Journal of Water Process Engineering 58 (2024) 104847

Contents lists available at ScienceDirect

Journal of Water Process Engineering

journal homepage: www.elsevier.com/locate/jwpe

Reducing wasted sewage sludge from wastewater treatment: Comparison of two membrane bioreactor pilot plants

Paulo Marcelo Bosco Mofatto, Alida Cosenza, Daniele Di Trapani, Giorgio Mannina *

Engineering Department, Palermo University, Viale delle Scienze, Ed. 8, 90128 Palermo, Italy

A R T I C L E I N F O

Keywords:
Waste sludge
Sludge minimisation
Oxic-settling-anaerobic (OSA) optimisation
Plant configuration

A B S T R A C T

This study compares two Oxic Settling Anaerobic (OSA) – Membrane Bioreactor (MBR) configurations fed with real wastewater.

The novelty of the study is to consider an MBR configuration (namely, MBR-OSA II) never investigated with the OSA process. The influence of the hydraulic retention time (HRT) inside the anaerobic reactor for each configuration (namely, MBR – OSA I and MBR – OSA II) was investigated. Four experimental periods were carried out: Period I and II for MBR – OSA I, under HRT of 4 and 6 h, respectively, and Period III and IV for MBR – OSA II, under HRT of 4 and 6 h, respectively. Results showed the decay rate b_H increased with the HRT, especially during Periods I and II (from 0.62 d^{-1} to 0.77 d^{-1}). The anaerobic HRT influenced the total sewage sludge production. With the increase of the HRT for the same configuration, the sludge production decreased. In terms of biological sludge production, a substantial reduction occurred under MBR – OSA II operation compared to MBR – OSA I. However, despite MBR-OSA II configuration providing the highest reduction of the biological sludge production (78 %) (during Period IV), during this Period, the highest N_2O (N_2O emission factor till to 3.1 %) production and membrane fouling occurred. Indeed, Period IV showed the highest membrane resistance (namely, 25 · 10^{-13} m^{-1}). Thus, the MBR-OSA I with anaerobic HRT equal to 6 h provided the best results.

1. Introduction

The management and disposal of excess sludge in wastewater treatment plants (WWTPs) represent one of the main operational costs in the plant operation [1,2]. This issue is exacerbated by the increased waste sludge produced due to the stringent effluent limits and the increased treated wastewater flow rate (caused by population growth) [3]. With this regard, several efforts have been made in the literature to find a technological solution to reduce the amount of waste sludge to be disposed of. Several technologies (chemical, biological and thermal) and operational conditions (e.g., hydraulic retention time or internal recirculation ratio) have been tested in the literature [3,4]. Among the biological processes, one of the most promising processes for reducing waste sludge is the oxic settling anaerobic (OSA) [5]. This process is applied in the water line and inserts an anaerobic side-stream reactor (ASSR) between the recirculation-activated sludge line and the biological reactors [5]. The OSA process has several advantages compared to the chemical/thermal processes since it does not require further operational costs due to chemical or energy consumption [6]. Furthermore, since the OSA process is applied to the water line, it reduces the amount of sludge to be treated in the sludge line, thus favouring the operational cost reduction for the sludge line management [7]. Several studies have demonstrated in literature the efficiency of the OSA process in reducing waste sludge production by testing several operational conditions of the ASSR and recirculation ratios [8–10]. Vitanza et al. [10] tested the OSA process by varying the plant operational conditions, obtaining a reduction of the excess till to 49.6 ± 20.7 % compared to the conventional activated sludge system (CAS) without ASSR. Ferrentino et al. [8] found up to 69 % sludge reduction by applying the OSA process in a full-scale WWTP. The advantages of the OSA process have been coupled with that of membrane bioreactors (MBR) by Fida et al. [9], who obtained up to 58 % of sludge reduction.

The literature study has mainly focused on the operational conditions of ASSR without considering the possibility of modifying the configuration, especially in an MBR system. Therefore, this study tests the possibility of varying the recirculation from the ASSR pipeline. As far as authors know, no similar studies have been performed using an MBR system. More precisely, this study tested a pre-denitrification MBR pilot

* Corresponding author.
E-mail address: Giorgio.mannina@unipa.it (G. Mannina).

https://doi.org/10.1016/j.jwpe.2024.104847
Received 3 August 2023; Received in revised form 13 January 2024; Accepted 14 January 2024
Available online 25 January 2024

plant fed with real wastewater under two plant configurations (namely, MBR-OSA I and MBR-OSA II). The MBR-OSA II configuration has never been tested in the literature. It has the novelty of recirculating the sludge from the ASSR to an aerobic reactor instead of an anoxic one. The influence of hydraulic retention time (HRT) inside the ASSR was investigated in view of reducing waste sludge production by maintaining high effluent quality and reducing greenhouse gas emissions.

2. Material and methods

2.1. Pilot plant configuration

Two different plant configurations (MBR – OSA I and MBR – OSA II) were set up at the Water Resource Recovery Facility of Palermo University [11,12]. Both configurations aim at carbon and nitrogen removal by implementing a pre-denitrification scheme. The pilot plants were fed with real wastewater from the dormitory and canteen inside the Palermo University Campus [11]. More specifically, the MBR-OSA I configuration was composed of an anoxic reactor followed by an aerobic reactor according to the pre-denitrification scheme (Fig. 1a). The solid-liquid separation occurred thanks to a membrane bioreactor (MBR) composed of an ultrafiltration hollow fibre membrane (porosity 0.03 μm) (Fig. 1a). Permeate was collected in a clean-in-place (CIP). Activated sludge (RAS) was recirculated from MBR to the anoxic reactor with an interposed oxygen depletion reactor (ODR) to reduce the recirculated oxygen mass. According to the OSA process, an anaerobic side-stream reactor (ASSR) was inserted in the RAS line; a portion of RAS was first destined for the ASSR. The MBR-OSA II configuration (Fig. 1b) differed from the MBR-OSA I configuration for the portion of RAS from ASSR. More precisely, in the MBR-OSA II configuration, this portion is recycled into the aerobic reactor instead of the anoxic reactor (Fig. 1b).

Table 1 summarises the average and standard deviation (SD) of the influent characteristics of the two pilot plants. Qualitative features reported in Table 1 have been measured according to Standard Methods [13]. The highest contribution to the influent wastewater variability is mainly due to the solids. Indeed, the TCOD values are mostly variable compared to the soluble compounds.

2.2. The experimental campaign and analytical methods

During the experimental campaign, the effect of HRT (4 and 6 h) inside the ASSR was tested for each configuration. More precisely, four experimental periods were considered: Period I, Period II, Period III, and Period IV. Periods I and II refer to MBR-OSA —I configuration and were characterised by HRT inside the ASSR of 4 and 6 h, respectively. In contrast, Periods III and IV refer to MBR-OSA -II configuration and were characterised by HRT inside the ASSR of 4 and 6 h, respectively.

During the experimental periods, dissolved oxygen (DO), pH and oxidation reduction potential (ORP), were monitored using a multimeter (WTW 3340). Chemical oxygen demand (COD), ammonia nitrogen

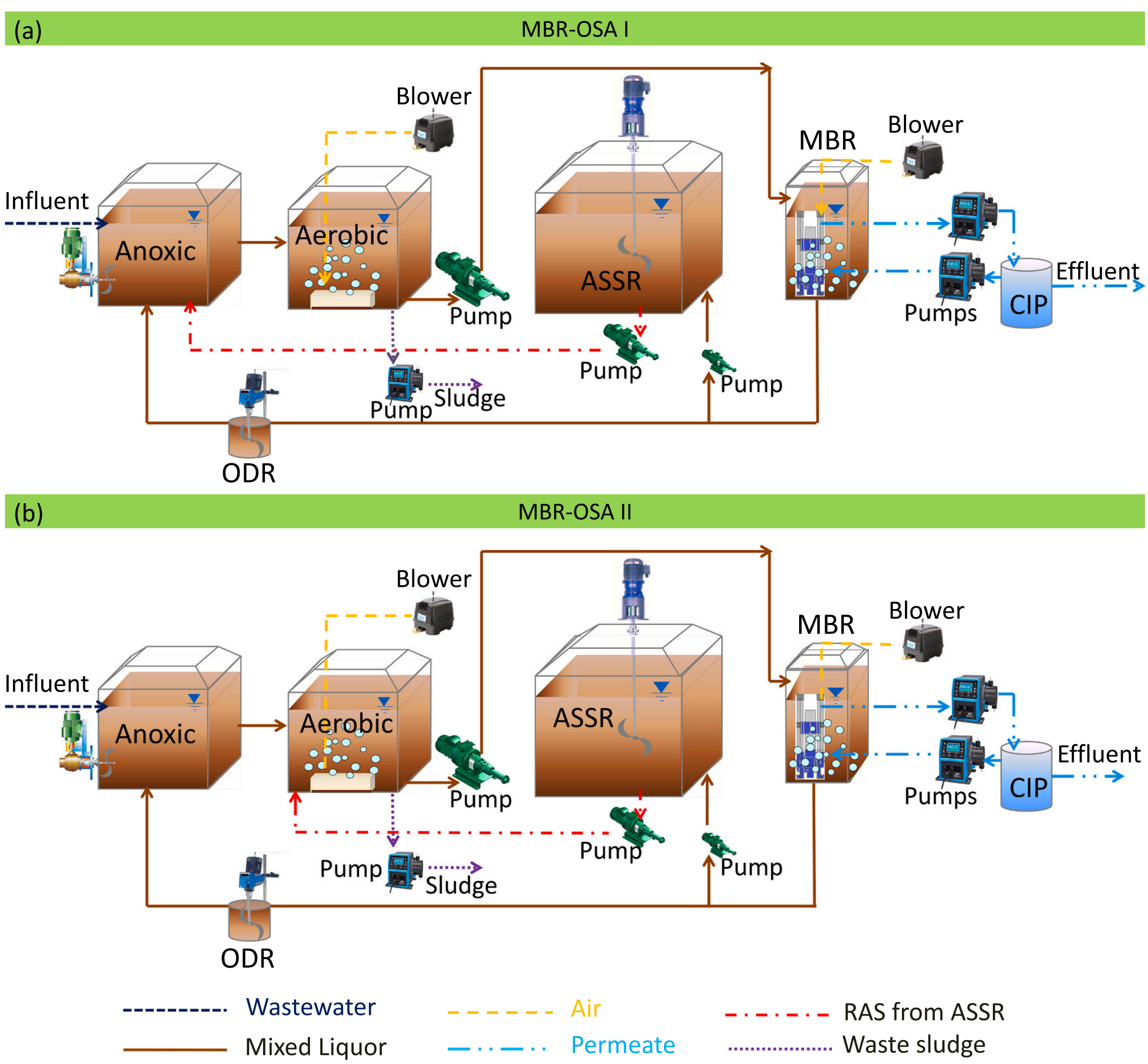

Fig. 1. Schematic layout of MBR-OSA I (a) and MBR-OSA II (b).

Table 1
Influent characteristics; SD = Standard Deviation; measure performed according to Standard Methods [13].

Compounds	Symbol	Units	Period I MBR-OSA I HRT = 4 h Average	SD	Period II MBR-OSA I HRT = 6 h Average	SD	Period III MBR-OSA II HRT = 4 h Average	SD	Period IV MBR-OSA II HRT = 6 h Average	SD
Total COD	TCOD	[mg L^{-1}]	1352	175	1152	207	2082	707	2354	688
Soluble COD	sCOD	[mg L^{-1}]	213	74	142	46	209	36.5	367	129
Biochemical oxygen demand	BOD	[mg L^{-1}]	305	95	314	100	229	8.5	373	135
Total nitrogen	TN	[mg L^{-1}]	38	6	29	3	44.6	5.3	45.9	8
Ammonium	NH_4-N	[mg L^{-1}]	32	4	27	3	39.6	5.5	35.8	4.4
Phosphate	PO_4-P	[mg L^{-1}]	14	3	9	3	21.1	12.5	23.9	8.6
Flow rate	Q_{IN}	[L h^{-1}]	17.5	2.5	17.9	0	18.2	0	18.2	0
Food/Microorganism ratio	F/M	[gCOD/gTSS d]	0.4	0.18	0.4	0.06	0.18	0.04	0.36	0.1
Sludge retention time	SRT	[d]	41	24	53	18	20	2	45	12

(NH_4^+-N), nitrate (NO_3^--N), nitrite (NO_2^--N), orthophosphate (PO_4^{-3}-P), total suspended solid (TSS) and volatile suspended solid (VSS) concentrations, biological oxygen demand (BOD) and Total Nitrogen (TN) were measured according to Standard Methods [13], twice a week. Respirometric tests were performed once per week to analyse the kinetic parameters according to the literature [23]. The sludge volume index (SVI) was evaluated in the aerobic mixed liquor once per week. Extrapolymeric substances (EPS) and soluble microbial substances (SMP) (carbohydrates and proteins) have been analysed one time per week in the mixed liquor contained inside the anoxic, aerobic, OSA and MBR tanks as suggested in the literature [14]. Nitrous oxide (N_2O—N) concentration in the liquid and gaseous phase of all the reactors has been measured two times per week by using a gas chromatograph equipped with an Electron Capture Detector (ECD) according to the literature [15]. Moreover, the N_2O–N fluxes have also been quantified using each reactor's off-gas flow rate measurement.

2.3. Excess sludge production quantification

In view of assessing sludge reduction, the primary (PSludge) and total sludge (TSludge) produced every day were calculated according to Mannina et al. [16]. Biological sludge was calculated as the difference between TSludge and PSludge. Furthermore, the observed biomass yield (Y_{obs}) was calculated based on the mass of TSS produced per mass of removed COD, according to Mannina et al. [16].

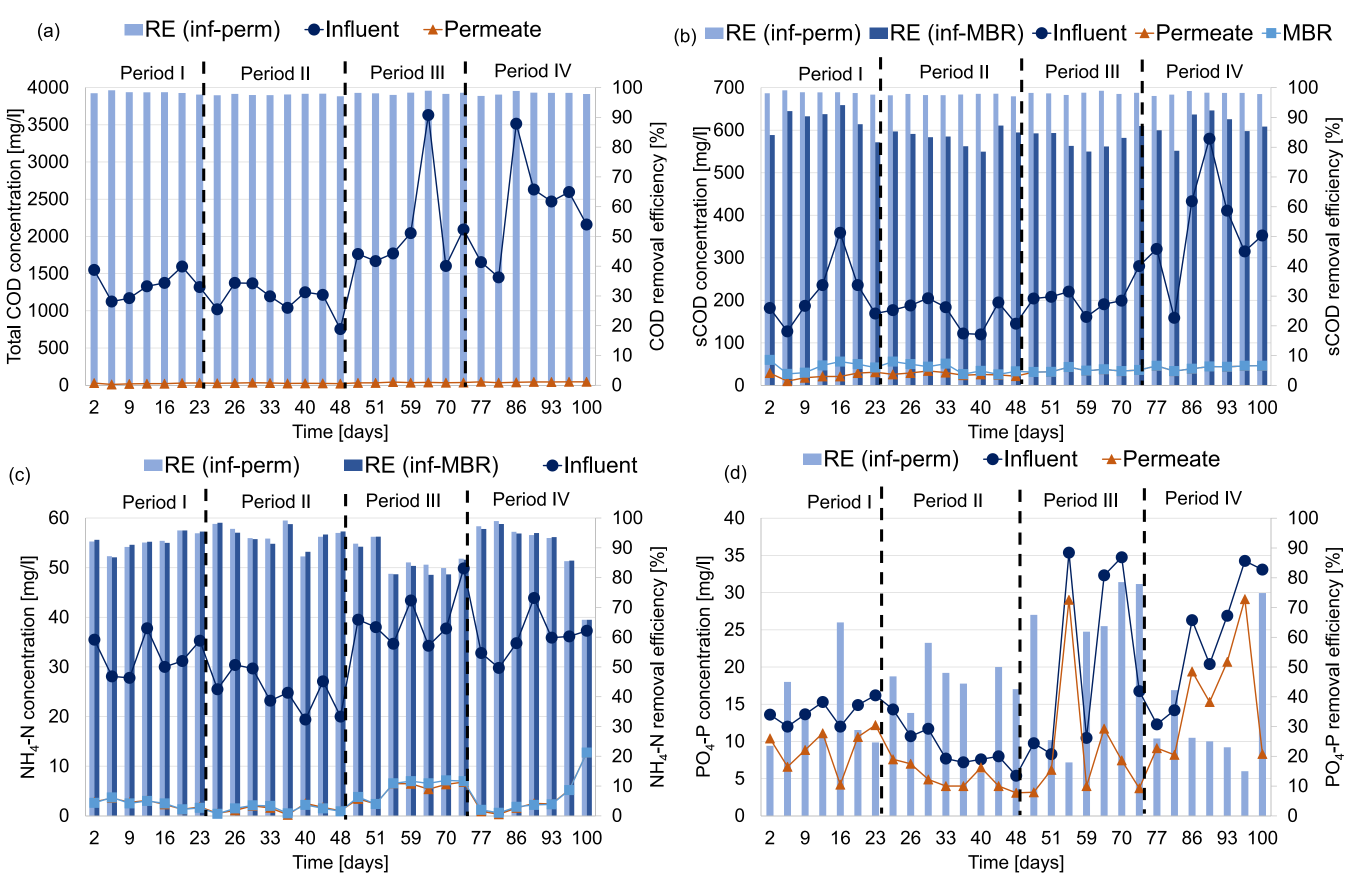

Fig. 2. The pattern of influent, effluent and removal efficiency (RE) for total COD (a), soluble COD (b) NH_4-N (c) and PO_4-P (d) for the different experimental periods; RE (inf-perm) represents the removal efficiency calculated by considering the influent and permeate concentration, RE (inf-MBR) is the removal efficiency calculated between influent and MBR supernatant.

2.4. Membrane fouling

In view of monitoring membrane fouling, the total resistance (R_T) was quantified according to Eq. (1).

$$R_T = \frac{\Delta P}{\mu \cdot J}\ (m^{-1}) \tag{1}$$

where ΔP [kPa] is the transmembrane pressure (TMP) variation and μ [Pa.s] is the permeate viscosity, and J ($m^3\ m^{-2}\ s^{-1}$) is the permeate flux. The total resistance was fractionated into clean membrane resistance (R_m), resistance due to the cake layer deposited on the membrane surface (R_C) and pore fouling (R_P) according to the resistance in series model (RIS) as proposed by Di Bella et al. [24] (Eq. (2)).

$$R_T = R_m + R_C + R_P\ (m^{-1}) \tag{2}$$

The RIS model was applied after the physical cleaning of the membrane [24].

3. Results and discussion

3.1. Pilot plant performance

Fig. 2 reports the main pollutants' influent, effluent, and removal efficiency (RE) trends for the different experimental periods.

Significant fluctuations were observed during experiments concerning COD removal (total and soluble), with a general increase of COD concentrations in Period III and IV (Fig. 2a-b). This result could be attributed to the fact that the characteristics of the real wastewater fed to the pilot plant were highly affected by the dormitory and canteen habits. Nevertheless, despite these variations, the overall removal efficiencies for total and soluble COD were excellent throughout experiments, with observed values always higher than 95 % (Fig. 2a-b). Referring to soluble COD biological removal, evaluated as the difference between COD concentrations in the influent and the supernatant of MBR tank (before membrane filtration), the observed data highlighted an excellent performance during experiments, with efficiencies higher than 80 %. The results highlighted that COD removal was not affected by plant configuration. Concerning nitrification, the plant performances were generally quite good during experiments (Fig. 2c), highlighting that the high SRT values, typical of MBRs, allow the growth of nitrifiers. In Period I and II, very high nitrification was observed, with average values higher than 90 % and no significant impact related to the ASSR reactor was observed, thus confirming previous studies [17–19]. When plant configuration was changed, a slight decrease in nitrification was observed, which contrasted with previous results [16], highlighting that the reduction of biomass exposure under a not aerated period was beneficial for nitrifying growth. Nevertheless, the decrease in nitrification efficiency observed in Period III in the present study could be related to the significant increase of influent ammonia concentrations that could have negatively affected nitrifying bacteria. However, after biomass acclimation, in Period IV, a significant restoration of nitrification with efficiencies higher than 90 % was observed. This latter result confirmed that the decrease in aerobic exposure is generally beneficial for the growth of nitrifiers.

Concerning PO_4-P removal (Fig. 2d), a good performance was observed. Implementing the anaerobic reactor in the RAS line might improve the selection of phosphate accumulating organisms (PAOs) or denitrifying phosphate accumulating organisms (DPAOs) due to the alternation between aerobic and anaerobic environments. Nevertheless, the significant fluctuations of influent PO_4-P concentrations did not allow to assess an evident influence of system configurations and ASSR HRT on P removal.

3.2. Sludge reduction and sludge properties

Concerning sludge production (Fig. 3), the data achieved highlighted that different behaviours were obtained throughout experiments related to configurations and wastewater characteristics. In general, the sludge production was lower under configuration MBR – OSA I (Periods I and II) compared to configuration MBR – OSA II (Periods III and IV), suggesting that prolonged exposure to a not-aerobic environment might have a more substantial effect in terms of sludge reduction. Nevertheless, the significant COD increase observed in Periods III and IV could have contributed to the increased sludge production. For the same configuration, the increase of HRT in the anaerobic reactor was enhanced to improve the sludge minimisation further. Indeed, the total sludge production during MBR-OSA I configuration was equal to 2260 g and 750 g during Period I and Period II, respectively. In contrast, the total sludge production during MBR-OSA II configuration was equal to 5204 g and 1764 g during Period III and Period IV, respectively.

Table 2 summarises the average values of the observed heterotrophic yield growth Y_{obs}, the temperature (T), the SVI and the percentage of biological sludge reduction. This latter was calculated by comparing the sludge production between periods with the same HRT for the two configurations (Period I versus Period III, and Period II versus Period IV). Considering the biological sludge production one can focus the attention on the Y_{obs}. The lowest Y_{obs} value was obtained for Period I and Period IV (0.21 gTSS/gCOD for both). This result partially corroborates the above discussion on the total amount of sludge production since we refer only to the biological production of sludge in this case.

In terms of sludge settleability, no influence was revealed in plant configuration or HRT. Indeed, the average measured SVI values are always lower than 150 mL·gTSS^{-1}. SVI values lower than 100 mL·gTSS^{-1} and not above 150 mL gTSS^{-1} indicate good sludge settling conditions [20]. Considering the biological sludge reduction, 45 % of the reduction occurred in Period III compared to Period I; while 78 % occurred in Period IV compared to Period II. This result is also attributed to the low temperature registered during Periods III and IV, which limited the biomass growth. Biological sludge reduction seems to be in contrast with the discussion of Fig. 3; however, one has to be precise that data reported in Fig. 3 refers to the total sludge production (not only biological) and includes the primary sludge.

In Fig. 4 the average percentages of SMPp, SMPc, EPSp and EPSc sludge concentration inside the plant for each Period are reported. During Periods I and II the total EPS (sum between EPSp and EPSc) percentage was 96.96 % (Fig. 4a) and 96.91 % (Fig. 4b), respectively. Thus, the HRT inside the ASSR does not influence the floc structure in MBR – OSA I configuration. Whereas, during Periods III and IV the total EPS percentage was 90.24 % (Fig. 4c) and 87.12 % (Fig. 4d),

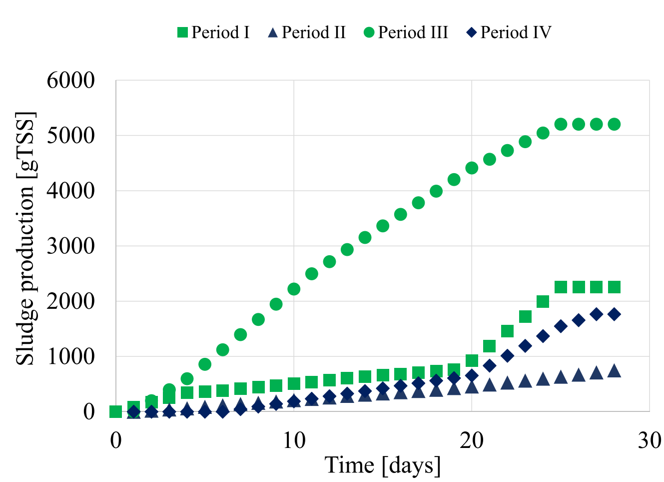

Fig. 3. Cumulative sludge production during experiments.

Table 2
Average values of observed heterotrophic yield growth (Y_{obs}), temperature (T), Sludge volume index (SVI) and percentage of biological sludge reduction.

Period	I	II	III	IV
Y_{obs} [gTSS/gCOD]	0.21	0.24	0.32	0.21
T [°C]	27.9	28.6	14	18.5
SVI [mL $gTSS^{-1}$]	110	150	127	90
Biological sludge reduction [%]	–	–	45	78

respectively. This suggests that the change of layout configuration from MBR – OSA I to MBR – OSA II caused the floc sludge destructuration with a reduction of the EPS amount and the increase of total SMP (sum between SMPp and SMPc). Indeed, an increase in total SMP percentage occurred from configuration MBR – OSA I to MBR – OSA II both comparing Period I (total SMP percentage of 3.04 %) to Period III (total SMP percentage of 9.76 %) and Period II (total SMP percentage of 3.09 %) to Period IV (total SMP percentage of 12.88 %) (Fig. 4). This result contrasted with previous results [16], highlighting a floc destructuration in a configuration where RAS was directed to the anoxic reactor, not the aerobic reactor. However, the different results obtained here are mainly due to the presence of an MBR, which modifies the biomass nature and consortium.

3.3. Heterotrophic and autotrophic biomass kinetics

Table 3 summarises the average values of kinetic and stoichiometric parameters for both heterotrophic and autotrophic species during experiments.

Concerning heterotrophic species, when the MBR – OSA I configuration was implemented, a slight decrease of Y_H was observed from

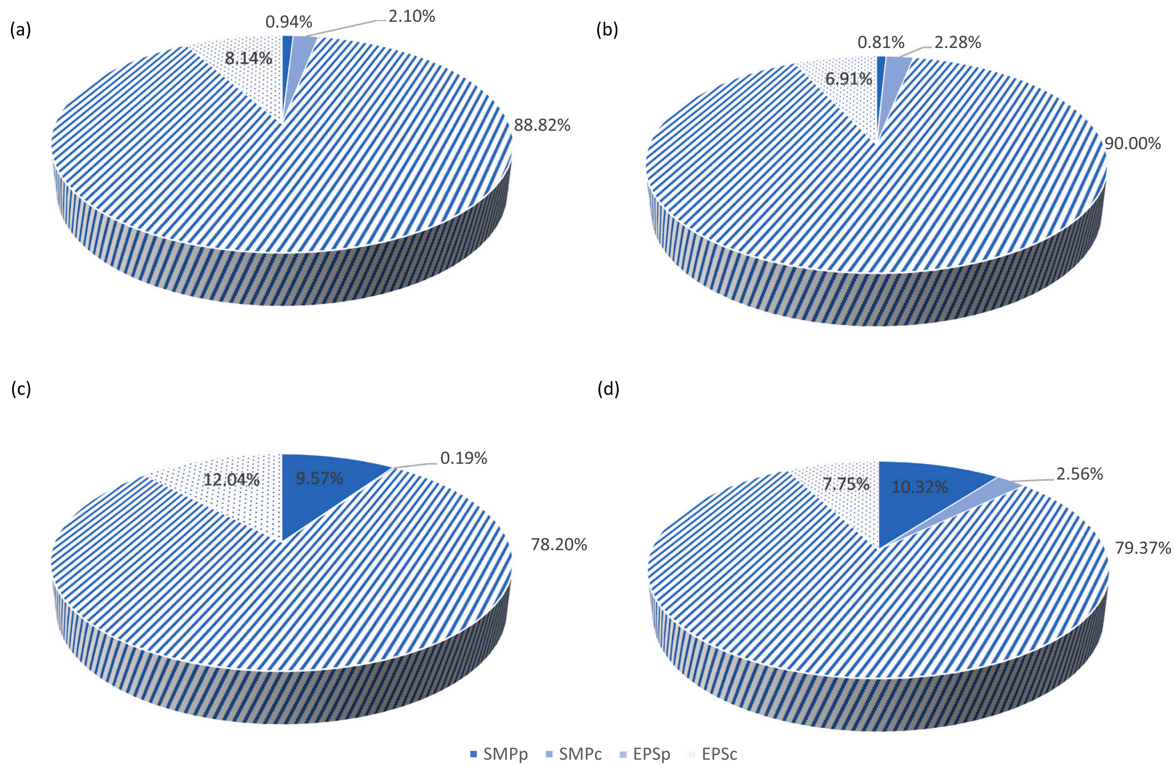

Fig. 4. The average percentage of SMPp, SMPc, EPSp and EPSc sludge concentration during Period I (a), Period II (b), Period III (c), and Period IV (d).

Table 3
Summary of the average values of the main heterotrophic kinetic and stoichiometric parameters (in brackets the standard deviation values).

Configuration			MBR - OSA I		MBR - OSA II	
HRT in ASRR [h]			4	6	4	6
Parameter	Symbol	Units	Period I	Period II	Period III	Period IV
			Heterotrophic			
Max. growth yield	Y_H	[gVSS g^{-1}COD]	0.40 (±0.01)	0.37 (±0.04)	0.44 (±0.03)	0.43 (±0.04)
Decay rate	b_H	[d^{-1}]	0.62 (±0.02)	0.77 (±0.05)	0.41 (±0.01)	0.58 (±0.02)
Max. growth rate	μ_H	[d^{-1}]	2.27 (±0.61)	2.33 (±0.42)	2.44 (±0.73)	2.90 (±0.01)
Max. removal rate	ν_H	[d^{-1}]	4.59 (±1.63)	6.46 (±1.87)	5.56 (±1.17)	6.79 (±0.57)
Net growth rate	μ_H- b_H	[d^{-1}]	1.67 (±0.63)	1.55 (±0.39)	2.03 (±0.44)	2.32 (±0.03)
Active fraction	f_X	[%]	30.51 (±4.96)	21.41 (±7.42)	34.22 (±2.56)	35.29(±0.70)
			Autotrophic			
Max. growth yield	Y_A	[gVSS·$g^{-1}$$NH_4$-N]	0.18 (±0.04)	0.12 (±0.03)	0.25 (±0.01)	0.24 (±0.007)
Decay rate	b_A	[d^{-1}]	0.14 (±0.01)	0.12 (±0.01)	0.10 (±0.02)	0.12 (±0.02)
Max. growth rate	μ_A	[d^{-1}]	0.23 (±0.03)	0.21 (±0.01)	0.26 (±0.04)	0.24 (±0.04)
Max. removal rate	ν_A	[d^{-1}]	1.32 (±0.19)	2.22 (±1.07)	1.03 (±0.08)	1.00 (±0.16)
Nitrification rate	N_R	[mgNH_4·L^{-1} h^{-1}]	3.10 (±1.68)	5.95 (±0.76)	3.76 (±0.96)	3.95 (±0.27)

Period I to Period II, when the HRT in the ASRR was increased from 4 to 6 h, respectively. The decay rate b_H increased from Period I to Period II, thus suggesting that endogenous decay and uncoupled metabolism could be considered significant for sludge reduction. The maximum growth rate μ_H remained constant from Period I to Period II, while the active fraction of heterotrophs significantly decreased. When the plant configuration was changed to the configuration MBR – OSA II (Period III and IV), the biomass was subject to a more extended starvation phase compared to the previous periods since the sludge was recycled in the aerobic reactor of the mainstream line, where the availability of organic carbon was lower compared to the anoxic reactor. Therefore, this prolonged starvation might have induced a maintenance metabolism and cell lysis, as confirmed by the increased SMP discussed above during Periods III and IV [21]. Indeed, under the MBR – OSA II configuration, despite the oxygen availability in the aerobic reactor, the lack of organic carbon did not support the ATP restoration, thus limiting biomass growth. Nevertheless, the main parameters observed in Periods III and IV for the heterotrophic biomass were slightly higher than the corresponding Periods I and II, suggesting a reduced sludge minimisation performance under MBR – OSA II. This result was not in line with recent results achieved in a previous study [16], even if the observed behaviour could be related to the variation of other operating parameters (such as, temperature, F/M, etc.) which might have affected the biomass growth. A slight variation of F/M occurred during the experimental periods due to the real nature of the treated wastewater.

Regarding autotrophic species, respirometric batch tests carried out during experiments revealed an excellent development of nitrifying species, with experimental values in line with literature data [15]. As a general comment, it could be highlighted that the MBR configuration, entailing high SRT values, enhances the development of autotrophic species. Moreover, the presence of the anaerobic reactor in the sludge recycling line did not represent a significant stressor on nitrifying species. Referring to Period I (configuration MBR – OSA I), excepting the first days of anaerobic reactor operation, no considerable stress effect on nitrifying bacteria was observed. Indeed, a significant development of autotrophic activity was noticed, with respirogram charts highlighting significant respiration rates. In Period II (configuration MBR – OSA I), the increase of HRT in the ASRR from 4 to 6 h produced only a small effect on nitrifiers with a low decrease of the main kinetic and stoichiometric parameters, suggesting that this HRT increase did not provide a further stress effect on autotrophic species. In contrast, when configuration MBR – OSA II was implemented (Periods III and IV), it was observed a significant increase of the main parameters compared to the corresponding previous Periods I and II, and only a small decrease of autotrophic activity when the HRT in the ASRR was increased from 4 (Period III) to 6 h (Period IV), respectively. The achieved results confirmed that a long biomass exposure to unaerated conditions induces a reduction of nitrifiers activity, thus worsening the nitrification performance of the system.

3.4. Membrane fouling

Fig. 5a shows the trend of the total resistance R_T. Data from Fig. 5a show that the highest values of R_T were achieved during Periods II and IV. This result suggests that the increase of ASSR HRT increases the membrane fouling for both configurations. In view of better understanding which fraction of resistance mainly contributes to the R_T, Fig. 5b shows the average percentage of the resistance due to the membrane (Rm), the cake layer (Rc) and the pore blocking (Rp).

Data from Fig. 5b shows that with the same configuration, the increase of ASSR HRT causes the rise of pore fouling. Specifically, for configuration MBR-OSA I Rp increases from 74 % (Period I) to 82 % (Period II). Meanwhile, for configuration MBR-OSA II, Rp increases from 85 % (Period III) to 87.5 % (Period IV). Moreover, one can observe that the configuration also influences the Rp value. In particular, under the same ASSR HRT, the Rp value is higher in the MBR-OSA II configuration than in the MBR-OSA I configuration. This latter result is mainly attributed to the increase of total SMP in configuration MBR-OSA II, which causes the increase of irreversible fouling.

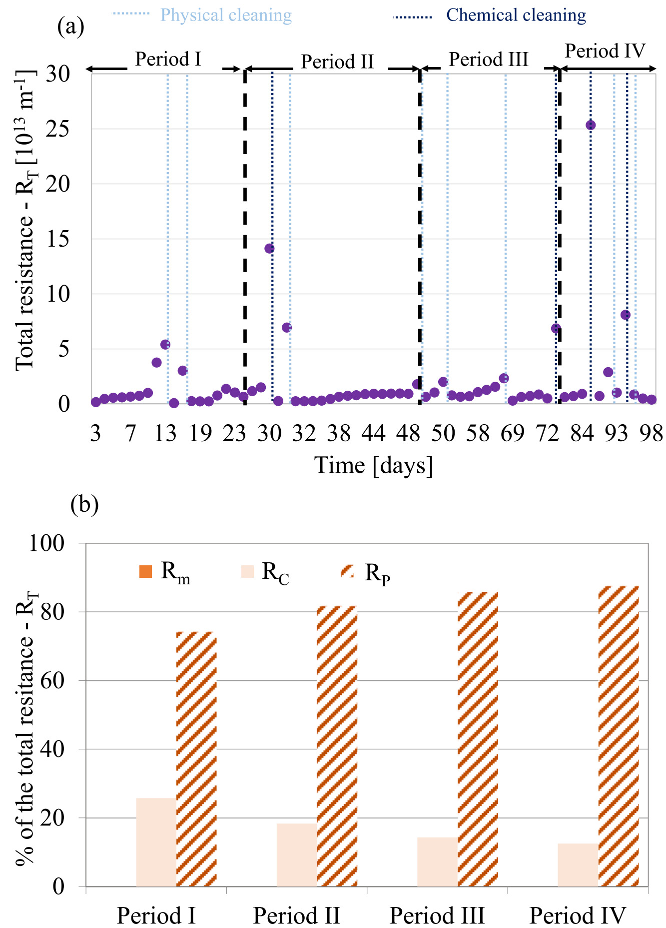

Fig. 5. The trend of R_T (a) and percentage of R_m, R_C and R_{p} were obtained by applying the RIS model (b).

3.5. Greenhouse gases

Fig. 6 depicts the measured gaseous and dissolved N_2O-N concentration inside the biological reactors during the experiments. Data from Fig. 6 shows that the increase of ASSR HRT in the same configuration caused the increase of both liquid and dissolved N_2O concentration in the ASSR. This result is likely due to the anaerobic pathways of N_2O production [22]. Indeed, with the increase of HRT inside the ASSR the anaerobic pathways of N_2O production are favoured. Under anaerobic or anoxic conditions, the presence of hydroxylamine (NH_2OH) could be directly detoxified into N_2O [22]. Indeed, for configuration MBR-OSA I the average (considering all reactors) gaseous and dissolved N_2O concentration was around 0.38 mgN_2O-N L^{-1} during Period I (Fig. 6a and Fig. 6e). During Period II this value increased to 0.46 mgN_2O-N L^{-1} and 0.42 mgN_2O-N L^{-1} for gaseous and dissolved N_2O concentration respectively (Fig. 6 b and Fig. 6f). Specifically, the increase was mainly debited to the anoxic and anaerobic contributions. During Period I, the average gaseous concentrations of N_2O in anoxic and anaerobic reactors were equal to 0.099 and 0.07 mgN_2O-N L^{-1}, respectively (Fig. 6a); the dissolved concentrations were equal to 0.094 and 0.088 mgN_2O-N L^{-1}, respectively (Fig. 6e). With the ASSR HRT (Period II) increase, the gaseous N_2O concentration in the anoxic and anaerobic reactors increased on average to 0.115 and 0.096 mgN_2O-N L^{-1}, respectively. During Period II, the dissolved N_2O concentration in the anoxic and anaerobic reactors increased on average to 0.11 and 0.104 mgN_2O-N

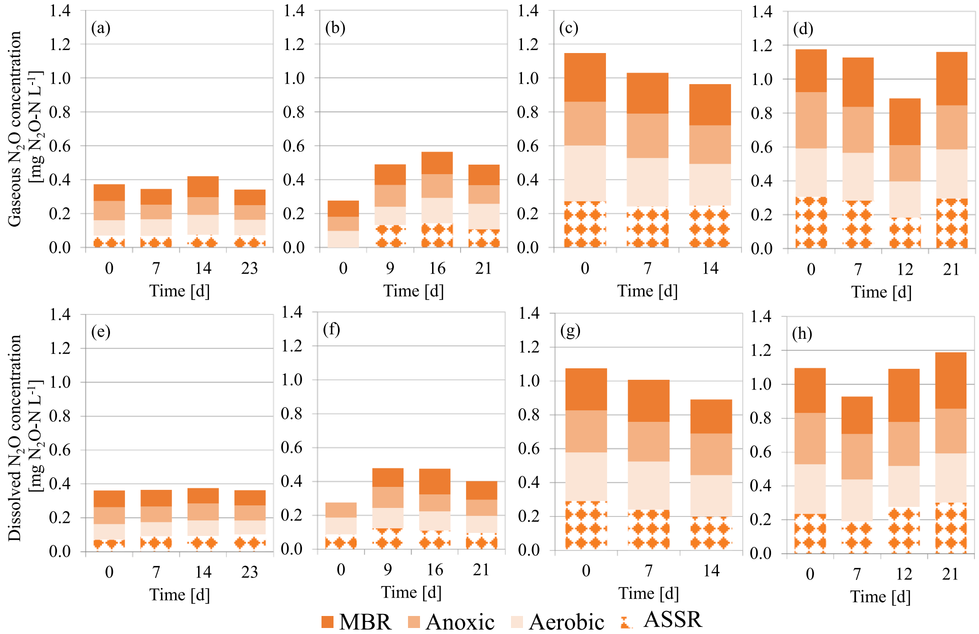

Fig. 6. Average gaseous N_2O-N concentration per biological reactor in Periods I (a), II (b), III (c) and IV (d) and dissolved N_2O-N concentration Periods I (e), II (f), III (g) and IV (h).

L^{-1}, respectively. On average, 35 % of N_2O increased from Period I to Period II in anoxic and anaerobic reactors. The percentage above achieved the average value of 48 % for the configuration MBR-OSA II. When the MBR-OSA II configuration was implemented both gaseous and dissolved N_2O concentrations increased considerably in all reactors with more than double values. Indeed, during Period III, the average gaseous and dissolved N_2O concentrations increased to 1 and 0.98 mgN_2O-N L^{-1}, respectively (Fig. 6c and Fig. 6g). Under MBR-OSA II configuration, the average N_2O concentration (both gaseous and dissolved) increased with the increase of HRT of the ASSR. Indeed, dissolved and gaseous N_2O concentration increased to 1.18 mgN_2O-N L^{-1} during Period IV. The increase of N_2O concentration from MBR-OSA I to MBR-OSA II configuration was likely due to the significant growth of influent ammonia concentrations during Periods III and IV that could have exerted a negative effect on nitrifying bacteria favouring the N_2O production.

Regarding the emission factor, 1.3 % and 2.1 % of the total influent nitrogen were emitted as N_2O during Periods I and II. Meanwhile, 2.9 % and 3.1 % of the total influent nitrogen were emitted as N_2O during Periods III and IV. Thus, the increase of ASSR HRT has mostly influenced the N_2O emission during MBR-OSA I configuration.

4. Conclusions

Two MBR-OSA pilot plant configurations (MBR-OSA I and MBR-OSA II) fed with real wastewater were monitored in view of testing the influence of HRT on the ASSR. The COD removal was not affected by the different OSA configurations and HRT, while nitrification decreased during MBR-OSA II, especially under ASSR HRT of 4 h. The lowest total sewage sludge production occurred during MBR-OSA I under ASSR HRT of 6 h. However, during MBR-OSA II configuration and under ASSR HRT of 6 h, the highest biological sludge reduction occurred (78 % compared to MBR-OSA I configuration). Despite the result above, the MBR-OSA II configuration showed several disadvantages. Indeed, the floc destructuration during MBR-OSA II under ASSR HRT of 6 h caused the highest membrane fouling. Furthermore, the highest N_2O emission factor (3.1 % Period IV) occurred under the MBR-OSA II configuration. Such a fact reveals that MBR-OSA I configuration under ASSR HRT of 6 h provided the best results.

CRediT authorship contribution statement

Paulo Marcelo Bosco Mofatto: Methodology, Validation, Formal analysis, Investigation, Data Curation, Writing - Original Draft, Writing - Review & Editing. **Alida Cosenza:** Conceptualization, Methodology, Validation, Formal analysis, Investigation, Data Curation, Writing - Original Draft, Writing - Review & Editing; **Daniele Di Trapani:** Conceptualization, Methodology, Validation, Formal analysis, Investigation, Data Curation, Writing - Original Draft, Writing - Review & Editing. **Giorgio Mannina:** Conceptualization, Data curation, Formal analysis, Supervision, Validation, Visualization, Writing – original draft, 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.

Data availability

The data that has been used is confidential.

Acknowledgments

This work was funded by the project “Achieving wider uptake of water-smart solutions—WIDER UPTAKE” (grant agreement number: 869283) financed by the European Union’s Horizon 2020 Research and Innovation Programme, in which the first author of this paper, Giorgio Mannina, is the principal investigator for the University of Palermo. The Unipa project website can be found at: https://wideruptake.unipa.it/.

References

[1] M.C. Collivignarelli, A. Abbà, G. Bertanza, M. Baldi, M. Setti, A. Frattarola, M. Carnevale Miino, Treatment of high strength wastewater by thermophilic aerobic membrane reactor and possible valorisation of nutrients and organic carbon in its residues, J. Clean. Prod. 280 (2021).

[2] G. Mannina, L. Barbara, A. Cosenza, Z. Wang, Treatment and disposal of sewage sludge from wastewater in a circular economy perspective, Curr. Dev. Biotechnol. Bioeng. 11-30 (2023), https://doi.org/10.1016/B978-0-323-99920-5.00011-1.

[3] C. Cheng, J. Geng, Z. Zhou, Q. Yu, R. Gao, Y. Shi, L. Wang, H. Ren, A novel anoxic/aerobic process coupled with micro-aerobic/anaerobic side-stream reactor filled with packing carriers for in-situ sludge reduction, J. Clean. Prod. 311 (2021) 127192.

[4] X. Liu, A. Iqbal, H. Huang, F. Zan, G. Chen, D. Wu, Life cycle assessment of deploying sludge minimisation with (sulfidogenic-) oxic-settling-anaerobic configurations in sewage-sludge management systems, Bioresour. Technol. 335 (2021) 125266.

[5] J. Wang, Q. Zhao, W. Jin, J. Lin, Mechanism on minimisation of excess sludge in oxic-settling-anaerobic (OSA) process, Front. Environ. Sci. Eng. China 2 (1) (2008) 36–43.

[6] R. Morello, F. Di Capua, L. Pontoni, S. Papirio, D. Spasiano, U. Fratino, F. Pirozzi, G. Esposito, Microaerobic digestion of low-biodegradable sewage sludge: effect of air dosing in batch reactors, Sustainability 13 (17) (2021) 9869.

[7] G.H. Chen, K.J. An, S. Saby, E. Brois, M. Djafer, Possible cause of excess sludge reduction in an oxic-settling-anaerobic activated sludge process (OSA process), Water Res. 37 (16) (2003) 3855–3866.

[8] R. Ferrentino, M. Langone, G. Andreottola, Sludge reduction by an anaerobic side-stream reactor process: a full-scale application, Environ. Chall. 2 (2021) 100016.

[9] Z. Fida, W.E. Price, B.K. Pramanik, B.R. Dhar, M. Kumar, G. Jiang, F.I. Hai, Reduction of excess sludge production by membrane bioreactor coupled with anoxic side-stream reactors, J. Environ. Manag. 281 (2021).

[10] R. Vitanza, A. Cortesi, M.E. De Arana-Sarabia, V. Gallo, I.A. Vasiliadou, Oxic settling anaerobic (OSA) process for excess sludge reduction: 16 months of management of a pilot plant fed with real wastewater, J. Water Process Eng. 32 (2019) 100902.

[11] G. Mannina, R. Alduina, L. Badalucco, L. Barbara, F.C. Capri, A. Cosenza, D. Di Trapani, G. Gallo, V.A. Laudicina, S.M. Muscarella, D. Presti, Water resource recovery facilities (Wrrfs): the case study of Palermo University (Italy), Water 13 (2021) 3413.

[12] G. Mannina, L. Badalucco, L. Barbara, A. Cosenza, D. Di Trapani, G. Gallo, V. A. Laudicina, G. Marino, S.M. Muscarella, D. Presti, H. Helness, Enhancing a transition to a circular economy in the water sector: the EU project WIDER UPTAKE, Water 13 (7) (2021) 946, https://doi.org/10.3390/w13070946.

[13] APHA, Standard Methods for the Examination of Water and Wastewater. Standard Methods, 2012.

[14] G. Mannina, M. Capodici, A. Cosenza, P. Cinà, D. Di Trapani, A.M. Puglia, G. A. Ekama, Bacterial community structure and removal performances in IFAS-MBRs: a pilot plant case study, J. Environ. Manag. 198 (2017) 122–131.

[15] G. Mannina, C. Morici, A. Cosenza, D. Di Trapani, H. Ødegaard, Greenhouse gases from sequential batch membrane bioreactors: a pilot plant case study, Biochem. Eng. J. 112 (2016) 114–122.

[16] Mannina, G., Cosenza, A., Di Trapani, D., Bosco Mofatto, P.M., 2023b. Sludge reduction and phosphorus removal by oxic settling anaerobic (OSA) in view of a circular economy approach. Submitted to J. Environ. Manag.

[17] M. Coma, S. Rovira, J. Canals, J. Colprim, Minimization of sludge production by a side-stream reactor under anoxic conditions in a pilot plant, Bioresour. Technol. 129 (2013) 229–235.

[18] J. Huang, Z. Zhou, Y. Zheng, X. Sun, S. Yu, X. Zhao, A. Yang, C. Wu, Z. Wang, Biological nutrient removal in the anaerobic side-stream reactor coupled membrane bioreactors for sludge reduction, Bioresour. Technol. 295 (2020) 122241.

[19] V.F. Velho, P. Foladori, G. Andreottola, R.H.R. Costa, Anaerobic side-stream reactor for excess sludge reduction: 5-year management of a full-scale plant, J. Environ. Manag. 177 (2016) 223–230.

[20] Metcalf & Eddy, Inc., Wastewater Engineering: Treatment and Reuse, McGraw-Hill, Boston, 2003.

[21] A. Cosenza, D. Di Trapani, P.M. Bosco Mofatto, G. Mannina, Sewage sludge minimisation by OSA-MBR: a pilot plant experiment, Chemosphere 347 (2023) 140695, https://doi.org/10.1016/j.chemosphere.2023.140695.

[22] Z. Lin, K. Ma, Y. Yang, Nitrous oxide emission from full-scale Anammox-driven wastewater treatment systems, Life (Basel) 12 (7) (2022) 971.

[23] D. Di Trapani, G. Di Bella, G. Mannina, M. Torregrossa, G. Viviani, Comparison between moving bed-membrane bioreactor (MB-MBR) and membranebioreactor (MBR) systems: influence of wastewater salinity variation, Bioresour.Technol. 162 (2014) 60–69.

[24] G. Di Bella, D. Di Trapani, S. Judd, Fouling mechanism elucidation in membrane bioreactors by bespoke physical cleaning, Separation and Purification Technology 199 (2018) 124–133.