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Co-digestion of sewage sludge and food waste in a wastewater treatment plant based on mainstream anaerobic membrane bioreactor technology: A techno-economic evaluation

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HIGHLIGHTS

- The economic feasibility to co-digest sewage sludge and food waste was evaluated.
- The higher electricity revenue offsets the higher cost in co-digestion scenarios.
- Treating nutrient backloads in the sidestream was costlier than in the mainstream.
- Biosolids disposal cost was the most important gross cost contributor.
- Food waste gate fee had a noticeable impact on co-digestion economic feasibility.

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ABSTRACT

The implementation of anaerobic membrane bioreactor as mainstream technology would reduce the load of sidestream anaerobic digesters. This research evaluated the techno-economic implications of co-digesting sewage sludge and food waste in such wastewater treatment plants to optimise the usage of the sludge line infrastructure. Three organic loading rates (1.0, 1.5 and 2.0 kg VS m^{-3} d^{-1}) and different strategies to manage the additional nutrients backload were considered. Results showed that the higher electricity revenue from co-digesting food waste offsets the additional costs of food waste acceptance infrastructure and biosolids disposal. However, the higher electricity revenue did not offset the additional costs when the nutrients backload was treated in the sidestream (partial-nitritation/anammox and struvite precipitation). Biosolids disposal was identified as the most important gross cost contributor in all the scenarios. Finally, a sensitivity analysis showed that food waste gate fee had a noticeable influence on co-digestion economic feasibility.

1. Introduction

Wastewater treatment plants (WWTPs) are essential facilities in our society. Aerobic-based technologies, which are widely used in WWTPs, have successfully improved worldwide sanitation for more than a century (Van Loosdrecht and Brdjanovic, 2014). However, these technologies are not suitable in the current context of climate change and resource depletion as they fail to recover the resources contained in municipal sewage (Akyol et al., 2020). Pursuing sustainable technologies able to valorise these resources is required to promote the circular economy in WWTPs (Guest et al., 2009).

At present, anaerobic digestion is widely used to transform the organic matter contained in sewage sludge into biogas (Foladori et al., 2015). However, the development of technologies able to provide an effective retention of the slow growing anaerobic microorganisms at ambient temperature has broadened the applicability of anaerobic digestion to the mainstream of the WWTP (Akyol et al., 2020; Stazi and Tomei, 2018). Anaerobic membrane bioreactor (AnMBR) is an emerging technology for municipal sewage treatment where the membrane provides an excellent retention of the anaerobic microorganisms in the bioreactor while providing a high-quality effluent suitable for reuse (Vinardell et al., 2020).

The transition from aerobic-WWTPs to AnMBR-WWTPs is challenging since most aerobic-WWTPs are already constructed and under

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operation. However, existing facilities are ageing, which makes it necessary to retrofit WWTPs to meet the stricter discharge requirements and achieve a cost-effective long-term operation (Garrido-Baserba et al., 2018; Tian et al., 2020). Infrastructure retrofit could allow reducing the initial investment and land use in comparison with a newly constructed AnMBR-WWTP. Understanding the main implications of retrofitting an existing WWTP to implement mainstream AnMBR is important to make an efficient use of the different process units in the retrofitted plant.

A lower sludge production is one of the main implications of implementing mainstream AnMBR technology in a WWTP. This is because the biomass yield of anaerobic microorganisms (ca. 0.10 $gCOD_x/gCOD_s$) is significantly lower than the biomass yield of aerobic microorganisms (ca. 0.60 $gCOD_x/gCOD_s$) (Henze et al., 2008; Stazi and Tomei, 2018). Accordingly, the organic loading rates (OLR) of the sidestream anaerobic digester (AD) would be reduced after retrofitting the aerobic-WWTP to an AnMBR-WWTP. This would result in a lower biogas production and a poor operation of the existing sludge line infrastructure. Therefore, it is important to look for strategies to increase biogas production and take full advantage of the existing infrastructure in the retrofitted AnMBR-WWTP.

Anaerobic co-digestion (AcoD) is a strategy to increase biogas production of the sidestream AD in the retrofitted AnMBR-WWTP (Nghiem et al., 2017; Vinardell et al., 2021). AcoD consists of the combined digestion of sewage sludge with one or more co-substrates to increase biogas production (Macintosh et al., 2019; Mata-Alvarez et al., 2014). Food waste is the most used co-substrate in WWTP full-scale applications due to its easy accessibility and relatively high methane yield (Nghiem et al., 2017). The high biodegradability of food waste allows increasing the biogas production with a minor increase in the amount of biosolids to be managed (Capson-Tojo et al., 2016; Nghiem et al., 2017). Accordingly, AcoD of sewage sludge and food waste has the potential to increase the profitability of the AnMBR-WWTP.

The economic and technical feasibility of food waste AcoD cannot be limited to its capacity to increase biogas and power production since AcoD has a plant-wide impact on the WWTP (Aichinger et al., 2015; Macintosh et al., 2019). In the sludge line, the higher amount of biosolids would increase the consumption of polyelectrolyte and the biosolids management cost (Aichinger et al., 2015). In the mainstream, the higher nutrients concentration in the backload due to AcoD increases the consumption of energy and chemicals for their removal (Sembera et al., 2019). Moreover, food waste AcoD involves the implementation of a new installation for food waste acceptance and processing, as well as the negotiation of a gate or delivery fee for food waste (Nghiem et al., 2017; Sembera et al., 2019). The economic evaluation of sewage sludge and food waste AcoD needs to consider all these factors to have a reliable estimation of the costs associated with the implementation of AcoD.

Some studies have analysed the economic feasibility of co-digesting sewage sludge and food waste in a WWTP (Morelli et al., 2020; Sembera et al., 2019). However, an economic analysis evaluating the co-digestion of sewage sludge and food waste in the sidestream AD of a future AnMBR-WWTP has not yet been analysed. Evaluating the economic drivers and constraints of implementing AcoD strategies in a retrofitted AnMBR-WWTP is important to better understand the impact that the implementation of mainstream AnMBR has on sludge line and on the sidestream AD biogas production.

This theoretical study aims to analyse the techno-economic feasibility of implementing sewage sludge and food waste AcoD in the sludge line of a retrofitted AnMBR-WWTP. To this end, different factors influencing the economics of AcoD were considered such as biogas production, nutrients backload, combined heat and power (CHP) unit upgrading, polyelectrolyte consumption, dewatering energy consumption, biosolids management, food waste acceptance installation and food waste gate/delivery fee.

2. Methodology

2.1. Scenarios definition

Fig. 1 shows the different scenarios evaluated in this study. A high-sized WWTP with a population equivalent (PE) capacity of 500,000 PE (100,000 m^3 d^{-1}) was considered in this economic evaluation. The four scenarios evaluated in the present study are described below:

- Baseline Scenario represented the WWTP before retrofitting (Fig. 1A). The sewage sludge consisted of a mixture of primary and secondary sludge. The secondary sludge was produced in an activated sludge (AS) process using a modified Ludzack-Ettinger configuration (see supplementary material). The thickened sewage sludge (50% primary sludge and 50% secondary sludge on VS basis) was treated in a sidestream AD working at an OLR of 1.0 kg VS m^{-3} d^{-1}. The biogas produced in the sidestream AD was combusted in a CHP unit. The digestate was dewatered with a centrifuge before its final disposal.
- Scenario 1 was the retrofitted AnMBR-WWTP without implementing AcoD in the sidestream AD (Fig. 1A). In this scenario, the AS from the Baseline Scenario was retrofitted to an AnMBR and partial nitritation-anammox (PN/Anammox) processes for the removal of organic matter and nitrogen, respectively (see supplementary material). The AnMBR was a two-stage system where the membrane was submerged in a separated membrane tank. The AnMBR was considered to be operated at an HRT and SRT of 1 and 60 days, respectively (Vinardell et al., 2021). The sewage sludge was a mixture of primary and secondary sludge. The secondary sludge consisted of the wasted sludge from the mainstream AnMBR and excess sludge from PN/Anammox. In Scenario 1, no additional equipment nor equipment upgrading was needed for the sludge line since the existing infrastructure was oversized due to the lower amount of sludge produced in the AnMBR and PN/Anammox processes in comparison with the AS process. Therefore, Scenario 1 had the same sludge line infrastructure than the Baseline Scenario (Fig. 1A). In this scenario, the OLR of the sidestream AD was 0.63 kg VS m^{-3} d^{-1} considering (i) the amount of primary and secondary sludge produced and (ii) the volume of the exiting sidestream AD.
- Scenario 2 was the retrofitted AnMBR-WWTP with AcoD of sewage sludge and food waste in the sidestream AD (Fig. 1B). In Scenario 2, new infrastructure for food waste acceptance was necessary and the CHP unit was upgraded to adapt the existing infrastructure to the higher biogas production. In this scenario, three AcoD alternatives were evaluated based on the total OLR ($OLR_{sludge} + OLR_{food\ waste}$) of the sidestream AD: (A) 1.0 kg VS m^{-3} d^{-1}, (B) 1.5 kg VS m^{-3} d^{-1} and (C) 2.0 kg VS m^{-3} d^{-1}. Considering that the sludge OLR was 0.63 kg VS m^{-3} d^{-1}, the OLR provided by the food waste was 0.37, 0.87 and 1.37 kg VS m^{-3} d^{-1}, respectively. The OLR range was chosen based on the OLR of full-scale digesters at WWTPs co-digesting sewage sludge and food waste (Aichinger et al., 2015; Koch et al., 2016; Macintosh et al., 2019). The first alternative (1.0 kg VS m^{-3} d^{-1}) only added the amount of food waste needed to compensate the VS load reduced by the AnMBR.
- Scenario 3 was an extension of Scenario 2 and included nutrients treatment of the centrate in the sidestream (Fig. 1C). Specifically, PN-Anammox and struvite crystallisation were used to reduce the impact of nutrients backload on the mainstream of the WWTP (Caffaz et al., 2008; Rodriguez-Garcia et al., 2014). Struvite crystallisation was placed after PN/Anammox process since this configuration reduces the sodium hydroxide requirements for struvite crystallisation as a result of the alkalinity consumption in the previous PN/Anammox process (Campos et al., 2017). In Scenario 3, the same three OLRs of Scenario 2 were evaluated: (A) 1.0 kg VS m^{-3} d^{-1}, (B) 1.5 kg VS m^{-3} d^{-1} and (C) 2.0 kg VS m^{-3} d^{-1}.

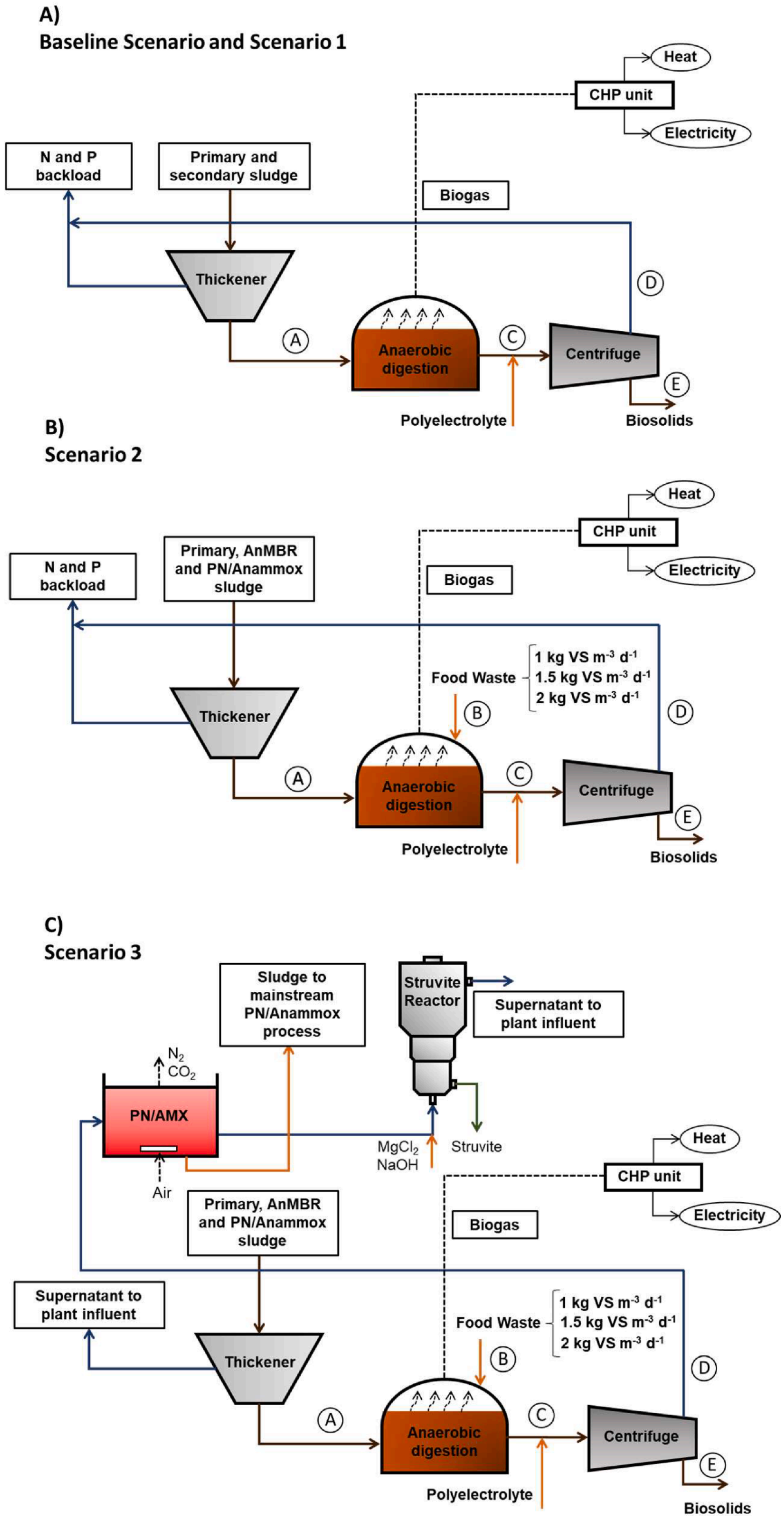

Fig. 1. Schematic representation of the different scenarios under study. (top) Baseline Scenario (aerobic-WWTP) and Scenario 1 (AnMBR-WWTP without AcoD); (middle) Scenario 2 (AnMBR-WWTP including food waste AcoD); (bottom) Scenario 3 (AnMBR-WWTP with food waste AcoD and sidestream nutrients backload treatment).

2.2. Sludge and food waste production and characterisation

The sludge production for the different scenarios was calculated considering a municipal sewage with a chemical oxygen demand (COD) concentration of 700 mg COD L^{-1} and 56 mg N L^{-1}, respectively (Henze et al., 2008). The COD content of sewage, before primary settling, was fractioned in biodegradable soluble COD (36%), biodegradable particulate COD (40%), inert soluble COD (4%) and inert particulate COD (20%) (Henze et al., 2008).

The sludge production of the aerobic-WWTP (before retrofitting) was calculated to obtain the capacity of the existing sludge line infrastructure. The primary sludge production was calculated assuming that 67% of the particulate COD was separated in the primary settler. This means that 40% of the total sewage COD was separated in the primary settler (Henze et al., 2008). The secondary sludge production of the aerobic-WWTP was calculated through steady-state equations considering the growth rate of autotrophic nitrifiers, heterotrophic denitrifiers and heterotrophic oxidisers (see supplementary material). The thickened sewage sludge consisted of a total solids (TS) concentration of 3.5% (Astals et al., 2013). Sewage sludge composition was obtained as the average of the seven different sewage sludges reported by Astals et al. (2013).

In the retrofitted AnMBR-WWTP, the sewage sludge production was lower than in the aerobic-WWTP. The sludge produced in the AnMBR and PN/Anammox processes was calculated through steady-state equations (see supplementary material). The sewage sludge was mixed with food waste in Scenario 2 and Scenario 3. The amount of food waste added was calculated from the OLR of each alternative. The food waste had a TS concentration of 23.4% and a VS/TS ratio of 91.0%, which was obtained as the average of seven different food wastes reported in literature (see supplementary material).

2.3. Modelling AD performance

Model equations for a continuous stirred tank reactor (CSTR) at steady-state conditions were used to calculate the VS removal, methane yield and nutrients solubilisation in the sidestream AD. The model was applied for each substrate, namely, primary sludge, secondary sludge and food waste. The sewage sludge composition was used as representative for both primary and secondary sludge due to the limited data available in literature. No synergism was considered in the AcoD process. The model parameters (i.e. first-order kinetic constant and biodegradability) used for each substrate were obtained as the average of five different studies (see supplementary material).

The biodegradable VS concentration in the AD effluent was calculated by using a mass balance in VS (Eq. (1)), which considers that the degradation of VS over time follows a first-order kinetic (Garcia-Heras, 2003).

$$S_{eff,bio} = S_{0,bio} \cdot \frac{1}{1 + k \cdot HRT} \quad (1)$$

where $S_{eff,bio}$ is the biodegradable VS concentration in the AD effluent (g VS L^{-1}), $S_{0,bio}$ is the biodegradable VS concentration in the AD influent (g VS L^{-1}), k is the first-order kinetic constant (d^{-1}), and HRT is the hydraulic retention time (d).

The methane yield of a CSTR digester can be calculated at steady-state conditions by means of Eq. (2) as shown elsewhere (Garcia-Heras, 2003).

$$B = B_0 \cdot \frac{k \cdot HRT}{1 + k \cdot HRT} \quad (2)$$

where, besides the described above, B is the methane yield (mL CH_4 g^{-1} VS) and B_0 is the substrate ultimate methane yield (mL CH_4 g^{-1} VS).

Finally, the amount of NH_4^+-N and PO_4^{3-}-P in the AD effluent were calculated using Eq. (3) and Eq. (4), respectively. These equations consider that the release of nutrients is proportional to organic matter degradation.

$$N_{eff,NH_4^+} = N_{0,NH_4^+} N_{0,org} \cdot \frac{S_{0,bio} - S_{eff,bio}}{S} \quad (3)$$

$$P_{eff,PO_4^{3-}} = P_{0,PO_4^{3-}} + P_{0,org} \cdot \frac{S_{0,bio} - S_{eff,bio}}{S} \quad (4)$$

where $N^+_{eff,NH4}$ is the NH_4^+-N concentration in the AD effluent (g N L^{-1}), $N_{0,org}$ is the organic nitrogen concentration in the AD influent (g N L^{-1}), $N^+_{0,NH4}$ is the NH_4^+-N concentration in the AD influent (g N L^{-1}), $P_{eff,PO4}^{3-}$ is the PO_4^{3-}-P concentration in the AD effluent (g P L^{-1}), $P_{0,org}$ is the organic phosphorus concentration in the AD influent (g P L^{-1}), $P_{0,PO4}^{3-}$ is the PO_4^{3-}-P concentration in the AD influent (g P L^{-1}), $S_{eff,bio}$ is the biodegradable VS concentration in the AD effluent (g VS L^{-1}), $S_{0,bio}$ is the biodegradable VS concentration in the AD influent (g VS L^{-1}), and S is the total VS concentration in the AD influent (g VS L^{-1}). Organic matter and nutrient initial concentrations were calculated considering the flow of each substrate to the digester.

2.4. Costs and revenue calculation

The implementation of AcoD is expected to increase the revenue of the sidestream AD due to the higher biogas production. However, AcoD increases the capital and operating costs of the sludge line and the consumption of energy and chemical reagents in the mainstream to remove the nutrients backload. The most sensitive factors to AcoD were included in this economic evaluation. These factors were classified into four groups: (i) food waste acceptance, (ii) digestate dewatering and biosolids management, (iii) nutrients backload treatment and (iv) energy production. In this study, the costs and revenue that are not influenced by AcoD were not included since they are expected to be similar regardless of AcoD implementation (e.g. operation of the AS and AnMBR in the mainstream of the WWTP). The following subsections discuss the parameters and considerations used to calculate the costs and revenue for the different scenarios. The parameters used for cost and electricity revenue calculations can be found in the supplementary material.

2.4.1. Food waste acceptance

The use of food waste as co-substrate for AcoD requires the installation of a new infrastructure for food waste acceptance and processing as well as the negotiation of a gate/delivery fee for the co-substrate. These costs were obtained from the Grüneck WWTP (Germany), where co-digestion of sewage sludge and food waste is used since 2014. In this WWTP, the construction of the facility for food waste acceptance cost 150,000 € (Macintosh et al., 2019). The plant receives 2,100 t y^{-1} of food waste from a processing plant. The transportation cost from the processing plant to the Grüneck WWTP is paid by the WWTP at 3 € t^{-1} (Macintosh et al., 2019). However, the criteria to establish a gate/delivery fee is still unclear and differs depending on the WWTP and the food waste source (Sembera et al., 2019). In Section 3.3.4, the impact of gate/delivery fee on AcoD profitability was analysed through a sensitivity analysis.

2.4.2. Digestate dewatering and biosolids disposal

The digestate from the anaerobic digester was considered to be dewatered to 30% TS before its final disposal. Polyelectrolyte was dosed to improve sludge dewaterability and to achieve the final biosolids concentration. A polyelectrolyte dosage of 9 kg t^{-1} TS was considered (Aichinger et al., 2015). After polyelectrolyte dosage, the digestate was centrifuged at an energy consumption of 0.045 kWh kg^{-1} TSS (Pretel et al., 2014).

The biosolids were transported for its final disposal at 54 € t^{-1} TS, which represents the average cost in Europe for the transportation of dewatered digestate with a TS content of 30% (Foladori et al., 2015). The biosolids were used for agriculture at a cost of 93 € t^{-1} TS since land

Table 1
Main operation and flow data for the different scenarios under study.

			Baseline Scenario	Scenario 1	Scenario 2A	Scenario 2B	Scenario 2C	Scenario 3A	Scenario 3B	Scenario 3C
Description	WWTP configuration		Aerobic-WWTP	AnMBR-WWTP	AnMBR-WWTP	AnMBR-WWTP	AnMBR-WWTP	AnMBR-WWTP	AnMBR-WWTP	AnMBR-WWTP
	AcoD application		No	No	Yes	Yes	Yes	Yes	Yes	Yes
	OLR_{sludge} (kg VS m^{-3} d^{-1})		1.0	0.63	0.63	0.63	0.63	0.63	0.63	0.63
	$OLR_{food\ waste}$ (kg VS m^{-3} d^{-1})		–	–	0.37	0.87	1.37	0.37	0.87	1.37
	OLR_{total} (kg VS m^{-3} d^{-1})		1.0	0.63	1.0	1.5	2.0	1.0	1.5	2.0
	Nutrients backload treatment		Mainstream	Mainstream	Mainstream	Mainstream	Mainstream	Sidestream	Sidestream	Sidestream
Mass and volumetric flows[1]	(A) Sewage sludge	TS (%)	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5
		VS/TS (%)	75	75	75	75	75	75	75	75
		Q (m^3 d^{-1})	1,192	746	746	746	746	746	746	746
	(B) Food waste	TS (%)	–	–	23.4	23.4	23.4	23.4	23.4	23.4
		VS/TS (%)	–	–	91	91	91	91	91	91
		Q (m^3 d^{-1})	–	–	57	133	208	57	133	208
	(C) Digestate	TS (%)	2.5	2.3	2.6	3.0	3.3	2.6	3.0	3.3
		VS/TS (%)	64	60	61	63	64	61	63	64
		Q (m^3 d^{-1})	1,175	733	781	845	908	781	845	908
	(D) Centrate	$F_{NH4\text{-}N}$ (kg NH_4^+-N d^{-1})	1,051	776	1,037	1,375	1,702	1,037	1,375	1,702
		$F_{PO4\text{-}P}$ (kg PO_4^{3-}-P d^{-1})	242	162	222	301	378	222	301	378
		Q (m^3 d^{-1})	1,077	678	714	761	807	714	761	807
	(E) Biosolids	TS (%)	30	30	30	30	30	30	30	30
		VS/TS (%)	64	60	61	63	64	61	63	64
		Q (m^3 d^{-1})	98	55	67	84	101	67	84	101
Chemicals	Polyelectrolyte consumption (kg d^{-1})		274	155	189	236	284	189	236	284
	$MgCl_2{\cdot}6H_2O$ consumption (kg d^{-1})		–	–	–	–	–	1,313	1,824	2,319
	NaOH consumption (kg d^{-1})		–	–	–	–	–	331	439	541
Energy	Methane yield (L CH_4 kg^{-1} VS)		268	324	343	353	358	343	353	358
	Methane production (m^3 d^{-1})		8,389	6,345	10,887	16,903	22,858	10,887	16,903	22,858
	Biogas production (t d^{-1})		17	13	22	34	46	22	34	46
	Electricity production (kWh d^{-1})		30,505	23,072	39,587	61,462	83,117	39,587	61,462	83,117
	Energy requirements sludge line (kWh d^{-1})		3,634	2,487	3,254	4,260	5,247	3,484	4,603	5,702

[1]The different flows (A, B, C, D and E) are illustrated in Fig. 1.

application is still the main management route in Europe (Foladori et al., 2015). Therefore, the total disposal cost (transport + disposal) was 147 € t^{-1} TS. Nevertheless, the cost of biosolids disposal largely depends on its final use (i.e. agriculture, landfill, composting or incineration) and country, which may have a big impact on total costs. In Section 3.3.1, the impact of biosolids disposal cost was analysed through a sensitivity analysis.

2.4.3. Nutrients backload treatment

Food waste contains a high content of organic nitrogen and phosphorus, which are partially solubilised into ammonium and phosphate during anaerobic digestion (Nghiem et al., 2017). Accordingly, food waste AcoD increases the concentration of these compounds in the centrate. In this study, two approaches were considered to remove the nutrients backload of the centrate: (i) mainstream nutrients treatment (Baseline Scenario, Scenario 1 and 2) and (ii) sidestream nutrients treatment by PN/Anammox and struvite crystallisation (Scenario 3).

2.4.3.1. Mainstream nutrients treatment. Energy consumption for nitrogen removal and ferric chloride ($FeCl_3$) consumption for phosphorus precipitation were considered to calculate the cost to remove the nutrients backload in the mainstream of the WWTP. A specific energy consumption for mainstream nitrogen removal of 2.38 kWh kg^{-1}N was used according to Horstmeyer et al. (2018). It was considered that nitrification/denitrification (Baseline Scenario) and mainstream PN/Anammox (Scenario 1, 2 and 3) processes had the same specific energy consumption since PN/Anammox is still not fully optimised for mainstream nitrogen removal (Schaubroeck et al., 2015). The amount of ferric chloride necessary to precipitate phosphate was estimated considering that 30 mg $FeCl_3$ L^{-1} are needed to decrease phosphate concentration from ~ 2.3 mg PO_4^{3-}-P L^{-1} to ~ 0.2 mg PO_4^{3-}-P L^{-1} (Taboada-Santos et al., 2020).

2.4.3.2. Sidestream nutrients treatment. The PN/Anammox process was also selected for sidestream nitrogen removal of the centrate since it is an autotrophic nitrogen removal process suitable to treat streams with a low COD/N ratio (Guo et al., 2020; Vázquez-Padín et al., 2009). The PN/Anammox process was designed to treat a nitrogen loading rate (NLR) of 0.42 kg N m^{-3} d^{-1} and achieved a nitrogen removal efficiency of 89%, which are average values from full-scale PN/Anammox processes (Lackner et al., 2014; Schaubroeck et al., 2015). The sludge produced in the sidestream PN/Anammox system was transferred to the mainstream system to enrich its anammox and ammonia oxidising bacteria biomass of the full-scale PN/Anammox (Schaubroeck et al., 2015; Wett et al., 2013). The capital cost for PN/Anammox was assumed to be 1,600 €/(kg N/day), between the 1,300 and 1,900 €/(kg N/day) reported in literature (Van Eekert et al., 2012; Vandekerckhove et al., 2020). This capital cost range was obtained by dividing the initial investment (€) by the nitrogen load (kg N/day) reported by Van Eekert et al. (2012) and Vandekerckhove et al. (2020). The PN/Anammox specific energy consumption was 1.5 kWh kg^{-1}N, which is a typical energy consumption for nitrogen removal of the centrate (Lackner et al., 2014; Schaubroeck et al., 2015). Finally, the total operating cost of the PN/Anammox process was calculated considering a unit cost of 0.8 € kg^{-1}N (Van Eekert et al., 2012; Vandekerckhove et al., 2020).

Struvite crystallisation was used to recover the phosphorus from the centrate since this is the most mature technology for phosphorus recovery (Bolzonella et al., 2006; Münch and Barr, 2001) and struvite ($MgNH_4PO_4{\cdot}6H_2O$) can be valorised as a slow release fertiliser (Peng et al., 2018). An average capital cost of 10,000 €/(kg P/day) was considered (Vaneeckhaute et al., 2017). Phosphate removal efficiencies of 90% were considered for the struvite reactor (Peng et al., 2018). The energy consumption for struvite crystallisation was 5.9 kWh kg^{-1}P, between the 2.2 and 10 kWh kg^{-1}P reported in literature (Ghosh et al., 2019). Magnesium chloride hexahydrate ($MgCl_2{\cdot}6H_2O$) was used for struvite crystallization at a unit cost of 370 € t^{-1} (Bouzas et al., 2019). The $MgCl_2{\cdot}6H_2O$ dosage was calculated with the stoichiometric relationship with phosphate and considering that the centrate contained 27.2 mg Mg^{2+} L^{-1} (Campos et al., 2017). Sodium hydroxide (NaOH) was dosed to increase the pH from 7.3 to 9.0, which is the optimum pH for struvite crystallisation (Peng et al., 2018). The previous PN/Anammox process allowed reducing the NaOH consumption in the struvite crystalliser since it already consumes alkalinity (i.e. HCO_3^- and NH_4^+). In this research, a molar HCO_3^-:NH_4^+ ratio of 1:1 in the centrate was considered (Campos et al., 2017). Subsequently, the NaOH consumption was calculated through acid-base equilibrium after subtracting the alkalinity consumed in the PN/Anammox process. The NaOH cost was 620 € t^{-1} (Bouzas et al., 2019). No revenue was considered from the struvite produced since this is still managed as a waste in many countries (Peng et al., 2018). In Section 3.3.3, the impact of struvite commercialisation was evaluated through a sensitivity analysis.

2.4.4. Energy production

An electrical efficiency of 33% for the CHP unit was considered, which represents the average electrical efficiency reported in literature (Riley et al., 2020; Vinardell et al., 2021). The methane yield of the sidestream AD for each scenario was used for the energy calculations (see Section 2.3 for further details on methane yield calculations). The higher methane production in AcoD scenarios makes it necessary to upgrade the existing CHP unit to utilise all the produced biogas (minimise biogas flaring) and increase the WWTP energy production. The capital cost to upgrade existing CHP unit was calculated considering a unit cost of 712 € kW_{el}^{-1} (Riley et al., 2020; Smith et al., 2014). The operating cost of the CHP unit was 0.0119 € kWh_{el}^{-1} (Riley et al., 2020; Smith et al., 2014). A lifetime of 20 years was considered for the CHP unit (Whiting and Azapagic, 2014). All methane volumetric flows are reported in standard temperature and pressure conditions (0 °C and 1 atm).

The electricity produced through cogeneration was considered to be sold at a price of 0.1149 € kWh^{-1} (Eurostat, 2019). However, the electricity price is very variable and can range between 0.06 and 0.18 € kWh^{-1} depending on the country or region (Eurostat, 2019). In Section 3.3.2, the impact of electricity price on process profitability was analysed through a sensitivity analysis.

2.5. Economic evaluation

Capital expenditures (CAPEX), operating expenditures (OPEX) and revenue were calculated to evaluate the economic feasibility of each scenario. The CAPEX was annualised by using Eq. (5), while the net cost was calculated as the difference between gross cost and revenue (Eq. (6)) (Bolzonella et al., 2018; Vinardell et al., 2021).

$$\text{Annualised CAPEX}\left(€\ y^{-1}\right) = \frac{i\cdot(1+i)^t}{(1+i)^t - 1}\cdot \text{CAPEX} \tag{5}$$

$$\text{Net cost}\left(€\ y^{-1}\right) = \frac{i\cdot(1+i)^t}{(1+i)^t - 1}\cdot \text{CAPEX} + \text{OPEX} - \text{R} \tag{6}$$

where CAPEX is the capital expenditures (€), R is the revenue (€ y^{-1}), OPEX is the operating expenditures (€ y^{-1}), i is the discount rate (5%) and t is the project lifetime (20 years). The electricity revenue from the sidestream AD was included in all sections since this is the main revenue obtained in all scenarios. The revenue from selling struvite was only considered in Section 3.3.3.

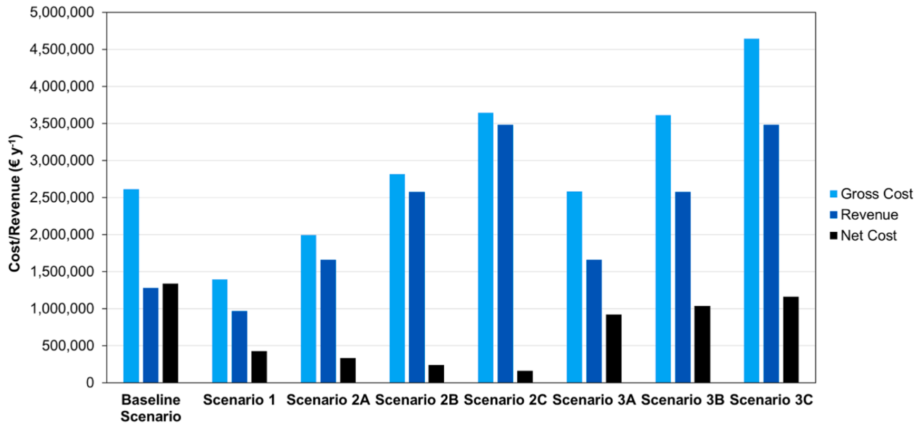

Fig. 2. Cost, revenue and net cost for the different scenarios under study.

3. Results and discussion

3.1. Economic feasibility of co-digesting sewage sludge and food waste in an AnMBR-WWTP

Table 1 shows a summary of the main operation data for each scenario, while Fig. 2 illustrates the gross cost, revenue and net cost for each scenario. The gross cost (light blue bar in Fig. 2) includes the capital and operating costs. The gross cost is mainly driven by the operating cost since the capital cost has a relatively low influence on gross cost (7–23%) because retrofitting the existing sludge line infrastructure allows reducing the initial investment in comparison with the construction of a new infrastructure. The Baseline Scenario is the worst scenario since it presents the highest net cost (1,336,000 € y^{-1}). This is mainly caused by the large amount of secondary sludge produced, which is characterised by its poor biodegradability (~37%) and methane yield (~200 mL CH_4 g^{-1} VS). Scenario 1 results show that retrofitting an aerobic-WWTP to an AnMBR-WWTP would reduce the net cost of the sludge line, primarily due to the lower secondary sludge production. The implementation of AcoD in the AnMBR-WWTP (Scenario 2 and 3) significantly increases the revenue from electricity production (dark blue bar in Fig. 2), which has a direct impact on the net cost (black bar in

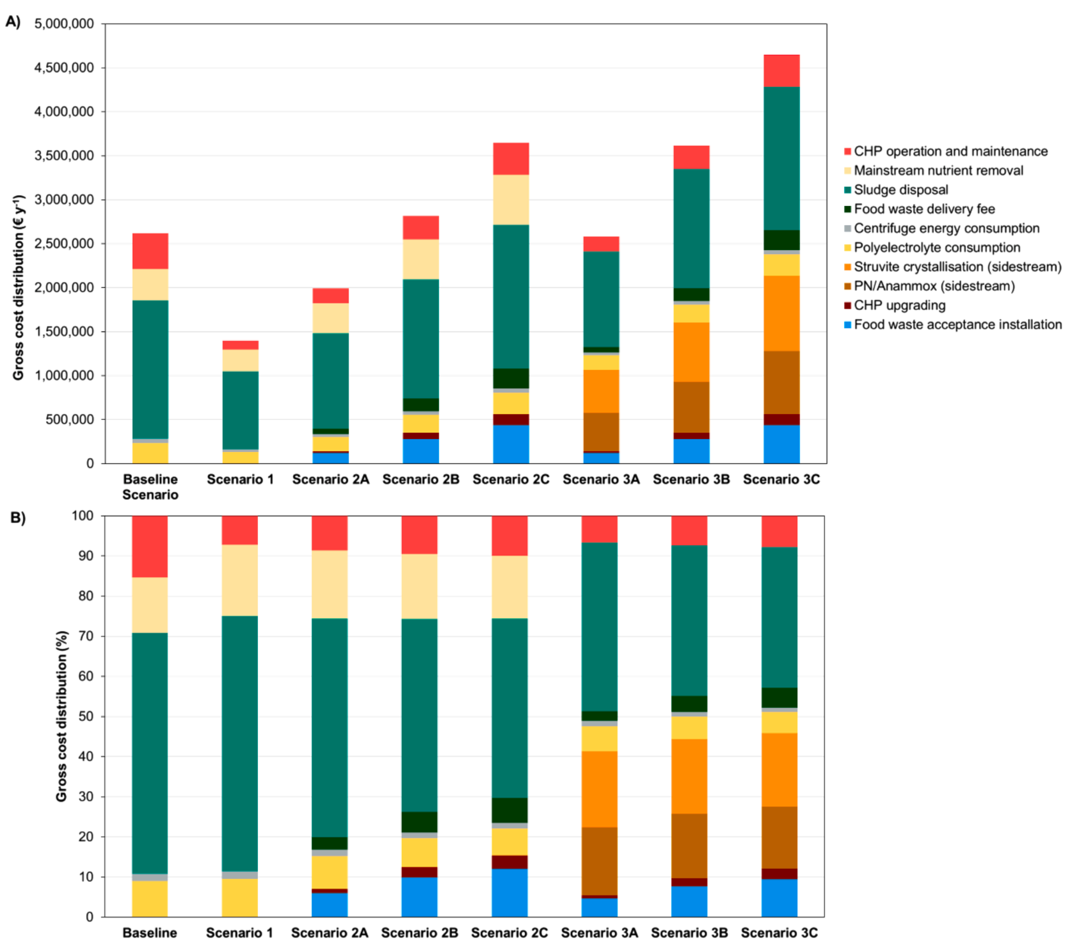

Fig. 3. Gross cost contribution for the different scenarios under study. (A) Absolute gross cost distribution (€ y^{-1}); (B) relative gross cost distribution (%).

Fig. 2). The electricity revenue from food waste co-digestion exceeds the underlying costs associated with food waste acceptance infrastructure and biosolids management/disposal. However, the higher electricity revenue did not offset the additional costs when the nutrients backload was treated in the sidestream (Scenario 3) rather than in the mainstream (Scenario 2).

Scenario 2, where food waste is co-digested with sewage sludge and the nutrients backload is treated in the mainstream, features the lowest net cost among the different scenarios. Scenario 2C is the most competitive alternative in Scenario 2 as a result of the higher biogas production (due to the higher OLR) in the sidestream AD. Specifically, the net cost decreased from 333,000 to 160,000 € y^{-1} as the OLR increased from 1.0 to 2.0 kgVS m^{-3} d^{-1}, respectively (Fig. 2). These results clearly show that the increased electricity production at higher OLRs improves the economic balance of Scenario 2. However, increasing the OLR of the sidestream AD would not always imply a better economic prospect since high OLRs could compromise: (i) the performance and stability of the AD due to overloading, (ii) the quality and stability of the biosolids and (iii) the capacity of the mainstream units to handle the nutrients backload (Mata-Alvarez et al., 2014; Usack et al., 2018; Xie et al., 2018). Therefore, a compromise solution considering the electricity revenue and the technical and environmental prospects of the WWTP is needed to maximise the profit from AcoD.

Scenario 3, where food waste is co-digested with sewage sludge and the nutrients backload is treated in the sidestream, features a net cost higher than Scenario 2. Unlike Scenario 2, the economic balance of Scenario 3 worsens as the OLR increases. Specifically, the net cost increased from 922,000 to 1,162,000 € y^{-1} as the OLR increased from 1 to 2 kg VS m^{-3} d^{-1}, respectively. In Scenario 3, the higher revenue from electricity production from food waste co-digestion does not offset the higher gross cost as the OLR increases. The higher gross cost of Scenario 3 is attributed to the implementation of PN/Anammox and struvite crystallisation for the removal of N and P from the centrate. The addition of food waste increases the content of N and P in the centrate, which has a direct impact on the capital and operating costs of both processes. It was estimated that the NH_4^+-N and PO_4^{3-}-P backload increased from 1,037 to 1,702 kg N d^{-1} and from 222 to 378 kg P d^{-1} as the OLR increased from 1 to 2 kg VS m^{-3} d^{-1}, respectively. These results suggest that treating the N and P in the sidestream is costlier than treating these compounds in the mainstream of the WWTP. However, the treatment of the extra N and P backload in the mainstream of the WWTP could make necessary to expand existing facilities with a direct impact on capital costs (out of the scope of the present study). Additionally, revenue from struvite crystallisation would have a noticeable influence on the economic balance of Scenario 3 as discussed in Section 3.3.3. Besides economic considerations, implementing N and P removal technologies in the sidestream of the WWTP (i) reduces disturbances in the mainstream biological nitrogen removal step (Sembera et al., 2019), (ii) prevents piping blockage because of uncontrolled and spontaneous struvite precipitation (Bouzas et al., 2019) and (iii) reduces the environmental impacts related to eutrophication if the mainstream does not have the spare capacity to handle the extra nutrients load (Rodriguez-Garcia et al., 2014).

3.2. Cost distribution for the different scenarios

Fig. 3 shows the gross cost distribution for the different scenarios under study. Biosolids disposal (including transport) is the most important cost contributor in all the scenarios. In absolute values, the biosolids disposal cost increases as the OLR increases due to the higher biosolids production at higher OLRs (Fig. 3A). However, in relative values, the disposal contribution to the gross cost decreases as the OLR increases in AcoD scenarios due to the presence of other cost contributors in the AcoD scenarios (Fig. 3B). It is worth mentioning that biosolids disposal cost does not increase linearly with the OLR since food waste digestion produces a relatively low amount of biosolids as a result of its high biodegradability (~85%).

Food waste AcoD implies the construction of a new facility for food waste acceptance. The capital cost to construct a food waste acceptance facility ranges between 4 and 12% in AcoD scenarios (Fig. 3B). The

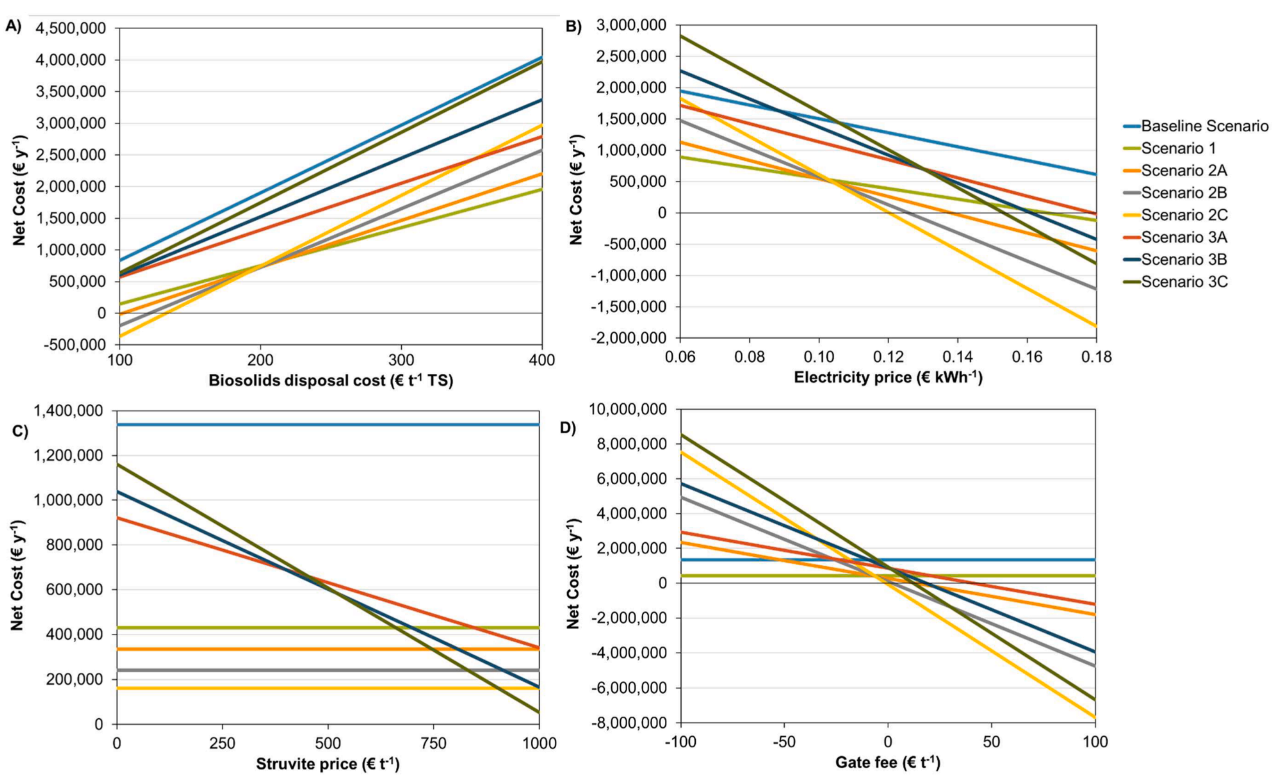

Fig. 4. Sensitivity analysis for: (A) biosolids disposal cost, (B) electricity price, (C) struvite price, and (D) gate fee.

contribution of the food waste acceptance facility increases as the OLR increases since larger amounts of food waste are needed at higher OLRs. Food waste AcoD also implies the negotiation of a gate/delivery fee for food waste acceptance. For a delivery fee of 3 € t^{-1}, the cost contribution of the food waste delivery fee ranges between 2 and 7% of the gross cost. This delivery fee represents the amount of money that the WWTP has to pay to obtain the food waste. However, some WWTPs obtain a revenue for the acceptance of the food waste (gate fee). For instance, Nghiem et al. (2017) reported a gate fee of 86 € t^{-1} for Rovereto WWTP (Italy). The difference between Grüneck and Rovereto could be attributed to the non-processed origin of food waste in the Rovereto WWTP (Sembera et al., 2019). The quality of the food waste received at the WWTP determines the need to implement a food waste processing infrastructure to remove impurities (e.g. glass, debris, metals and plastics) before feeding the food waste into the digester. However, the criteria to establish a gate/delivery fee by the WWTP is still unclear regardless of the food waste origin. Therefore, it is important to evaluate how the gate/delivery fee influences the net cost to better understand the role of this parameter in AcoD economics (see Section 3.3.4).

Treating the nutrients backload in the mainstream represents between 13 and 18% of the gross cost (Fig. 3B). This contribution is much lower than when the nutrients backload is treated in the sidestream of the WWTP, where the contribution represents 33–36% of the gross cost. These results support the idea that PN/Anammox and struvite crystallisation sidestream implementation is potentially costlier than treating the nutrients backload in the existing mainstream facility. However, in those WWTPs operating close to their design capacity, the implementation of nutrients backload treatment in the sidestream could be more economical than expanding the existing mainstream processes. In Scenario 3, the gross cost contribution of the sidestream nutrients backload treatment is close to the biosolids disposal contribution, which highlights the impact of sidestream PN/Anammox and struvite crystallisation on AcoD economics. Struvite crystallisation is slightly costlier (18–19%) than PN/Anammox (15–17%) as a result of the high consumption of NaOH needed to adjust the pH to 9 and $MgCl_2$ needed to provide the amount of Mg^{2+} to precipitate struvite (Table 1).

3.3. Sensitivity analysis of the most critical factors for sewage sludge and food waste co-digestion

Fig. 4 shows the sensitivity analysis for the four parameters evaluated: (i) biosolids disposal cost, (ii) electricity price, (iii) struvite price and (iv) food waste gate fee.

3.3.1. Biosolids disposal cost

Fig. 4A shows the net cost variation when the biosolids disposal cost (including transport) ranges between 100 and 400 € t^{-1} TS (Foladori et al., 2015). This interval was chosen since it comprises biosolids disposal costs for other disposal alternatives such as incineration, landfilling or composting (Foladori et al., 2015). Scenario 2C is the most competitive scenario for disposal costs below 200 € t^{-1} TS. However, when disposal cost is above 200 € t^{-1} TS, Scenario 1, 2A and 2B outcompete Scenario 2C due to the lower amount of biosolids to be managed in these scenarios. Interestingly, Scenario 1 is the most competitive scenario when the disposal cost is above 200 € t^{-1} TS. These results suggest that implementing AcoD of sewage sludge and food waste in an AnMBR-WWTP is recommendable for economical management options (<200 € t^{-1} TS) such as agriculture and composting; however, it is less economically attractive for costly management options (>200 € t^{-1} TS) such as landfilling or incineration. Scenario 3 is costlier than Scenario 1 and 2 regardless of the biosolids disposal cost as a result of the extra cost needed to implement PN/Anammox and struvite crystallisation in the sidestream of the WWTP. The Baseline Scenario presents the worst economic prospect since it produces a large amount of poorly biodegradable secondary sludge.

3.3.2. Electricity price

Fig. 4B shows the net cost variation when the electricity purchase price ranges between 0.06 and 0.18 € kWh^{-1}. This interval was chosen since it represents the current electricity prices in the European Union (Eurostat, 2019). Scenario 2C is the most favourable scenario when the electricity price is above 0.10 € kWh^{-1} due to the high amount of energy produced when the sidestream AD co-digests sewage sludge and food waste at an OLR of 2 kg VS m^{-3} d^{-1}. All AcoD scenarios feature a net benefit (net cost < 0 € y^{-1}) at an electricity price of 0.18 € kWh^{-1}. Conversely, Scenario 1 is the most favourable scenario when the electricity price is below 0.10 € kWh^{-1}. At low electricity prices, the higher electricity revenue of AcoD scenarios is not enough to offset their higher gross costs. These results show that implementing AcoD is particularly attractive when the price of electricity is high since a negative impact on AcoD profitability is observed as the price of the electricity decreases. Another example of this factor is that Scenario 3C and 3B are the less favourable scenarios at an electricity price of 0.10 and 0.09 € kWh^{-1}, respectively. Overall, these results show that the application of AcoD in the AnMBR-WWTP is particularly attractive when the electricity price is above 0.10 € kWh^{-1}, where the higher amount of energy recovered and sold compensates the higher gross costs needed to implement AcoD. This is relevant considering that energy prices are expected to increase in the future as a result of the increased fuel and CO_2 prices (Panos and Densing, 2019). Future higher energy prices would make AcoD more attractive to WWTP operators to maximise the revenue from electricity generation.

3.3.3. Struvite price

Fig. 4C shows the net cost variation of the different scenarios when the struvite price ranges between 0 and 1,000 € t^{-1}, which comprises struvite prices reported in literature (Akyol et al., 2020; Molinos-Senante et al., 2011). It should be noted that the impact of struvite price is only applicable in Scenario 3 because this is the only scenario that included struvite precipitation in the sidestream of the WWTP (Fig. 1). The net cost of Scenario 3 decreases from 921,000–1,161,000 to 54,000–340,000 € y^{-1} as the struvite price increases from 0 to 1,000 € t^{-1}, respectively. Scenario 3C is more competitive than Scenario 3A and 3B when the struvite price is above 450 € t^{-1} since Scenario 3C achieves the highest revenue from struvite commercialisation. This is mainly attributed to the higher amount of phosphorus released in Scenario 3C in comparison with Scenario 3A and Scenario 3B. Scenario 3C becomes the most favourable scenario when the struvite price is above 850 € t^{-1} (Fig. 4C). However, a struvite price of 850 € t^{-1} is little realistic since struvite prices between 188 and 763 € t^{-1} have been reported in literature (Akyol et al., 2020; Molinos-Senante et al., 2011). Besides economic considerations, novel legislations forcing the recovery of phosphorus can be a major driver for the implementation of struvite recovery in WWTPs.

3.3.4. Food waste gate fee

Fig. 4D shows the net cost variation when the food waste gate fee ranges between –100 and 100 € t^{-1}, which is a representative interval for full-scale plants using food waste as co-substrate (Nghiem et al., 2017). The gate fee has a big impact on net cost since Scenario 2 and 3 achieve net benefit (net cost < 0 € y^{-1}) when the gate fee is above 43 € t^{-1}. Scenario 2C and 3C are the most favourable scenarios at gate fees above 30 € t^{-1} since these scenarios accept large amounts of food waste (OLR of 2.0 kg VS m^{-3} d^{-1}), which leads to the highest revenue from the gate fee. For a gate fee of 86 € t^{-1}, the gate fee represents 51, 61 and 65% of the total revenue in Scenario 3A, 3B and 3C, respectively. This agrees with Sembera et al. (2019), who reported that the gate fee could represent an important revenue for WWTPs. However, the increasing interest in AcoD would also increase the number of WWTPs performing AcoD, which would increase the competition for co-substrates, implying a drop in gate fee prices that could even become negative (delivery fee). As shown in Fig. 4D, paying a delivery fee for the food waste would have a negative impact on net cost. Indeed, Scenario 1 is the most favourable scenario when the delivery fee is above 10 € t^{-1} (gate fee < -10 € t^{-1}) since this scenario does not implement AcoD and does not have to pay for the food waste. Thus, the implementation of AcoD in an AnMBR-WWTP would only be economically attractive when the food waste delivery fee is below 10 € t^{-1} (gate fee > -10 € t^{-1}). These results reinforce the idea that the gate/delivery fee is a key factor in AcoD economics.

4. Conclusions

The economic feasibility of implementing sewage sludge and food waste co-digestion in the sidestream AD of an AnMBR-WWTP was evaluated. Results showed that the higher electricity revenue derived from co-digestion offsets the higher costs associated with the food waste acceptance infrastructure and biosolids management/disposal. However, the electricity revenue did not offset the additional costs when the nutrients backload was treated using sidestream equipment (partial-nitritation/anammox, struvite crystallisation). Biosolids disposal was the most important gross cost contributor in all scenarios. Finally, a sensitivity analysis revealed that the food waste gate fee had a noticeable impact on the net cost.

CRediT authorship contribution statement

Sergi Vinardell: Conceptualization, Formal analysis, Investigation, Data curation, Methodology, Visualization, Writing - original draft, Writing - review & editing. **Sergi Astals:** Conceptualization, Methodology, Supervision, Writing - review & editing. **Konrad Koch:** Supervision, Writing - review & editing. **Joan Mata-Alvarez:** Supervision, Writing - review & editing, Funding acquisition. **Joan Dosta:** Supervision, Writing - review & editing.

Declaration of Competing Interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biortech.2021.124978.

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