4.8 € t^{-1} TSS (Ferrer et al., 2015). However, a higher disposal cost was considered in Scenario 2B and Scenario 3. Scenario 2B does not have sidestream AD for mixed sludge and, therefore, the mixed sludge needs to be incinerated or disposed in a landfill. In Scenario 3, the $Fe_3(PO_4)_2$ sludge produced from phosphorus removal is disposed in a landfill since it is not suitable for land application.

2.3.6 Methane recovery and valorisation

The methane produced in the AnMBR and the sidestream AD was calculated considering that: (i) all biodegradable COD was biologically degraded in the anaerobic digesters, (ii) a fraction of COD was used for biomass growth (0.076 mg TSS mg^{-1} COD), (iii) SRB consumed 2.01 mg of biodegradable COD per mg of SO_4^{2-}-S (Giménez et al., 2011), and (iv) a fraction of methane remained dissolved in the AnMBR effluent (17.8 mg L^{-1}). The dissolved methane concentration was calculated with Henry's law at ambient temperature (20 ºC). The methane produced was combusted in a combined heat and power (CHP) unit with an electricity yield of 33% (Appels et al., 2011). The methane calorific power was 38,800 kJ m^{-3} (0 ºC and 1 atm) (Metcalf & Eddy, 2014).

Dissolved methane was partially recovered through degassing membrane, which was designed for a membrane flux of $3 \cdot 10^{-8}$ kmol CH_4 m^{-2} s^{-1} (Rongwong et al., 2017; Sethunga et al., 2019), and a lifetime of 7 years (Cookney et al., 2016). A methane recovery efficiency of 70% was considered based on pilot-scale reported efficiencies (Lim et al., 2019; Seco et al., 2018). It was considered that degassing membrane was operated at an energy input of 0.01 kWh m^{-3} (Evans et al., 2019; Lim et al., 2019). The methane recovered by the degassing membrane was accounted for energy production.

2.4 Economic evaluation

Capital expenditure (CAPEX) and operating expenditure (OPEX) for the different scenarios were calculated. Electricity revenue from the energy produced through biogas cogeneration was also included in the economic evaluation. CAPEX was annualised over the project lifetime with Eq. (1). The net treatment cost, including CAPEX, OPEX and electricity revenue (ER), was calculated using Eq. (2). Finally, the net treatment cost was referred to the volume of sewage treated to facilitate the comparison with other studies and treatment configurations.

$$\text{Annualised CAPEX } (€\ \mathrm{y}^{-1}) = \frac{\mathrm{i} \cdot (1+\mathrm{i})^{\mathrm{t}}}{(1+\mathrm{i})^{\mathrm{t}} - 1} \cdot \text{CAPEX} \qquad \text{Eq. (1)}$$

$$\text{Net treatment cost } (€\ \mathrm{y}^{-1}) = \frac{\mathrm{i} \cdot (1+\mathrm{i})^{\mathrm{t}}}{(1+\mathrm{i})^{\mathrm{t}} - 1} \cdot \text{CAPEX} + \text{OPEX} - \text{ER} \qquad \text{Eq. (2)}$$

Where CAPEX is the initial investment (€), OPEX is the operating cost (€ y^{-1}), ER is the electricity revenue (€ y^{-1}), i is the discount rate, and t is the plant lifetime (20 years). The discount rate was established at 5% in the scenarios where the influence of the discount rate was not analysed. In Section 3.4, four discount rates (i.e. 5, 10, 15 and 20 %) were used to evaluate the influence of the discount rate on treatment costs.

3. RESULTS AND DISCUSSION

3.1 Energy and economic evaluation of an AnMBR-WWTP with dissolved methane recovery

Figure 2 shows the energy balance and the net treatment cost of Scenario 1 for COD concentrations between 100 and 1200 mg COD L^{-1}. The energy balance (green line in Figure 2) shows that (i) the AnMBR process with dissolved methane recovery achieves energy self-sufficiency for COD concentrations above 550 mg COD L^{-1} and (ii) the maximum net energy production (0.32 kWh m^{-3}) is reached when sewage has a COD concentration of 1100 mg COD L^{-1}. The energy recovery increases from 0.05 to 1.01 kWh m^{-3} as the COD concentration increases from 100 to 1200 mg COD L^{-1} (yellow line in Figure 2). The energy consumption suddenly increases at 1100 mg COD L^{-1} (blue line in Figure 2), due to the higher recirculation flow rate from the bioreactor to the membrane tank. A higher recirculation flow rate is needed to keep the MLSS concentration in the membrane tank constant at 18 g L^{-1} since the MLSS concentration in the bioreactor increases as a result of the higher sewage COD concentration. Controlling the MLSS concentration in the membrane tank (i) reduces the gas sparging energy requirements, (ii) minimises the use of intensive and complex membrane cleaning protocols and (iii) lowers the membrane replacement frequency.

Scenario 1 features a net treatment cost (brown line in Figure 2) between 0.42 and 0.35 € m^{-3} for COD concentrations between 100 and 1200 mg COD L^{-1}. These results agree with Smith et al. (2014), who reported similar net treatment costs (ca. 0.37-0.41 $ m^{-3}, i=5%, 40 years plant lifetime) for an AnMBR-WWTP without degassing membrane and treating 18,950 m^3 d^{-1} of sewage. The production of energy is a distinctive feature of AnMBRs compared with other aerobic technologies such as CAS and MBRs with energy costs between 0.04-0.08 and 0.06-0.11 € m^{-3}, respectively (Iglesias et al., 2017).

For the WWTP under study, methane production allows achieving net energy production for COD concentrations above 550 mg COD L^{-1}. However, the methane dissolved in the permeate represents 8-100% of the methane produced under these operational conditionals. Therefore, its recovery is required to increase energy production and reduce uncontrolled methane emissions (Cookney et al., 2016; Lim et al., 2019). In the present study, the benefit-cost ratio for the degassing membrane was estimated at 1.1. The economic prospect of degassing membrane could be improved if environmental incomes were considered because degassing membrane reduces mainstream greenhouse gas emissions from 0.308 to 0.113 kg CO_2-eq m^{-3} (Sanchis-Perucho et al., 2020). The benefit-cost ratio of degassing membrane increases to 2.0 when the current European Union carbon price (27 € t^{-1} CO_2-eq) is considered (EMBER, 2020). Although degassing membrane technology still needs to be tested at full-scale, the recovery of methane from AnMBR effluents appears crucial to reduce environmental impacts of mainstream anaerobic digestion (Smith et al., 2014).

3.2 Economic evaluation of an AnMBR-WWTP integrating primary settler and sidestream AD

Figure 3A shows the net treatment cost of Scenario 2 for COD concentrations between 100 and 1200 mg COD L^{-1}. The scenario without primary settler (i.e. Scenario 2C) is the most competitive for COD concentrations below 1100 mg COD L^{-1}. Specifically, the net treatment cost of this scenario decreases from 0.42 to 0.35 € m^{-3} as the sewage COD concentration increases from 100 to 1100 mg COD L^{-1}. The net treatment cost of Scenario 2C is nearly the same than Scenario 1, which does not include neither primary settler nor sidestream AD (see Figure 2). These results show that the biogas produced from the sludge wasted from the AnMBR in Scenario 2C could offset the costs related to the construction and operation of the sidestream AD. Besides environmental incomes (out of the scope of this publication), further electricity revenue for the sidestream AD could be achieved by implementing co-digestion strategies (Macintosh et al., 2019).

Scenario 2A, which includes both primary settler and sidestream AD, features a net treatment cost 0.01 € m^{-3} higher than Scenario 2C (scenario without primary settler) for

COD concentrations below 1100 mg COD L^{-1}. However, Scenario 2A displays the cheapest cost at sewage COD concentrations above 1100 mg COD L^{-1} (Figure 3A). These results highlight that an AnMBR-WWTP treating high-strength sewage (> 1100 mg COD L^{-1}) should integrate primary settler to reduce chemicals and energy consumption associated with fouling control. In Scenario 2A, the high methane yield of primary sludge (ca. 400 mL CH_4 g^{-1} VS) is recovered in the sidestream AD instead of the AnMBR. The importance of the sidestream AD when the AnMBR-WWTP includes a primary settler is shown in Scenario 2B (scenario without sidestream AD), which presents the worse cost among the three configurations considered in Scenario 2 (Figure 3A). Scenario 2B fails to recover energy and to stabilise primary sludge and, therefore, it is considered unsuitable from both economic and environmental points of view.

The sewage COD:SO_4^{2-}-S ratio has been highlighted as a critical factor for AnMBR profitability and plant layout selection (Pretel et al., 2015a; Vinardell et al., 2020a). COD:SO_4^{2-}-S ratios between 43 and 60 have been reported in previous AnMBR publications (Harclerode et al., 2020; Pretel et al., 2015b; Smith et al., 2014). However, a lower COD:SO_4^{2-}-S ratio is possible when treating sulphate-rich sewage. Accordingly, evaluating the impact of COD:SO_4^{2-}-S ratio on net treatment cost is important to understand the influence of this variable on AnMBR-WWTP profitability.

Figure 3B shows the net treatment cost of Scenario 2 for COD:SO_4^{2-}-S ratios between 2 and 100 at a constant COD concentration of 700 mg COD L^{-1}. The impact of low COD:SO_4^{2-}-S ratios on methane yield is particularly relevant in Scenario 2C (scenario without primary settler) where the net treatment cost suddenly increases from 0.38 to 0.44 € m^{-3} as the COD:SO_4^{2-}-S ratio decreases from 15 to 2, respectively. The integration of a primary settler (Scenario 2A) should be considered when the

$COD:SO_4^{2-}$-S ratio is below 8 since primary settler allows valorising a fraction of the sewage COD to methane in the sidestream AD (Figure 3B). However, Scenario 2A decreases the $COD:SO_4^{2-}$-S ratio in the AnMBR influent and, subsequently, most of the COD is used to convert sulphate into sulphide rather than for methane production. Accordingly, mainstream AnMBR application does not appear suitable to treat sewage with a $COD:SO_4^{2-}$ ratio below 15 (700 mg COD L^{-1}) regardless of the presence of a primary settler in the WWTP layout.

Overall, Scenario 2C (including sidestream AD and without primary settler) presents the most favourable energy and economic prospects for AnMBR-WWTP treating sewage with COD concentrations between 100 and 1100 mg COD L^{-1} and $COD:SO_4^{2-}$-S ratios above 8. Scenario 2C appears also the most appropriate configuration when the nutrient-rich effluent can be directly used for agricultural irrigation. However, in most applications, the AnMBR effluent has to be discharged into the environment and, therefore, a certain level of post-treatment would be required to comply with N and P discharge limits. In Section 3.3 and 3.4, an energy-economic evaluation of an AnMBR-WWTP is conducted including nitrogen and phosphorus nutrient removal technologies.

3.3 Energy evaluation of an AnMBR-WWTP with dissolved methane recovery, sidestream AD and nutrients removal

3.3.1 Impact of COD concentration and $COD:SO_4^{2-}$-S ratio

Figure 4A shows the energy balance of Scenario 3 for a low-, medium-, and high-strength sewage (400, 700 and 1000 mg COD L^{-1}, respectively) and for $COD:SO_4^{2-}$-S ratios between 2 and 100. For $COD:SO_4^{2-}$-S ratios higher than 5, the treatment of medium- and high-strength sewage features a more favourable energy balance than the treatment of low-strength sewage. The energy balance for $COD:SO_4^{2-}$-S ratios below 5 is unfavourable regardless of the sewage strength with values ranging between -0.47 and -0.87 kWh m^{-3} (Figure 4). These results reinforce the idea that mainstream AnMBR is not suitable to treat sulphate-rich sewage. Furthermore, sulphide production has a direct negative impact on biological performance, membrane permeability and infrastructure durability (not included in this analysis), further worsening the economic and energetic prospects of AnMBR-WWTPs treating sulphate-rich sewage (Harclerode et al., 2020; Song et al., 2018).

Figure 4A also shows that the effect of COD:SO_4^{2-}-S ratio on the energy balance has a tipping point at 10, 15 and 20 for sewage concentrations of 400, 700 and 1000 mg COD L^{-1}, respectively. This tipping point represents the COD:SO_4^{2-}-S ratio where the impact of sulphate reduction on the energy balance and process profitability lowers its influence. Accordingly, the application of mainstream AnMBR for the treatment of low-, medium- and high-strength sewage should be considered for COD:SO_4^{2-}-S ratios above 10, 15 and 20, respectively. Operating below this threshold implies not only a poor energy balance, but also a high sensitivity of the energy balance towards sewage COD:SO_4^{2-}-S ratio fluctuations.

The energy balance plotted for the low-, medium-, and high-strength sewage is asymptotic to -0.28, -0.10 and +0.04 kWh m^{-3}, respectively. These results show that an AnMBR-WWTP including dissolved methane recovery, AnMBR, PN-Anammox, phosphorus precipitation and sidestream AD has potential to reduce the net energy requirements in comparison with aerobic-based WWTP configurations such as CAS process (0.3-0.6 kWh m^{-3}) (Fernández-Arévalo et al., 2017), aerobic MBR (0.4-0.6 kWh m^{-3}) (Xiao et al., 2019), or high-rate activated sludge (0.39 kWh m^{-3}) (Taboada-Santos et al., 2020). However, although the AnMBR allows reducing the WWTP energy consumption, energy neutrality is only achieved when treating the high-strength sewage with COD:SO_4^{2-}-S ratios above 40. For low- and medium-strength sewage, the energy consumption should be further reduced to achieve an energy self-sufficient AnMBR-WWTP. For low- and medium-strength sewage, the PN-Anammox process consumes 0.11 and 0.19 kWh m^{-3} accounting for 17 and 25% of the total energy consumption, respectively. Future PN-Anammox improvements could reduce its treatment cost and, consequently, overcome the constraints associated with nitrogen removal towards achieving an energy self-sufficient AnMBR-WWTP.

3.3.2 Impact of PN-Anammox energy consumption

Figure 4B illustrates the energy balance of Scenario 3 for nitrogen concentrations between 10 and 100 mg N L^{-1} and considering three PN-Anammox energy consumptions (i.e. 1, 3, and 5 kWh kg^{-1} N). This interval was selected because energy consumptions between 1 and 5 kWh kg^{-1} N have been previously reported for mainstream PN-Anammox (Batstone et al., 2015; Schaubroeck et al., 2015). Figure 4B shows that the AnMBR-WWTP energy consumption could be reduced up to 0.27 kWh m^{-3} if the PN-Anammox energy consumption is reduced from 5 to 1 kWh kg^{-1} N (100 mg N L^{-1}). A PN-Anammox energy consumption of 1 kWh kg^{-1} N would make the AnMBR-WWTP energy self-sufficient regardless of the sewage nitrogen concentration. However, further technological advances are still required to operate mainstream PN-Anammox process at 1 kWh kg^{-1} N (Schaubroeck et al., 2015). On the other hand, energy self-sufficiency is only achieved for nitrogen concentrations below 35 and 25 mg N L^{-1} when PN-Anammox has an energy consumption of 3 and 5 kWh kg^{-1} N, respectively. These results indicate that reducing the energy consumption of mainstream PN-Anammox is crucial to achieve a self-sufficient AnMBR-WWTP for medium- or low-strength sewage treatment. However, the economic prospects of the MBR PN-Anammox process also requires considering the energy consumption for membrane fouling control. To overcome the limitations associated with mainstream PN-Anammox, alternative physical methods are being researched. Specifically, ion exchange processes appear to be a promising alternative to valorise nitrogen from AnMBR effluents with relatively low costs (Cruz et al., 2019; Huang et al., 2020; Lim et al., 2019). Overall, the development of efficient technologies for nitrogen removal or recovery is important to make AnMBR technology competitive for municipal sewage treatment.

3.4 Economic evaluation of AnMBR-WWTP with dissolved methane recovery, sidestream AD and nutrients removal

The AnMBR-WWTP under assessment includes innovative technologies primarily tested at lab- and pilot-scale but still lacking demonstration at full-scale. The risk associated with the implementation of these novel technologies can be quantified through the discount rate. The discount rate is a financial parameter that allows including the value of money over time and the uncertainty related to future cash flows (Papapetrou et al., 2017). Since the use of mainstream AnMBR application is more risky than aerobic technologies, it is important to evaluate the influence that the discount rate has on AnMBR-WWTP treatment costs.

Figure 5 shows the net treatment cost of Scenario 3 for sewage COD concentrations between 100 and 1200 mg COD L^{-1} and considering discount rates of 5, 10, 15 and 20%. The net treatment cost does not experience important variations as the sewage COD concentration increases despite the tipping point observed at 1100 mg COD L^{-1} (see Section 3.1). Importantly, these results show that higher COD concentrations do not lead to lower net treatment costs in Scenario 3 because the increased methane production does not offset the higher CAPEX and OPEX associated with nutrients removal and membrane fouling control. For the lowest discount rate (5%), the net treatment cost ranges between 0.51 and 0.56 € m^{-3}, which is competitive compared with the 0.30-0.60 € m^{-3} treatment cost reported for CAS and MBR technologies (including CAPEX and OPEX) (Verstraete et al., 2009). However, a discount rate of 5% is applied for well-stablished technologies and, therefore, it is little realistic for an AnMBR-WWTP. A discount rate of 10% increases the net treatment cost to 0.68-0.74 € m^{-3}, whereas a discount rate above 15% leads to net treatment cost above 0.90 € m^{-3}. These results show that the net treatment cost of mainstream AnMBR application can be competitive compared with aerobic treatments. However, the risk associated with implementing a range of innovative technologies can significantly compromise the AnMBR-WWTP economic feasibility. Therefore, research at demonstration-scale is crucial to reduce the risk and uncertainty associated with these novel technologies and support the transition from WWTPs to WRRFs.

Figure 6 shows the costs distribution of AnMBR-WWTPs with AnMBR, dissolved methane recovery, PN-Anammox, phosphorus precipitation and sidestream AD for sewage COD concentration of 700 mg COD L^{-1}. The AnMBR is the most expensive unit, representing 57.6% of the treatment cost. The PN-Anammox process also represents an important fraction of the treatment cost (30.3%). Since AnMBR and PN-Anammox account for 87.9% of the treatment cost, future research efforts should aim to reduce costs associated with these technologies. Sludge treatment cost only represents 3.4% of the treatment cost since mainstream AnMBR application notably reduces sludge production compared with aerobic technologies. The revenue coming from methane production allows reducing 10.8% the treatment cost. Besides electricity, further revenue from the reutilisation of the high-quality effluent free of suspended solids and nutrients could be obtained in future AnMBR-WWTPs.

The OPEX of the AnMBR-WWTP only represents between 30.5 and 36.5% of the treatment cost since net energy consumption and sludge production are reduced in anaerobic systems (see electronic supplementary material). Accordingly, reducing CAPEX is crucial to reduce treatment costs. In this regard, retrofitting existing aerobic-based WWTPs to AnMBR-WWTPs stands as a promising alternative to implement mainstream AnMBR technology with reduced CAPEX. Indeed, the net treatment cost of AnMBR-WWTP could be reduced up to 0.12 € m^{-3} if only OPEX and the revenue from energy production were considered (see electronic supplementary material). Finally, it is worth mentioning that in a retrofitted AnMBR-WWTP the existing sidestream anaerobic digester would be oversized due to the lower amount of sludge produced in the AnMBR process. However, this represents an opportunity to implement co-digestion in the AnMBR-WWTP as a strategy to further increase biogas energy production and reduce the net treatment cost.

4. Conclusions

The economic feasibility of mainstream AnMBR-WWTP was investigated. The net treatment cost of a WWTP including AnMBR, degassing membrane and sidestream AD was between 0.42 and 0.35 € m^{-3} for a sewage COD concentration between 100 and 1200 mg COD L^{-1}. The incorporation of nutrient removal technologies increased the net treatment cost above 0.51 € m^{-3} despite a net energy production of 0.04 kWh m^{-3} was achieved for high-strength municipal sewage treatment (1000 mg COD L^{-1}). The results showed that reducing the treatment cost of AnMBR and PN-Anammox is important to make AnMBR-WWTP competitive for municipal sewage treatment.

Declaration of competing interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper. The authors also declare that this manuscript reflects only the authors' view and that the Executive Agency for SME/EU Commission are not responsible for any use that may be made of the information it contains.

Supplementary information

E-supplementary data of this work can be found in online version of the paper.

Acknowledgments

This work was supported by the European Union LIFE programme (LIFE Green Sewer project, LIFE17 ENV/ES/000341). Sergi Vinardell is grateful to the Generalitat de Catalunya for his predoctoral FI grant (2019FI_B 00394). Sergi Astals is grateful for his Ramon y Cajal fellowship (RYC-2017-22372).

References

1.Ab Hamid, N.H., Smart, S., Wang, D.K., Koh, K.W.J., Ng, K.J.C., Ye, L., 2020. Economic, energy and carbon footprint assessment of integrated forward osmosis membrane bioreactor (FOMBR) process in urban wastewater treatment. Environ. Sci. Water Res. Technol. 6, 153–165.

2.Andreoli, C.V., von Sperling, M., Fernandes, F., 2007. Biological Wastewater Treatment, Sludge treatment and disposal. IWA Publishing, London.

3.Appels, L., Lauwers, J., Degrve, J., Helsen, L., Lievens, B., Willems, K., Van Impe, J., Dewil, R., 2011. Anaerobic digestion in global bio-energy production: Potential and research challenges. Renew. Sustain. Energy Rev. 15, 4295–4301.

4.ASCE, AWWA, EPA, 1996. Management of water treatment plant residuals : technology transfer handbook. ASCE and AWWA. New York.

5.Aslam, A., Khan, S.J., Shahzad, H.M.A., 2019. Impact of sludge recirculation ratios on the performance of anaerobic membrane bioreactor for wastewater treatment. Bioresour. Technol. 288, 121473.

6.Bair, R.A., Ozcan, O.O., Calabria, J.L., Dick, G.H., Yeh, D.H., 2015. Feasibility of anaerobic membrane bioreactors (AnMBR) for onsite sanitation and resource recovery (nutrients, energy and water) in urban slums. Water Sci. Technol. 72, 1543–1551.

7.Batstone, D.J., Hülsen, T., Mehta, C.M., Keller, J., 2015. Platforms for energy and nutrient recovery from domestic wastewater: A review. Chemosphere 140, 2–11.

8.Cogert, K.I., Ziels, R.M., Winkler, M.K.H., 2019. Reducing Cost and Environmental Impact of Wastewater Treatment with Denitrifying Methanotrophs, Anammox, and Mainstream Anaerobic Treatment. Environ. Sci. Technol. 53, 12935–12944.

9.Cookney, J., Mcleod, A., Mathioudakis, V., Ncube, P., Soares, A., Jefferson, B., McAdam, E.J., 2016. Dissolved methane recovery from anaerobic effluents using hollow fibre membrane contactors. J. Memb. Sci. 502, 141–150.

10. Cruz, H., Law, Y.Y., Guest, J.S., Rabaey, K., Batstone, D., Laycock, B., Verstraete, W., Pikaar, I., 2019. Mainstream ammonium recovery to advance sustainable urban wastewater management. Environ. Sci. Technol. 53, 11066–11079.

11. Dai, W., Xu, X., Liu, B., Yang, F., 2015. Toward energy-neutral wastewater treatment: A membrane combined process of anaerobic digestion and nitritation-anammox for biogas recovery and nitrogen removal. Chem. Eng. J. 279, 725–734.

12.EMBER, 2020. https://ember-climate.org/carbon-price-viewer/. (Accessed 8/9/2020)

13. Evans, P.J., Parameswaran, P., Lim, K., Bae, J., Shin, C., Ho, J., McCarty, P.L., 2019.

A comparative pilot-scale evaluation of gas-sparged and granular activated carbon-fluidized anaerobic membrane bioreactors for domestic wastewater treatment. Bioresour. Technol. 288, 120949.

14. Fernández-Arévalo, T., Lizarralde, I., Fdz-Polanco, F., Pérez-Elvira, S.I., Garrido, J.M., Puig, S., Poch, M., Grau, P., Ayesa, E., 2017. Quantitative assessment of energy and resource recovery in wastewater treatment plants based on plant-wide simulations. Water Res. 118, 272–288.

15. Ferrer, J., Pretel, R., Durán, F., Giménez, J.B., Robles, A., Ruano, M. V., Serralta, J., Ribes, J., Seco, A., 2015. Design methodology for submerged anaerobic membrane bioreactors (AnMBR): A case study. Sep. Purif. Technol. 141, 378–386.

16. Giménez, J.B., Robles, A., Carretero, L., Durán, F., Ruano, M. V., Gatti, M.N., Ribes, J., Ferrer, J., Seco, A., 2011. Experimental study of the anaerobic urban wastewater treatment in a submerged hollow-fibre membrane bioreactor at pilot scale. Bioresour. Technol. 102, 8799–8806.

17. Guest, J.S., Skerlos, S.J., Barnard, J.L., Beck, M.B., Daigger, G.T., Hilger, H., Jackson, S.J., Karvazy, K., Kelly, L., Macpherson, L., Mihelcic, J.R., Pramanik, A., Raskin, L., Van Loosdrecht, M.C.M., Yeh, D., Love, N.G., 2009. A new planning and design paradigm to achieve sustainable resource recovery from wastewater. Environ. Sci. Technol. 43, 6126–6130.

18. Harclerode, M., Doody, A., Brower, A., Vila, P., Ho, J., Evans, P.J., 2020. Life cycle assessment and economic analysis of anaerobic membrane bioreactor whole-plant configurations for resource recovery from domestic wastewater. J. Environ. Manage. 269, 110720.

19. Henze, M., van Loosdrecht, M.C.M., Ekama, G.A., Brdjanovic, D., 2008. Biological wastewater treatment : principles, modelling and design. IWA Publishing, London.

20. Huang, X., Guida, S., Jefferson, B., Soares, A., 2020. Economic evaluation of ion-exchange processes for nutrient removal and recovery from municipal wastewater. npj Clean Water 3, 7.

21. Iglesias, R., Simón, P., Moragas, L., Arce, A., Rodriguez-Roda, I., 2017. Cost comparison of full-scale water reclamation technologies with an emphasis on membrane bioreactors. Water Sci. Technol. 75, 2562–2570.

22. Kwak, W., Rout, P.R., Lee, E., Bae, J., 2020. Influence of hydraulic retention time and temperature on the performance of an anaerobic ammonium oxidation fluidized bed membrane bioreactor for low-strength ammonia wastewater treatment. Chem. Eng. J. 386, 123992.

23. Lim, K., Evans, P.J., Parameswaran, P., 2019. Long-Term Performance of a Pilot-Scale Gas-Sparged Anaerobic Membrane Bioreactor under Ambient Temperatures for Holistic Wastewater Treatment. Environ. Sci. Technol. 53, 7347–7354.

24. Macintosh, C., Astals, S., Sembera, C., Ertl, A., Drewes, J.E., Jensen, P.D., Koch, K., 2019. Successful strategies for increasing energy self-sufficiency at Grüneck wastewater treatment plant in Germany by food waste co-digestion and improved aeration. Appl. Energy 242, 797–808.

25. Madden, P., Al-Raei, A.M., Enright, A.M., Chinalia, F.A., de Beer, D., O'Flaherty, V., Collins, G., 2014. Effect of sulfate on low-temperature anaerobic digestion. Front. Microbiol. 5.

26. Metcalf & Eddy, 2014. Wastewater Engineering: Treatment and Resource Recovery, fifth ed. McGraw Hill, New York.

27. 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.

28. Papapetrou, M., Cipollina, A., Commare, U. La, Micale, G., Zaragoza, G., Kosmadakis, G., 2017. Assessment of methodologies and data used to calculate desalination costs. Desalination 419, 8–19.

29. Pretel, R., Durán, F., Robles, A., Ruano, M. V., Ribes, J., Serralta, J., Ferrer, J., 2015a. Designing an AnMBR-based WWTP for energy recovery from urban wastewater: The role of primary settling and anaerobic digestion. Sep. Purif. Technol. 156, 132–139.

30. Pretel, R., Shoener, B.D., Ferrer, J., Guest, J.S., 2015b. Navigating environmental, economic, and technological trade-offs in the design and operation of submerged anaerobic membrane bioreactors (AnMBRs). Water Res. 87, 531–541.

31. Qasim, S.R., 1999. Wastewater treatment plants : planning, design, and operation, second ed. Technomic Publishing Co, Lancaster.

32. Robles, Á., Aguado, D., Barat, R., Borrás, L., Bouzas, A., Giménez, J.B., Martí, N., Ribes, J., Ruano, M.V., Serralta, J., Ferrer, J., Seco, A., 2020. New frontiers from removal to recycling of nitrogen and phosphorus from wastewater in the Circular Economy. Bioresour. Technol. 300, 122673.

33. Rongwong, W., Wongchitphimon, S., Goh, K., Wang, R., Bae, T.H., 2017. Transport properties of CO2 and CH4 in hollow fiber membrane contactor for the recovery of biogas from anaerobic membrane bioreactor effluent. J. Memb. Sci. 541, 62–72.

34. Sanchis-Perucho, P., Robles, Á., Durán, F., Ferrer, J., Seco, A., 2020. PDMS membranes for feasible recovery of dissolved methane from AnMBR effluents. J. Memb. Sci. 604, 118070.

35. 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.

36. Seco, A., Mateo, O., Zamorano-López, N., Sanchis-Perucho, P., Serralta, J., Martí, N., Borrás, L., Ferrer, J., 2018. Exploring the limits of anaerobic biodegradability of urban wastewater by AnMBR technology. Environ. Sci. Water Res. Technol. 4, 1877–1887.

37. Serrano, A., Peces, M., Astals, S., Villa-Gómez, D.K., 2019. Batch assays for biological sulfate-reduction: a review towards a standardized protocol. Crit. Rev. Environ. Sci. Technol. 50, 1195–1223.

38. Sethunga, G.S.M.D.P., Karahan, H.E., Wang, R., Bae, T.H., 2019. PDMS-coated porous PVDF hollow fiber membranes for efficient recovery of dissolved biomethane from anaerobic effluents. J. Memb. Sci. 584, 333–342.

39. Sheik, A.R., Muller, E.E.L., Wilmes, P., Clark, K.B., Zhang, X., 2014. A hundred years of activated sludge: time for a rethink. Frointiers Microbiol. 5, 1–7.

40. Shin, C., Bae, J., 2018. Current status of the pilot-scale anaerobic membrane bioreactor treatments of domestic wastewaters: A critical review. Bioresour. Technol. 247, 1038–1046.

41. Shoener, B.D., Zhong, C., Greiner, A.D., Khunjar, W.O., Hong, P.Y., Guest, J.S., 2016. Design of anaerobic membrane bioreactors for the valorization of dilute organic carbon waste streams. Energy Environ. Sci. 9, 1102–1112.

42. Smith, A.L., Skerlos, S.J., Raskin, L., 2013. Psychrophilic anaerobic membrane bioreactor treatment of domestic wastewater. Water Res. 47, 1655–1665.

43. Smith, A.L., Stadler, L.B., Cao, L., Love, N.G., Raskin, L., Skerlos, S.J., 2014. Navigating Wastewater Energy Recovery Strategies: A Life Cycle Comparison of

Anaerobic Membrane Bioreactor and Conventional Treatment Systems with Anaerobic Digestion. Environ. Sci. Technol. 48, 5972–5981.

44. Song, X., Luo, W., McDonald, J., Khan, S.J., Hai, F.I., Guo, W., Ngo, H.H., Nghiem, L.D., 2018. Effects of sulphur on the performance of an anaerobic membrane bioreactor: Biological stability, trace organic contaminant removal, and membrane fouling. Bioresour. Technol. 250, 171–177.

45. Stuckey, D.C., 2012. Recent developments in anaerobic membrane reactors. Bioresour. Technol. 122, 137–148.

46. Taboada-Santos, A., Rivadulla, E., Paredes, L., Carballa, M., Romalde, J., Lema, J.M., 2020. Comprehensive comparison of chemically enhanced primary treatment and high-rate activated sludge in novel wastewater treatment plant configurations. Water Res. 169.

47. Verstraete, W., Van de Caveye, P., Diamantis, V., 2009. Maximum use of resources present in domestic “used water.” Bioresour. Technol. 100, 5537–5545.

48. Vinardell, S., Astals, S., Peces, M., Cardete, M.A., Fernández, I., Mata-Alvarez, J., Dosta, J., 2020a. Advances in anaerobic membrane bioreactor technology for municipal wastewater treatment: A 2020 updated review. Renew. Sustain. Energy Rev. 130, 109936.

49. Vinardell, S., Astals, S., Mata-Alvarez, J., Dosta, J., 2020b. Techno-economic analysis of combining forward osmosis-reverse osmosis and anaerobic membrane bioreactor technologies for municipal wastewater treatment and water production. Bioresour. Technol. 297, 122395.

50. Xiao, K., Liang, S., Wang, X., Chen, C., Huang, X., 2019. Current state and challenges of full-scale membrane bioreactor applications: A critical review. Bioresour. Technol. 271, 473–481.

Table 1. Process units included in each scenario.

	PT	PS	AnMBR	DM	PN/AMX	PP	TK	AD	CG	CHP
Scenario 1	x	-	x	x	-	-	x	-	x	x
Scenario 2A	x	x	x	x	-	-	x	x	x	x
Scenario 2B	x	x	x	x	-	-	x	-	x	x
Scenario 2C	x	-	x	x	-	-	x	x	x	x
Scenario 3	x	-	x	x	x	x	x	x	x	x

PT: Preliminary treatment; PS: Primary settler; AnMBR: Anaerobic membrane bioreactor; DM: Degassing membrane; PN/AMX: Partial nitritation-Anammox; PP: Phosphorus precipitation; TK: Thickener; AD: Sidestream anaerobic digestion; CG: Centrifuge; CHP: Combined heat and power unit.

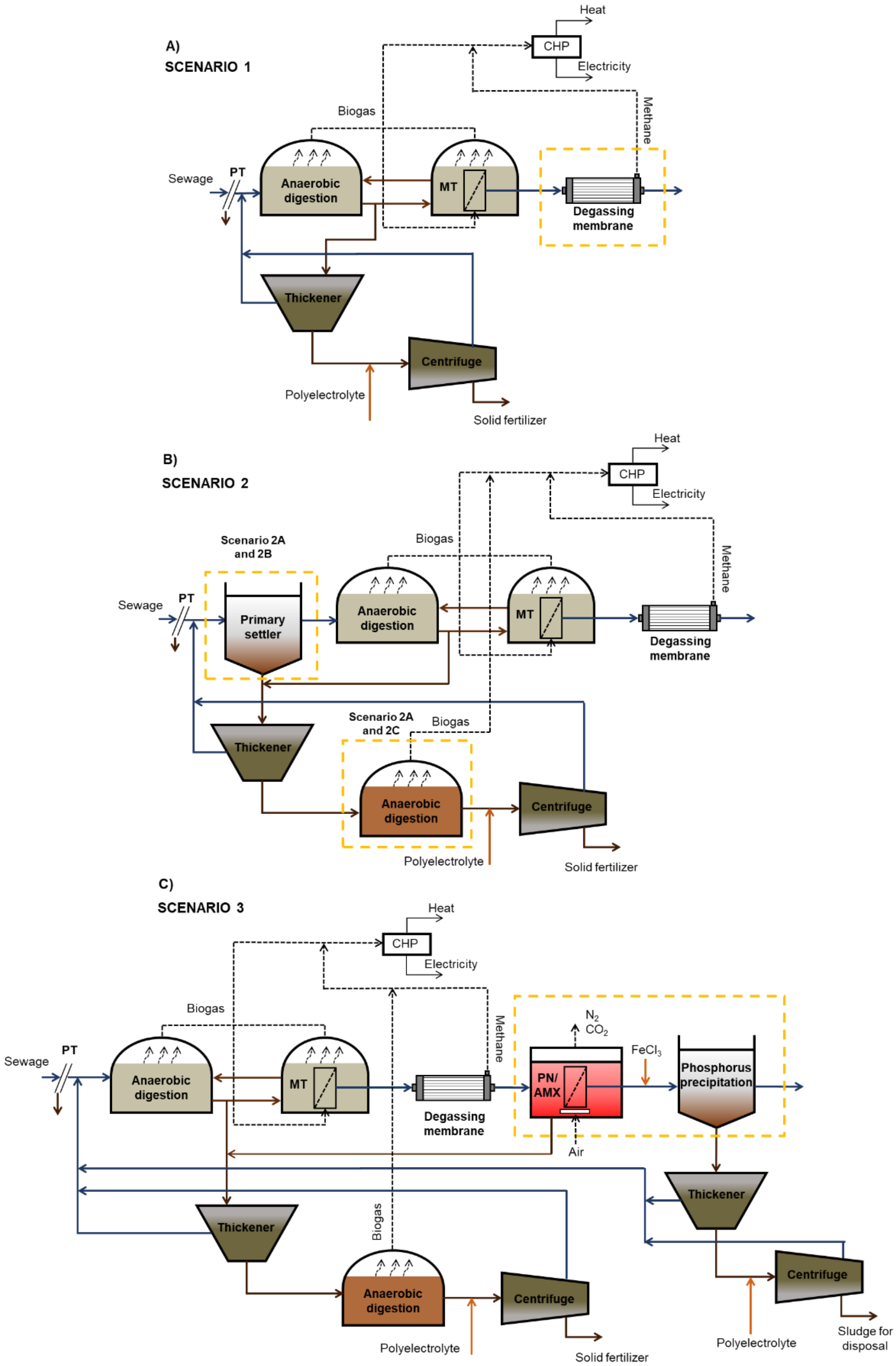

Figure 1. Schematic representation of the different scenarios. (Scenario 1) AnMBR and degassing membrane; (Scenario 2) AnMBR, degassing membrane, primary settler and sidestream anaerobic digestion; (Scenario 3A) AnMBR, degassing membrane, sidestream anaerobic digestion, and nutrients treatment. (PT: Preliminary treatment; MT: Membrane tank; CHP: Combined heat and power unit; PN/AMX: Partial nitritation-Anammox).

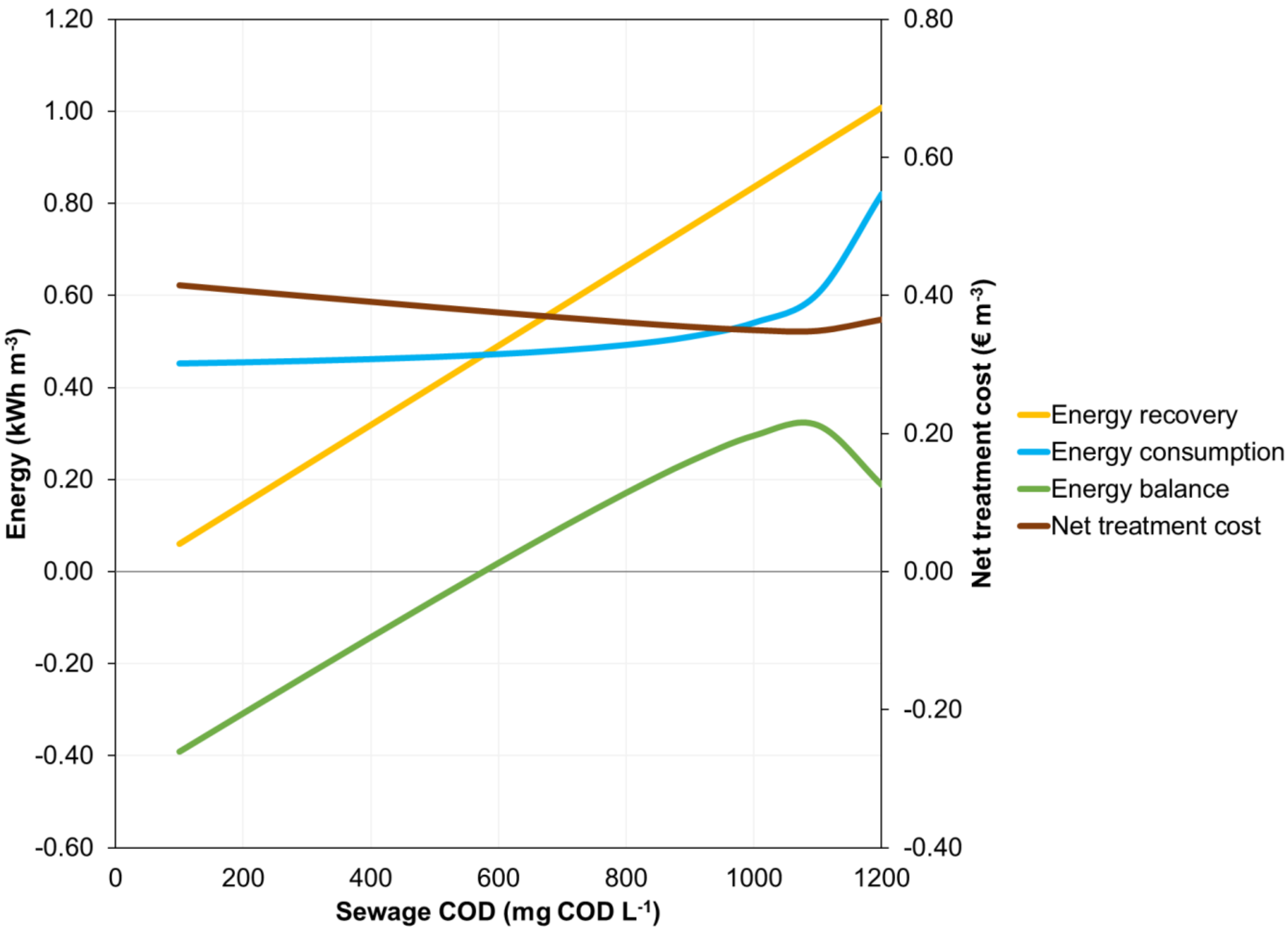

Figure 2. Energy balance and net treatment cost of Scenario 1 for different sewage COD concentrations ($COD:SO_4^{2-}$-S=57).

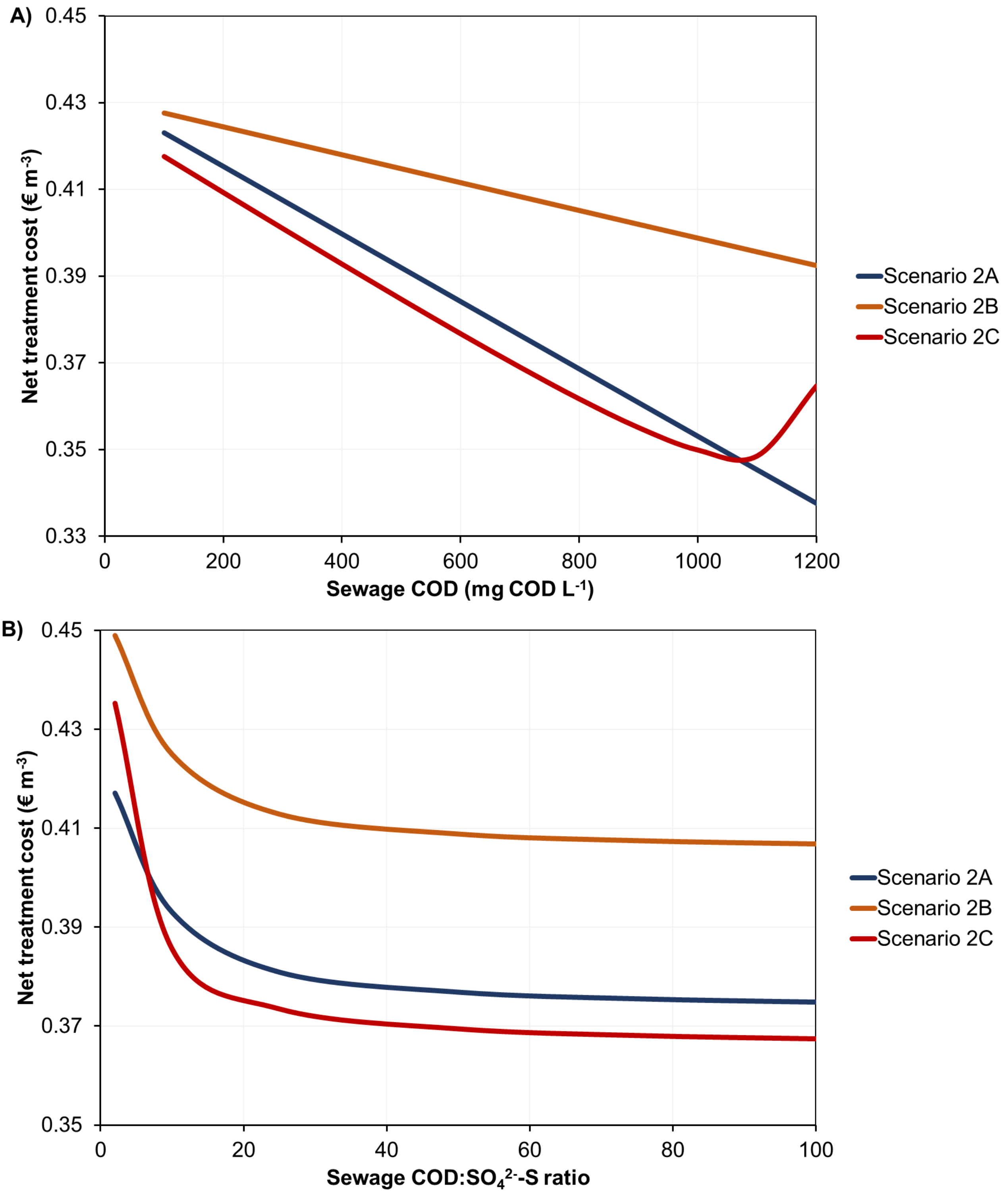

Figure 3. Net treatment cost of Scenario 2A, Scenario 2B and Scenario 2C. (A) Influence of sewage COD concentration ($COD:SO_4^{2-}$-S=57); (B) Influence of $COD:SO_4^{2-}$-S ratio (700 mg COD L^{-1}).

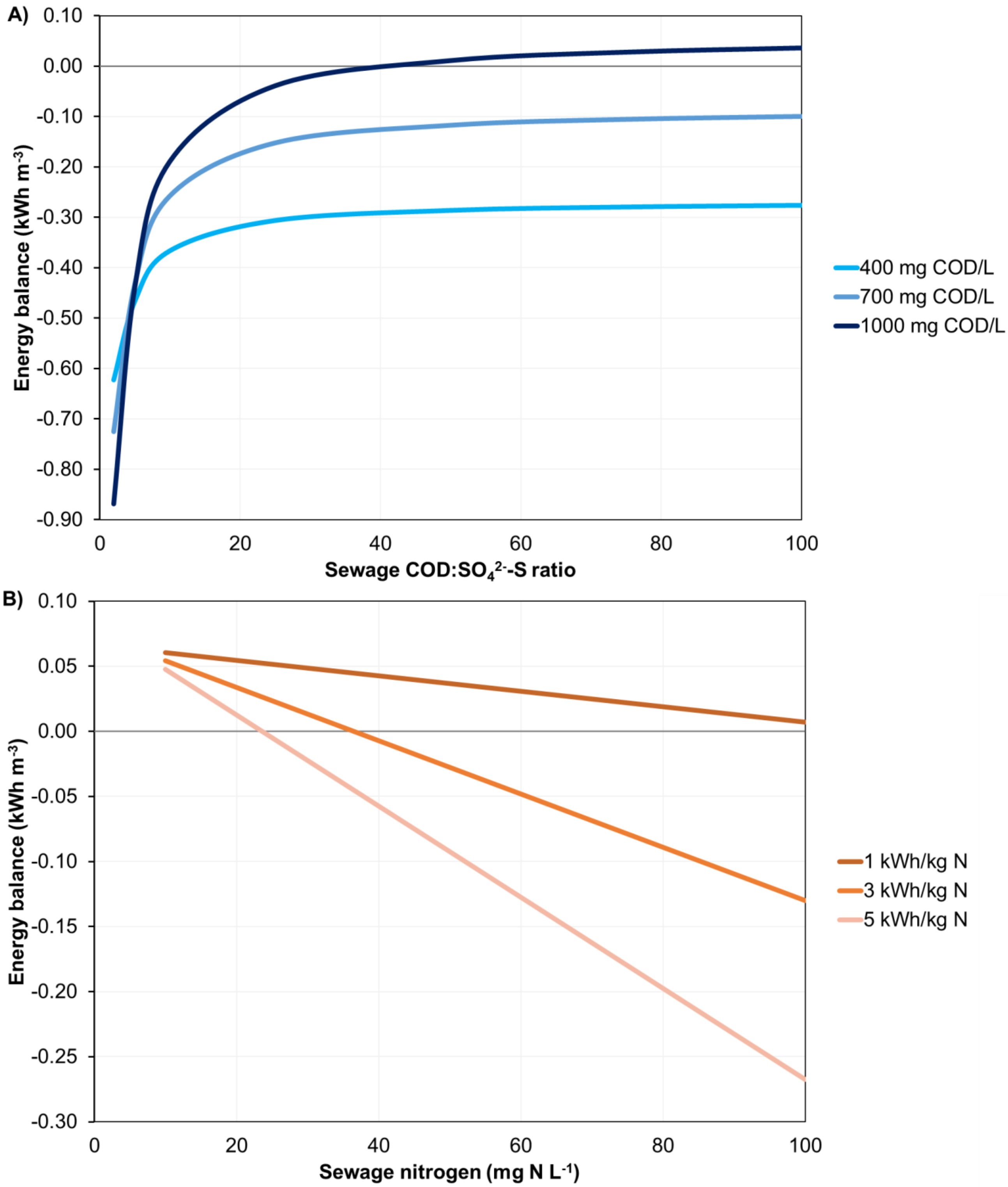

Figure 4. Energy balance of Scenario 3. (A) Influence of $COD:SO_4^{2-}$ ratio for three sewage COD concentrations (400, 700 and 1000 mg COD L^{-1}). (B) Influence of PN-Anammox energy consumptions (1, 3 and 5 kWh kg^{-1} N) for different sewage nitrogen concentrations (700 mg COD L^{-1}; $COD:SO_4^{2-}$-S=57; COD:P=51).

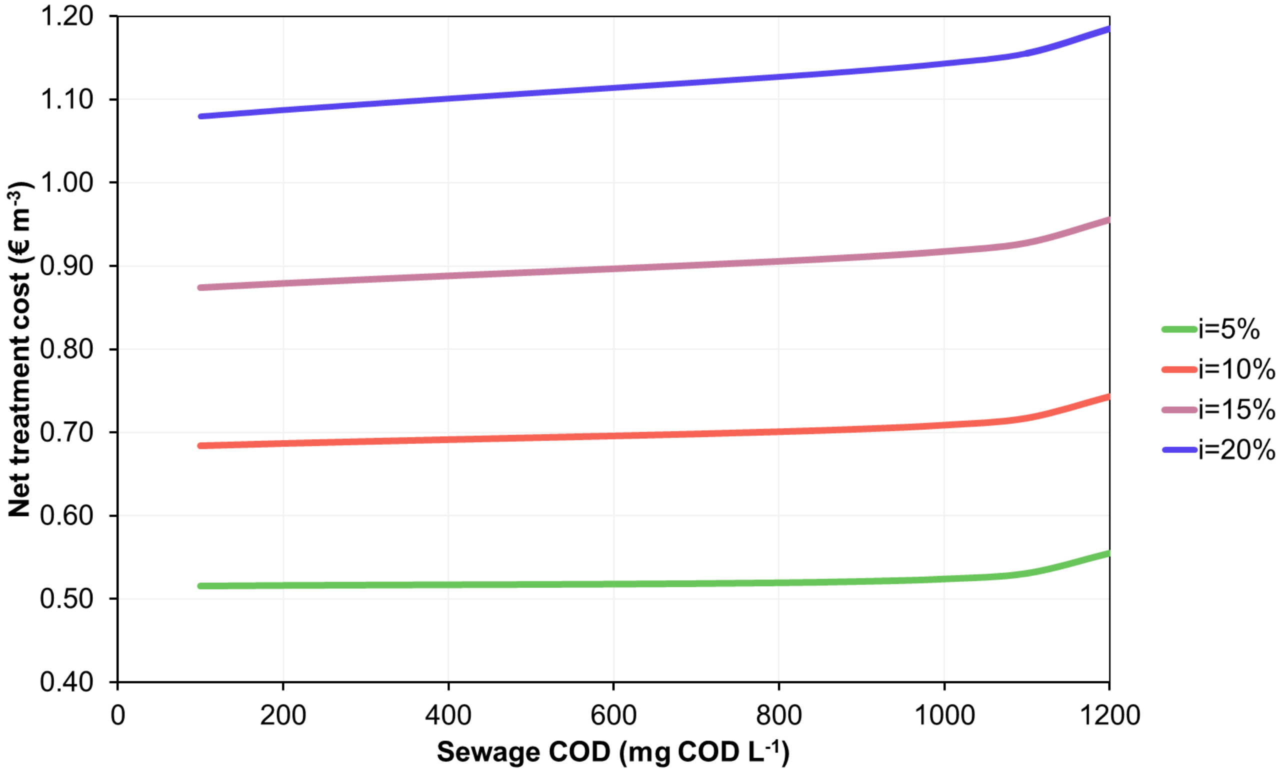

Figure 5. Net treatment cost of Scenario 3 for different sewage COD concentrations and considering discount rates of 5, 10, 15, and 20 % (COD:SO_4^{2-}-S=57; COD:N= 12.5; COD:P=51).

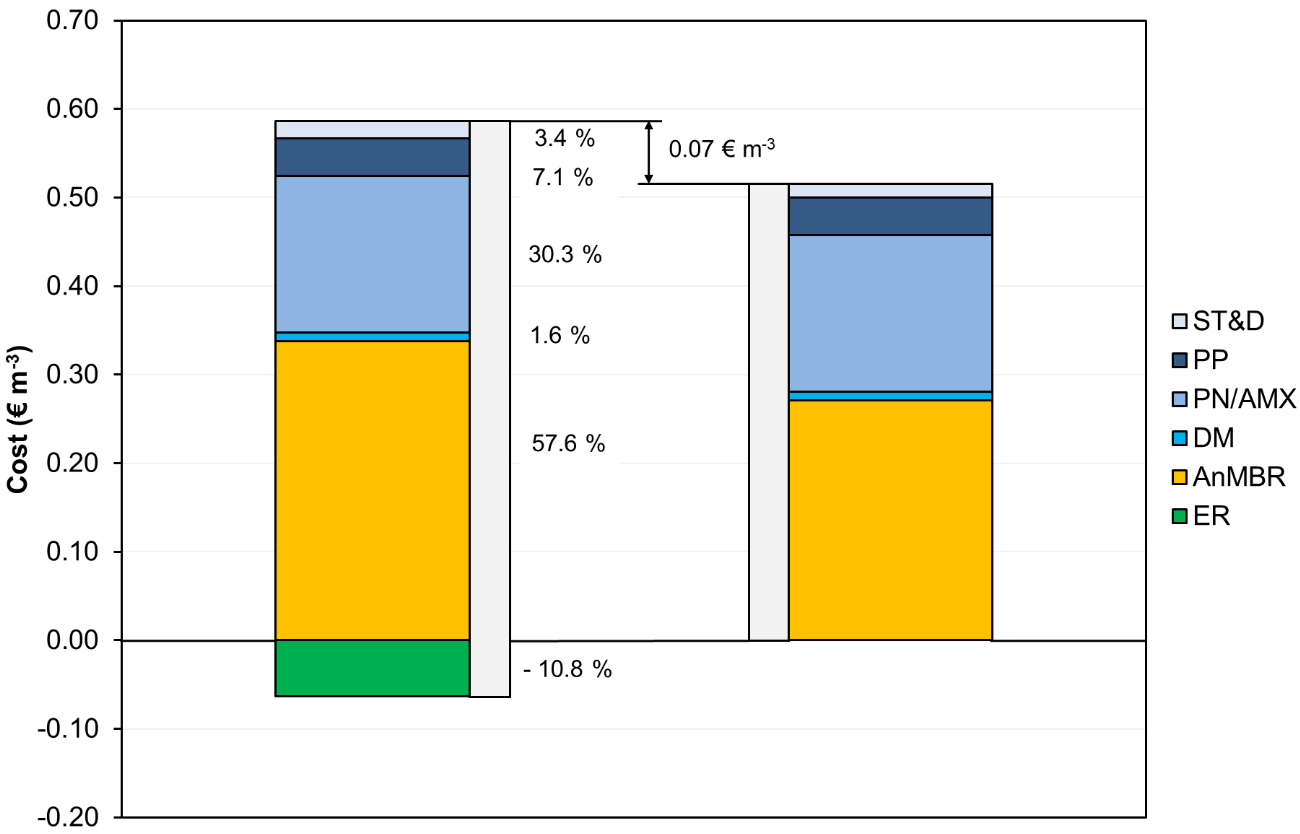

Figure 6. Cost distribution of Scenario 3 for a sewage COD concentration of 700 mg COD L^{-1} (COD:SO_4^{2-}-S=57; COD:N= 12.5; COD:P=51). (left) Without including electricity revenue (treatment cost); (right) including electricity revenue (net treatment cost). (ST&D: Sludge treatment and disposal; PP: Phosphorus precipitation; PN/AMX: Partial nitritation-Anammox; DM: Degassing membrane; AnMBR: Anaerobic membrane bioreactor; ER electricity revenue).