Dewaterability of CAS and MBR Sludge: Effect of Biological Stability and EPS Composition

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Abstract: The dewaterability of sludge from two conventional activated sludge (CAS) and three membrane bioreactor (MBR)–based wastewater treatment plants is investigated prior to and after anaerobic digestion. The concentration and composition of extracellular polymeric substances (EPS) mostly affect the dewaterability of all raw sludge samples. Better sludge dewaterability is observed when the concentration of proteins, carbohydrates, uronic acids, and humic acids is below approximately 400, 250, 200, and 40 mg/L, respectively. In contrast, the specific resistance to filtration (SRF) increases in the sludge samples with a higher EPS concentration. The MBR results in a lower EPS production and a uronic acid–dominating EPS composition. This especially affects the dewaterability of one MBR sludge, also characterized by high salinity and a smaller particle size. Anaerobic digestion results in a higher SRF for both CAS and MBR sludge, with the particle-size distribution having the preponderant effect on the digested sludge dewaterability. **DOI: 10.1061/(ASCE)EE.1943-7870.0001299.**

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Introduction

Sludge dewatering, stabilization, and disposal often result in significant management costs for wastewater treatment plants (WWTPs) (Wei et al. 2003). Dewatering is mainly aimed at decreasing the bound water content, which consists of interstitial, vicinal, and hydration water, which represent the most difficult fractions of water to separate. Moreover, it has been observed that an increasing energy demand is required to achieve water separation from sludge with a lower water content (Mowla et al. 2013). Therefore, a correct understanding of the dewatering phenomena would help to enhance water separation with the advantage of decreasing the amount of disposed sludge and the dewatering costs (Ginestet 2006).

Among the several parameters influencing sludge dewatering, the origin of sludge is one of the most important factors because it affects the morphological, physical, and chemical properties of the sludge flocs (Jin et al. 2004). The dewaterability of sludge from plants treating industrial wastewater strongly depends on the wastewater characteristics (Mannina et al. 2016c). Similarly, the activated sludge bioreactor technology has been demonstrated to affect the sludge properties (Pontoni et al. 2015). Sludge deriving from conventional activated sludge (CAS) and membrane bioreactor (MBR) systems results in different dewatering efficiency. The MBR configuration has been reported to produce a sludge with a lower mean particle size and, thus, worse dewaterability properties (Capodici et al. 2016). However, the different dewaterability of sludge from CAS and MBR plants remains rather unexplored and requires investigation.

The anaerobic digestion of excess sludge from WWTPs is a well-established technology to enhance the sludge energetic value and simultaneously reduce the amount of sludge requiring disposal and its environmental impact (Appels et al. 2008). In WWTPs, anaerobic digestion is normally performed prior to sludge dewatering, because this has been thought to increase sludge dewaterability

(Lawler et al. 1986). However, digested sludge often presents worse dewatering characteristics, which are strongly related to a higher concentration of extracellular polymeric substances (EPS) (Houghton et al. 2000). EPS up to a certain concentration are essential to the formation of sludge flocs and other forms of microbial aggregations such as biofilms or granules (Mattei et al. 2015; D'Acunto et al. 2016). However, an excessive EPS amount can result in a lower sludge water separation (Houghton et al. 2001; Li and Yang 2007). In addition to EPS abundance, EPS composition plays a major role in affecting sludge dewaterability (Pontoni et al. 2016). Proteins have the capacity to retain water, and elevated protein levels result in an increasing amount of interstitial water and lower sludge dewatering ability (Cetin and Erdincler 2004; Sheng et al. 2010). In contrast, the efficiency of sludge dewatering is higher when an increasing carbohydrate fraction is present in the EPS matrix (Cetin and Erdincler 2004). However, because the effect of EPS concentration and composition is still rather contradictory, further investigation is needed in this direction.

Finally, the particle size of sludge flocs is another parameter that has long been investigated as influencing factor of sludge dewaterability (Karr and Keinath 1978). The presence of supracolloidal (1–100 μm) particles is generally associated with a lower sludge dewatering efficiency (Liming et al. 2009). Anaerobic digestion contributes to the formation of smaller sludge particles and a more homogeneous particle-size distribution (PSD), which most likely hinder sludge dewatering (Lawler et al. 1986). Nevertheless, the knowledge of the different PSD of sludge samples deriving from CAS and MBR systems and its influence on sludge stability and dewaterability remains limited.

This paper investigates: (1) the biomethane potential of the sludge deriving from two CAS-based and three MBR-based WWTPs in order to evaluate the biological stability by biomethane potential (BMP) tests; (2) the dewaterability of both raw and digested sludge in terms of specific resistance to filtration (SRF), time to filter (TTF), and capillary suction time (CST); (3) the EPS characterization in terms of proteins, carbohydrates, uronic acids, and humic acids before and after anaerobic digestion; (4) the effect of EPS concentration and composition on the dewaterability of the sludge samples; and (5) the role of the PSD of the raw sludge flocs on sludge dewaterability.

Materials and Methods

Sources of Sludge

This study used sludge samples from two CAS-based and three MBR-based WWTPs for BMP and dewaterability tests. Table 1 reports the operating parameters of the originating WWTPs. All the sludge samples were directly taken from the secondary clarifiers or the membrane units in order to exclude the solids deriving from the primary clarification, if present. The CAS1 and CAS2 sludge samples were collected from the full-scale WWTPs located in Potenza, Italy and Anacapri, Italy, respectively. The MBR1 and MBR2 sludge samples were collected from two pilot-scale laboratory MBRs located in Palermo, Italy. A real municipal wastewater was mixed with a synthetic solution and fed to MBR1 in order to study the effect of the C/N ratio on the performance of MBR1, operated with a University of Cape Town (UCT) configuration (Mannina et al. 2016a). A high-salinity (20 g/L NaCl) and high–total petroleum hydrocarbon (TPH) (20 mg/L) synthetic wastewater was supplemented to MBR2 as reported by Mannina et al. (2016b, c). Finally, the MBR3 sludge was taken from the municipal WWTP in Capri, Italy. The sludge deriving from all WWTPs was thickened by clarification for 2 h in the laboratory and the supernatant was removed prior to performing BMP tests and each analysis.

BMP Tests

Biomethane potential batch tests were carried out under controlled mesophilic (35 ± 1°C) conditions using 1-L glass bottles (Schott Duran, Wertheim, Germany). Each bottle was fed with 700 mL of sludge used as both the organic substrate and the source of microorganisms. Subsequently, all the bottles were sealed with a 5-mm silicone disk. The BMP experiments were performed in triplicate and the partial biomethane production was recorded over 75–90 days. During this period, the bottles were manually shaken once per day. The biomethane production was measured by the water displacement method, in accordance with the procedure described by Esposito et al. (2012). The alkaline solution used to entrap CO_2 was 12% NaOH.

The digestate obtained after the completion of the BMP tests was collected and used for the measurement of the dewaterability parameters.

Determination of Dewaterability Parameters

The dewaterability of raw and digested sludge was assessed by the determination of SRF, TTF, and CST. The SRF and CST were measured using the method reported by Pontoni et al. (2016). The TTF was evaluated using the same apparatus adopted for SRF determination. An initial sludge volume of 100 mL was used for the MBR2 sludge, whereas 200 mL was used for the other sludge samples. The TTF was determined when 50% of the initial sludge volume was filtered according to the standard methods (APHA 1998). The final CST and TTF values were expressed in $s \cdot L/g$ total solids (TS), normalizing by the TS concentration of each sludge.

Statistical Analysis

In order to evaluate the significance of the correlation between the EPS constituents and the dewaterability parameters (SRF and CST), a statistical analysis was performed using Microsoft *Excel*. The results were considered statistically significant when the p-value obtained was below 0.05.

Table 1. Operating Conditions of the Five WWTPs Originating the CAS and MBR Sludge Used in This Study

Sludge sample	COD (mg/L)	N-NH4+ (mg/L)	NaCl (g/L)	TPH (mg/L)	HRT (h)	SRT (days)	TSS (g/L)	Membrane type
CAS1	200	15	—	—	24	30	3.52	—
CAS2	600	75	—	—	18	10	3.50	—
MBR1	455	76	—	—	20	144	3.45	Hollow fiber
MBR2	350	50	20	20	16	—	7.85	Hollow fiber
MBR3	350	35	—	—	24	35	9.05	Flat sheet

Note: COD = chemical oxygen demand; HRT = hydraulic retention time; TSS = total suspended solids.

Table 2. TS and VS of Each Raw and Digested Sludge

Sludge sample	Prior to AD			After AD			Δ (%)	
	TS (%)	VS (%)	VS/TS	TS (%)	VS (%)	VS/TS	TS (%)	VS (%)
CAS1	1.80[a]	1.27	0.71	1.20	0.70	0.59	−33	−45
CAS2	1.40	1.11	0.79	0.89	0.63	0.70	−36	−43
MBR1	1.26	1.03	0.82	0.83	0.60	0.72	−34	−42
MBR2	3.08	0.63	0.21	2.86	0.42	0.15	−7	−33
MBR3	0.98	0.72	0.73	0.71	0.46	0.65	−28	−36

[a]1% corresponds to 10 g TS/L.

Analytical Methods

Total and volatile solids (VS) of both raw and digested sludge were determined in according to the standard methods (APHA 1998) (Table 2). Extracellular polymeric substances were extracted from 100 mL of each sludge before and after anaerobic digestion. Extracellular polymeric substances extraction was performed using a Dowex Marathon C (Sigma-Aldrich, Munich, Germany) cation exchange resin using the method reported by Frølund et al. (1996). Subsequently, EPS were characterized in terms of proteins (Lowry et al. 1951), carbohydrates (Dubois et al. 1956), uronic acids (Blumenkrantz and Asboe-Hansen 1973; Kintner and Van Buren 1982), and humic acids (Frølund et al. 1996). Spectrophotometric determinations were acquired by means of a Photolab UV-Vis 3000 (WTW, Weilheim, Germany) spectrophotometer. The PSD of the raw sludge samples was measured by means of a QICPIC particle size image scanning analyzer (Sympatec, Clausthal-Zellerfeld, Germany), equipped with a flow-cell dispersion device.

Results and Discussion

BMP of CAS and MBR Sludge

The BMP of all CAS and MBR sludge samples was investigated in batch digestion experiments under mesophilic conditions (35 ± 1°C). Fig. 1 shows the cumulative biomethane production obtained with the five sludge samples as the average of the triplicates sampled over 75–90 days. The biomethane potential was substantially higher for the CAS1 and CAS2 sludge compared with that obtained for all MBR sludge samples. In particular, the CAS2 sludge resulted in a final specific biomethane production of 312 ± 1 mL CH_4/g VS, 17% higher than that achieved for the CAS1 sludge. Similar VS removal of 45 and 43% (Table 2) was obtained during the anaerobic digestion of the CAS1 and CAS2 sludge, respectively, indicating that VS removal did not affect the final biomethane production. The higher biomethane yield of the CAS2 sludge was due to the lower sludge retention time (SRT) and, thus, a lower initial stability.

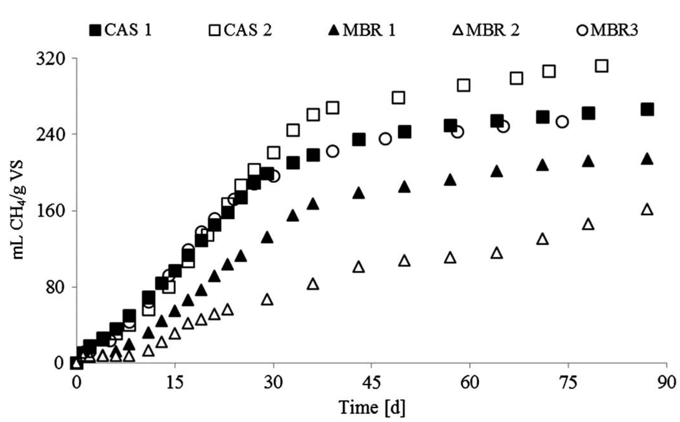

Fig. 1. Profiles of cumulative biomethane production from CAS and MBR sludge (with standard deviation in the range 0–7 mL CH_4/g VS)

Compared with CAS systems, higher SRT and lower sludge production are commonly reported among the advantages of MBRs (Liu and Tay 2001). This especially occurs in the case of MBR-upgraded CAS systems, because the upgrading results in a higher concentration of microorganisms and SRT (Ghyoot and Verstraete 2000). However, this is not generally observed in newly built MBRs, in which the higher suspended solid concentration used for the bioreactor sizing results in a lower reactor volume and an overall amount of microorganisms that is comparable to that maintained in CAS systems. This also implies a lower SRT and the need for further digestion treatment for MBR sludge (Ng and Hermanowicz 2005a). This was confirmed in the present study, with all the MBR sludge samples resulting in a final cumulative biomethane production higher than 160 mL CH_4/g VS. In particular, the highest biomethane production of approximately 255 mL CH_4/g VS was observed for the MBR3 sludge, which was only 3% lower than that achieved with the CAS1 sludge after 75 days.

The biomethane production for the MBR1 sludge reached 223 ± 4 mL CH_4/g VS, i.e., 16% lower than that obtained with the MBR3 sludge. This was most probably due to the significantly higher SRT and sludge stability achieved during MBR1 operation prior to anaerobic digestion in the BMP tests.

Finally, the lowest biomethane production was observed for the sludge from MBR2, aimed at the treatment of a high-salinity and petroleum hydrocarbon-containing wastewater. The final specific biomethane yield and the VS removal were only 161 ± 5 mL CH_4/g VS and 33%, respectively. This indicates a significant inhibitory effect of salinity and hydrocarbons on the activity of methanogenic bacteria, as also confirmed elsewhere. Anwar et al. (2016) observed only a 10% inhibition of the methanogenic activity with a sodium salt concentration below 8 g/L. Biomethane production was repressed up to 80% when 16 g/L of salinity was supplemented. Scherr et al. (2012) reported that methanogens required 11 months of adaptation to petroleum hydrocarbons during the anaerobic digestion of crude oil and the degradation of a hydrocarbon-contaminated soil.

CAS and MBR Sludge Dewaterability before and after Anaerobic Digestion

In this study, the dewaterability of the sludge samples from CAS and MBR systems was investigated by measuring the SRF, TTF, and CST (Table 3). The MBR1 and MBR3 samples were operated with synthetic and real domestic wastewater, respectively, and resulted in sludge with the lowest SRF (1.29×10^{13} and 0.61×10^{13} m/kg TS, respectively). The SRF for the CAS1 and CAS2

Table 3. Dewaterability Properties of Each Raw and Digested Sludge in Terms of SRF, TTF, and CST

Sludge sample	Prior to AD			After AD		
	SRF (10^{13} m/kg TS)	TTF (s · L/g TS)	CST (s · L/g TS)	SRF (10^{13} m/kg TS)	TTF (s · L/g TS)	CST (s · L/g TS)
CAS1	1.37	38.62	1.04	6.36	120.60	2.08
CAS2	3.24	86.62	1.71	33.80	208.70	9.41
MBR1	1.29	39.99	1.19	6.92	35.24	2.90
MBR2	4.39	51.88	1.87	2.21	100.15	1.80
MBR3	0.61	16.41	1.02	0.77	69.35	1.92

raw sludge was 1.37×10^{13} and 3.24×10^{13} m/kg TS, respectively, i.e., higher than that obtained with the MBR1 and MBR3 sludge. In terms of CST, the highest CST value was 1.71 s · L/g TS for the CAS2 sludge; the CAS1 and MBR3 sludge had similar CST (1.04 and 1.02 s · L/g TS, respectively), that were lower than the 1.19 s · L/g TS obtained for the MBR1 sludge. The significantly higher SRF of the CAS2 sludge was most likely due to the lower SRT, because a good inverse linear correlation between SRF and SRT has been previously reported (Pontoni et al. 2016). Comparable values of the dewaterability properties were observed for CAS1, MBR1, and MBR3 sludge, in agreement with Pollice et al. (2007), who reported similar SRF and CST for CAS and MBR sludge. However, this is contrary to other studies that reported a higher resistance to filtration of MBR sludge (Ng and Hermanowicz 2005b). Moreover, it is noteworthy that a good correlation between SRF, TTF, and CST was not observed in this study. This was mainly due to the different analytical procedures used for the determination of the three parameters (Smollen 1986). Specific resistance to filtration was considered as the most reliable parameter to evaluate the sludge dewaterability through filtration.

The dewaterability of all the sludge samples was also evaluated after anaerobic digestion. To date, the role that anaerobic digestion plays in sludge dewaterability is still controversial. In full-scale WWTPs, sludge dewatering is commonly performed after a chemical or biological stabilization. However, Houghton et al. (2000) observed a threefold-higher CST for a digested than a raw sludge. More recently, Capodici et al. (2016) also reported a lower CST for pre-digested CAS and MBR sludge samples. In the present study, the anaerobic digestion of the CAS1, CAS2, MBR1, and MBR3 sludge resulted in increasing SRF, TTF, and CST (Table 3). The greatest increase of the dewaterability parameters was observed for the digested CAS2 sludge, with SRF, TTF, and CST increasing to 33.80×10^{13} m/kg TS, 208.70 s · L/g TS, and 9.41 s · L/g TS, respectively. The CAS1 and MBR1 sludge dewaterability was similarly affected by anaerobic digestion, confirming their comparable behavior. The MBR3 sludge had the best dewaterability properties after anaerobic digestion.

The different operating conditions of MBR2 exerted a significant effect on the raw sludge dewaterability. Higher SRF, TTF, and CST were obtained for the raw MBR2 sludge compared with the other sludge samples. In particular, salinity has been reported to favor the release of fine sludge particles and decrease the dewatering efficiency (Raynaud et al. 2012). Mannina et al. (2016b) observed that 20 mg/L of NaCl resulted in twofold higher CST values than those obtained up to 14 mg/L of NaCl. Nonetheless, anaerobic digestion enhanced MBR2 sludge dewaterability properties, with an approximately 50% of SRF decrease, mainly due to the significant decrease of uronic acids in the EPS matrix.

Effect of EPS Concentration and Composition on Sludge Dewaterability

The importance of EPS for sludge structure and formation of sludge agglomerates has been thoroughly demonstrated. A low amount of EPS results in filtration resistance and a high degree of sludge particle dispersion (Mikkelsen and Keiding 2002). However, most authors agree that higher EPS concentration induces worse sludge dewaterability characteristics (Rosenberger and Kraume 2002), with the EPS composition also playing a major role (Liming et al. 2009).

Fig. 2 shows the EPS composition of the five raw and digested sludge samples in this study. The CAS1 and CAS2 raw sludge had a similar EPS composition, primarily made of proteins (57–61%), followed by carbohydrates (28–29%), humic acids (6–7%), and uronic acids (4–8%). A higher variability of the EPS composition was observed for the MBR raw sludge samples. The protein (17–50%), carbohydrate (13–22%), and humic acid (2–4%) percentages were lower on average than those observed for the CAS sludge, in spite of a higher uronic acid concentration (32–60%). Above all, the extreme halophilic conditions and the presence of hydrocarbons induced the microorganisms to develop an emulsifying uronic acid–dominating EPS structure in the MBR2 sludge (Iyer et al. 2006). Ozturk and Aslim (2010) also observed a modification of the EPS composition of three cyanobacteria under salt stress with an enhanced uronic acid production. They reported a higher overall EPS production in the presence of salt, because of the high tolerance of the specific cyanobacteria to salinity. This was not observed in the present study—the EPS concentration reached only 547 mg EPS/L [Fig. 2(b)] in the MBR2 sludge, most likely due to the deactivation of some metabolic functions.

The highest total EPS concentrations, 1,525 and 1,218 mg EPS/L, were observed in the CAS2 and CAS1 sludge, respectively. A total EPS production of 849 and 665 mg EPS/L was achieved for the MBR1 and MBR3 sludge, respectively. Therefore, in addition to a different EPS composition, MBR operation resulted in a lower total EPS production than that obtained in the CAS sludge. It is known that the retention of the microorganisms in the MBR systems due to the presence of the membranes leads to a lower biomass flocculation (Ng and Hermanowicz 2005a). However, such a flocculation ability could result in better dewaterability of the MBR sludge (Table 3), without having the detrimental effect which was observed for the CAS sludge at higher total EPS concentrations.

A direct correlation between total EPS concentration and raw sludge dewaterability parameters has been reported in previous studies (Li and Yang 2007; Ning et al. 2014; Pontoni et al. 2016). In this study, the SRF and CST of CAS1, CAS2, MBR1, and MBR3 sludge were plotted against the concentration (mg/L) of proteins, carbohydrates, uronic acids, and humic acids (Fig. 3). The MBR2 sludge was excluded from this analysis because its EPS composition substantially differed from that of the other sludge samples. In particular, the percentage of uronic acids was considerably higher than that observed in the other sludge samples. This might explain the higher water retention and the worse dewaterability properties of the MBR2 sludge. Uronic acids are associated with the formation of hydrogels, which are capable of establishing a strong polar interaction with the interstitial water in the presence

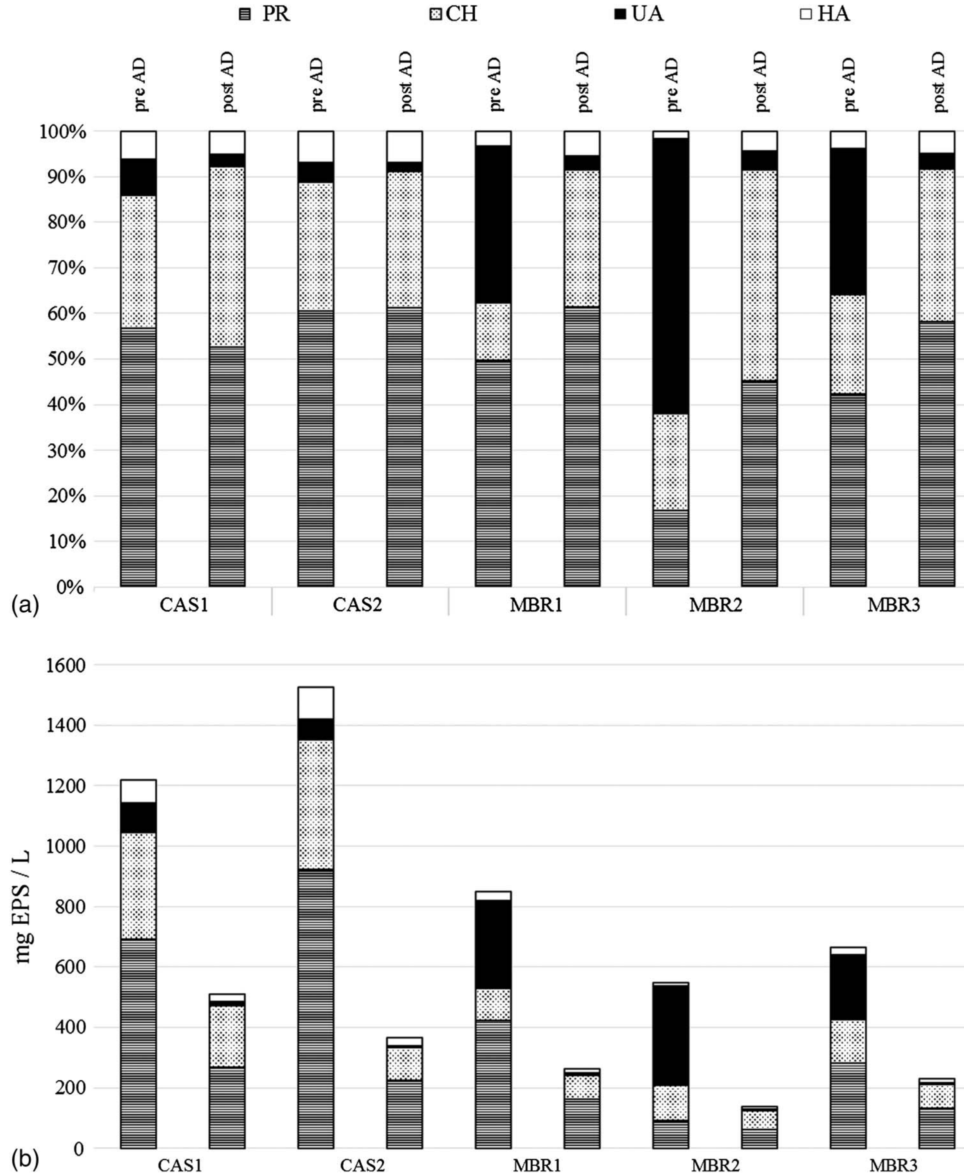

Fig. 2. Proteins (PR), carbohydrates (CH), uronic acids (UA), and humic acids (HA) in the EPS matrix of CAS and MBR sludge prior to and after AD: (a) percentage; (b) concentration

of charged carboxylic groups in the polyuronate molecules (Bellich et al. 2009).

Fig. 3 shows that all the EPS constituents had good quadratic correlation with the dewaterability parameters, regardless of the CAS or MBR technology used in the originating WWTP. However, only the correlation of proteins and carbohydrates with SRF was statistically significant, with p-values lower than 0.05 (Table 4). The parabolic trend of the regression curve indicates that proteins, carbohydrates, uronic acids, and humic acids positively influenced sludge dewaterability up to approximately 400, 250, 200, and 40 mg/L (Fig. 3), respectively. Extracellular polymeric substances are well known to be fundamental for sludge floc formation, and this effect likely prevailed at lower EPS constituent concentrations. However, at increasing protein, carbohydrate, uronic acid, and humic acid concentrations, higher SRF and CST were achieved. This both agrees with and contradicts the existing literature. Feng et al. (2009) and Ning et al. (2014) also observed a parabolic correlation between dewaterability parameters and EPS, but did not analyze the effect of each EPS constituent. Jin et al. (2004) confirmed that proteins and carbohydrates had a considerable contribution to enhance the water-binding ability of the sludge flocs. Cetin and Erdincler (2004) also reported a negative trend between increasing protein content and sludge filterability and compactibility. In contrast, Wilén et al. (2003) reported that flocculation ability and filterability of the sludge were steadily enhanced with increasing protein and carbohydrate content, but were negatively affected by the total EPS concentration. They explained this difference by highlighting the importance of the mechanisms of flocculation and the efficiency of EPS extraction. In contrast to proteins and carbohydrates, humic acid concentrations higher than approximately 30 mg/L negatively affected sludge dewaterability properties (Wilén et al. 2003), as also confirmed by the present study.

Anaerobic digestion resulted in a decrease of the total EPS concentration ranging between 58 and 76% for each sludge [Fig. 2(b)]. Nevertheless, the EPS composition of the CAS1 and CAS2 sludge was not significantly affected by anaerobic digestion. This indicates that all the EPS constituents were similarly degraded during the

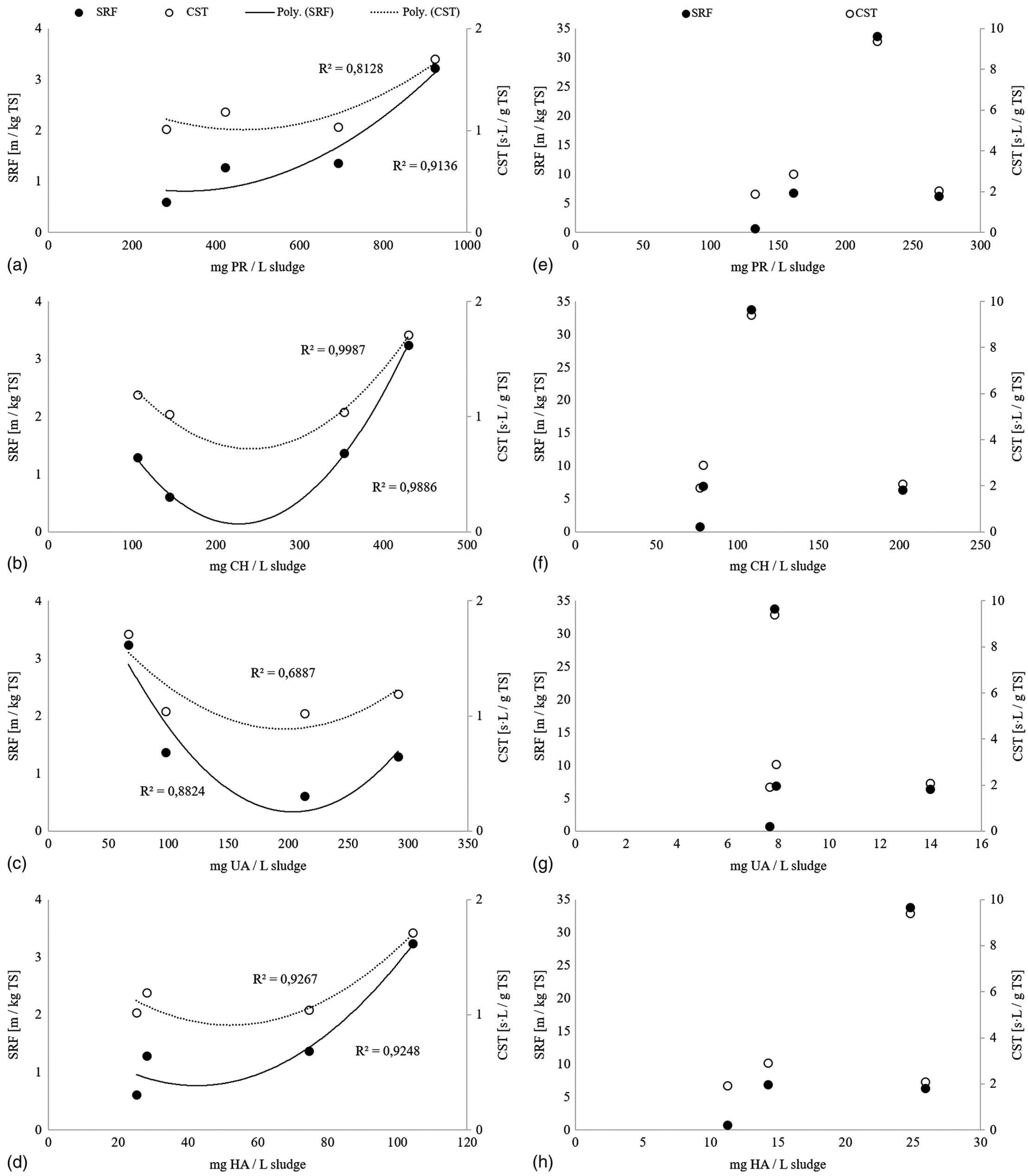

Fig. 3. Correlation between sludge dewaterability properties (SRF and CST) and (a and e) PR; (b and f) CH; (e and g) UA; (d and h) HA in (a–d) raw sludge; (e–h) digested sludge

digestion process. With regard to the MBR sludge samples, the highest removal was observed for uronic acids that decreased from 292, 329, and 214 mg/L to 8, 6, and 8 mg/L in the MBR1, MBR2, and MBR3 sludge, respectively. This presumably explained the better dewaterability properties of the MBR2 sludge after anaerobic digestion. In contrast, the effect of a smaller sludge particle size prevailed over EPS composition for the MBR1 and MBR3 sludge, resulting in higher SRF and CST for both sludge samples. The SRF and CST remained constant at increasing EPS constituent concentrations [Figs. 3(e–h)].

Table 4. *p*-Values Obtained by Performing a Statistical Analysis Evaluating the Significance of the Correlation of Proteins, Carbohydrates, Uronic Acids, and Humic Acids with SRF and CST in Raw CAS1, CAS2, MBR1, and MBR3 Sludge Samples

EPS constituent	SRF	CST
Proteins	**0.0468**	0.2519
Carbohydrates	**0.0354**	0.1069
Uronic acids	0.3429	0.5580
Humic acids	0.0826	0.0616

Note: Significant correlations are highlighted in bold.

Effect of Sludge Particle Size

Fig. 4 shows the profiles of the cumulative distribution and distribution density of the raw CAS and MBR sludge. The CAS1 and CAS2 samples resulted in sludge with the most heterogeneous PSD and the highest distribution density, corresponding to a particle size of approximately 94 and 77 μm (Table 5), respectively. A lower mean particle size was observed for the MBR sludge, with a minimum of approximately 63 μm for the MBR2 sludge (Table 5). This was validated by other studies that reported a higher deflocculation in MBRs (Wisniewski and Grasmick 1998; Ng and Hermanowicz 2005b). The presence of nonflocculated (1–100 μm) particles normally leads to lower sludge dewatering efficiency (Liming et al. 2009). This also contributed, along with the high salinity and uronic acid content, to the higher SRF and CST obtained for the MBR2 sludge compared with the other raw sludge samples.

The mean particle size of digested sludge generally decreases with a more homogeneous PSD (Lawler et al. 1986). In this study, the lower mean particle size after anaerobic digestion (data not available) most likely resulted in the lower dewaterability of CAS and MBR sludge. Moreover, PSD had a greater effect than EPS composition on the digested sludge dewaterability. The PSD did not differently affect the SRF and CST of the digested CAS and MBR sludge, because both parameters increased after anaerobic digestion. Similarly, Feng et al. (2009) observed good correlation of PSD with CST and SRF. The CST and SRF significantly increased in sludge with 90% of particles smaller than 80 μm. This was further confirmed by Liming et al. (2009), who reported a linear trend between CST and PSD for particles smaller than 15 μm.

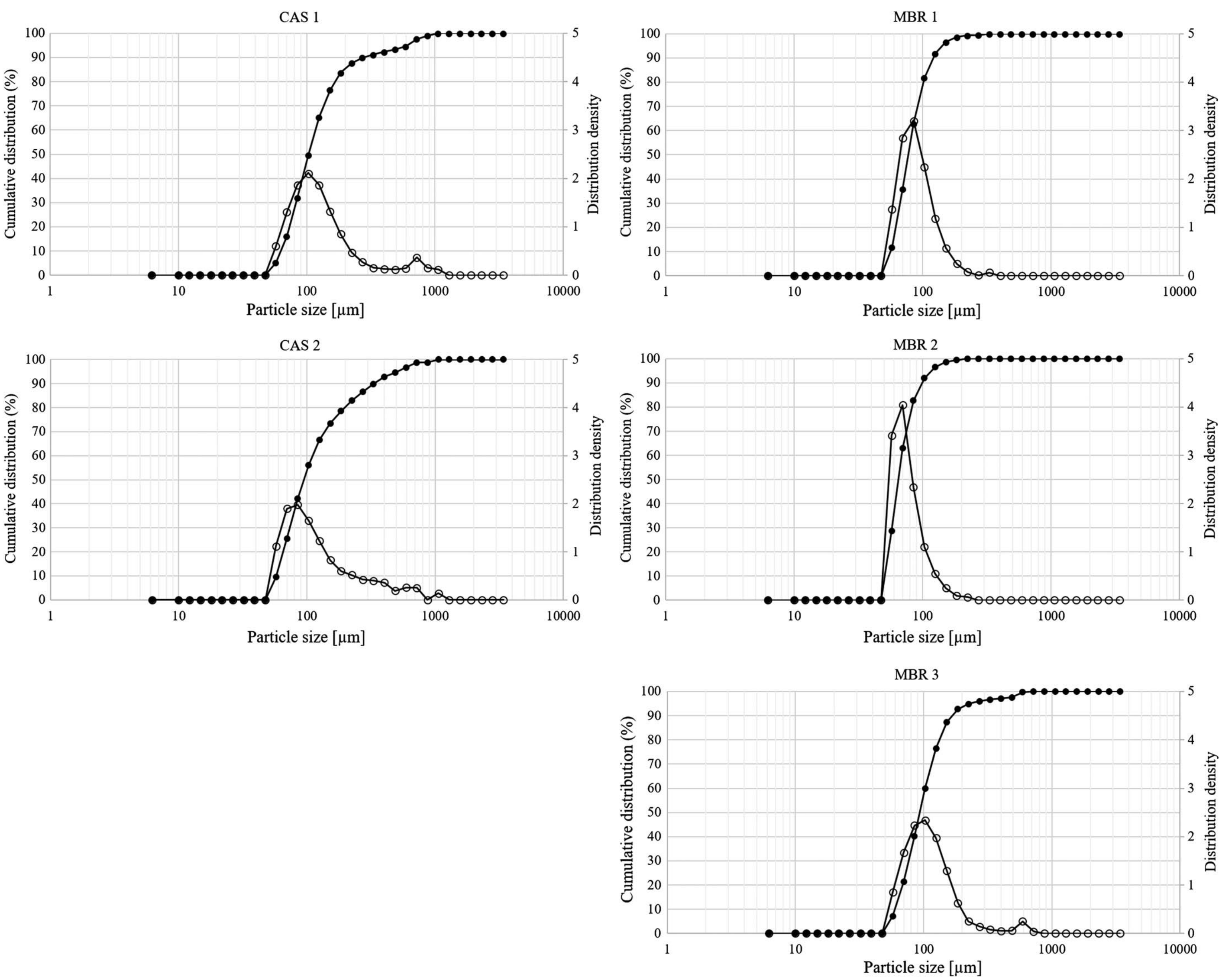

Fig. 4. Cumulative distribution (solid circles) and distribution density (hollow circles) of all raw sludge particles

Table 5. PSD (μm) of the Sludge Particles from CAS1, CAS2, MBR1, MBR2, and MBR3 Samples

Sludge sample	×(10%)	×(50%)	×(60%)	×(90%)	×(99%)
CAS1	63.08	103.60	117.64	272.70	877.77
CAS2	57.95	95.06	111.34	333.33	901.16
MBR1	56.12	77.77	883.30	121.28	206.78
MBR2	50.89	65.21	68.83	99.26	165.70
MBR3	59.98	93.91	103.16	168.12	558.75

Conclusions

Prior to anaerobic digestion, the raw CAS and MBR sludge dewaterability was mainly affected by the EPS concentration and composition. Lower overall EPS production in the MBR sludge resulted in a lower SRF, which was considered as the most reliable parameter for the evaluation of the sludge filterability. The raw sludge samples from both CAS and MBR showed a significant quadratic correlation between proteins, carbohydrates, and SRF. Uronic acids played a major role, especially for the raw MBR2 sludge dewaterability, which was also strongly affected by high salinity. The dewaterability properties of the sludge samples from CAS and MBR worsened after anaerobic digestion. The effect of a more homogeneous PSD and smaller sludge particles likely prevailed over EPS composition, resulting in higher SRF and CST. Finally, this study confirmed that all the sludge samples from MBRs were not biologically stable and required further stabilization. The sludge samples from MBR3 reached a final biomethane production of approximately 255 mL CH_4/g VS, i.e., 3% lower than that obtained with the CAS1 sludge.

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