


Feasibility of OFMSW co-digestion with sewage sludge for increasing biogas production at wastewater treatment plants

Annika Björn^{1,4}  · Sepehr Shakeri Yekta^{1,4} · Ryan M. Ziels^{2,4} · Karl Gustafsson^{1,4} · Bo H. Svensson^{1,4} · Anna Karlsson^{3,4}

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Abstract Sweden has the ambition to increase its annual biogas production from the current level of 1.9 to 15 TWh by 2030. The unused capacity of existing anaerobic digesters at wastewater treatment plants is among the options to accomplish this goal. This study investigated the feasibility of utilizing the organic fraction of municipal solid waste (OFMSW) as a co-substrate, with primary and waste-activated sewage sludge (PWASS) for production of biogas, corresponding to 3:1 ratio on volatile solid (VS) basis. The results demonstrated that co-digestion of OFMSW with PWASS at an organic loading rate of 5 gVS l⁻¹ day⁻¹ has the potential to increase the biogas production approximately four times. The daily biogas production increased from 1.0 ± 0.1 to 3.8 ± 0.3 l biogas

l⁻¹ day⁻¹, corresponding to a specific methane production of 420 ± 30 Nml methane gVS⁻¹ during the laboratory experiment. Co-digestion of OFMSW with PWASS showed a 50:50 distribution of hydrogenotrophic and acetoclastic methanogens in the digester and enhanced the turnover kinetics of intermediate products (acetate, propionate, and oleate). Practical limitations potentially include the need for sludge dewatering to maintain a sufficient hydraulic retention time (17 days in this study), as well as additional energy consumption for mixing due to an increased sludge apparent viscosity (from 1.8 ± 0.1 to 45 ± 4.8 mPa*s in this study) at elevated OFMSW-loading rates.

Keywords Anaerobic co-digestion · Biogas · OFMSW · Sewage sludge · Viscosity

✉ Annika Björn
annika.bjorn@liu.se

Sepehr Shakeri Yekta
sepehr.shakeri.yekta@liu.se

Ryan M. Ziels
ziels@uw.edu

Karl Gustafsson
keb.gustafsson@gmail.com

Bo H. Svensson
bo.svensson@liu.se

Anna Karlsson
anna.karlsson@scandinavianbiogas.com

- ¹ Department of Thematic Studies-Environmental Change, Linköping University, Linköping, Sweden
- ² Department of Civil Engineering, University of British Columbia, Columbia, Canada
- ³ Scandinavian Biogas Fuels AB, Stockholm, Sweden
- ⁴ Biogas Research Center, Linköping University, Linköping, Sweden

Introduction

Sweden has the ambition to increase its annual biogas production from the current level of 1.9 to 15 TWh by 2030 (Swedish Energy Agency 2016). The unused capacity of existing anaerobic digesters at wastewater treatment plants (WWTP) can be a means to accomplish this goal. As for most WWTP in Europe, the Swedish plants have anaerobic digestion facilities and infrastructure to make use of produced biogas for heating and for electricity generation (Aichinger et al. 2015). However, the anaerobic digesters at WWTP are often operated at suboptimal conditions, i.e., low organic loading rates (OLR), typically 1.0–1.5 kg volatile solids (VS) m⁻³ day⁻¹, with low biogas yields and production rates (Aichinger et al. 2015). Thus, the biogas production at WWTP could be improved substantially by supplementing organic wastes other than sewage sludge to

the anaerobic digester units (Aichinger et al. 2015; Kaan Dereli et al. 2010). The organic fraction of municipal solid waste (OFMSW) is an attractive substrate for anaerobic digestion due to its high abundance and availability (e.g., Mata-Alvarez et al. 2000; Kaan Dereli et al. 2010; Moestedt et al. 2016) and a biogas potential of 400–500 ml gVS⁻¹ (Hansen et al. 2007). Furthermore, a directive by the Swedish government enforcing biological treatment of 50% of the OFMSW from year 2018 (Swedish Ministry of Environment 2012) has motivated the Swedish municipalities to invest in source separation of OFMSW.

Anaerobic digestion biotechnologies offer sustainable biological treatment approaches, which enable simultaneous volume reduction, waste stabilization, and energy recovery from OFMSW (Kaan Dereli et al. 2010; Aichinger et al. 2015). The co-digestion of WWTP sludge with OFMSW implies management and energy recovery from two major organic waste streams of our society, as energy-carrying biogas is produced (Hartmann et al. 2002). In addition, the nutrient content of OFMSW can be recovered by application of the residues from the anaerobic digestion processes for soil conditioning and fertilization. Several studies report on the use of OFMSW for co-digestion with sewage sludge (e.g., Hartmann et al. 2002; Sosnowski et al. 2003; Heo et al. 2004; Bolzonella et al. 2006; Kaan Dereli et al. 2010). In an investigation of the co-digestion of waste-activated sludge with OFMSW, Bolzonella et al. (2006) observed a 24% increase in biogas yield (from 0.21 to 0.26 m³ kgVS⁻¹), with an increase in the OLR from 1.0 to 1.2 kgVS m⁻³ day⁻¹. Aichinger et al. (2015) investigated the co-digestion of OFMSW with thermally pre-treated primary and waste-activated sewage sludge (PWASS) in digesters operated at an OLR of 5.3 kgVS m⁻³ day⁻¹. A stepwise increase in OLR using thermally pre-treated OFMSW to 7 kgVS m⁻³ day⁻¹ resulted in a 35% higher biogas production rate and an increase in VS reduction from 53 to 57%. During stepwise increase of the OLR, there was no indication of disturbances of the digester performance (Aichinger et al. 2015). Heo et al. (2004) investigated the stability and efficiency of a laboratory-scale anaerobic reactor co-digesting food waste (FW) and waste-activated sludge (WAS) at various hydraulic retention times (HRT: 10–20 days) and waste mixture ratios (FW:WAS). They reported an HRT of 13 days and a FW:WAS ratio of 50:50 on VS basis as optimum, which yielded a specific methane production of 0.32 m³ kgVS⁻¹ and a VS reduction of 57% at an OLR of 2.4 kgVS m⁻³ day⁻¹.

Common limiting factors for increasing the OLR with, e.g., OFMSW in anaerobic digesters, are related to accumulation of degradation intermediate products such as volatile fatty acids (VFA) and long-chain fatty acids (LCFA), due to an imbalance in the rates of the microbial pathway of the anaerobic degradation chain (Yenigüngen

and Demirel 2013). Another concern is the risk of bulking of the sludge due to an increase in viscosity caused by co-digestion. Bulking of the sludge may cause severe damage to biogas installations (Lindorfer and Demmig 2016). Such shifts in viscosity of the reactor sludge may result in problems with foaming and inadequate mixing, ultimately leading to breakdown of stirrers (Nordberg and Ekström 2005; Björn et al. 2012; Kougias et al. 2014; Moeller et al. 2012; Moeller and Görsch 2015). Previous research on the rheology, including viscosity, of anaerobic digester sludge has mainly focused on plants digesting sewage sludge. Yet, there are some publications reporting that substrate composition, other than sewage sludge, affects the reactor sludge viscosity (e.g., Björn et al. 2012; Nges et al. 2012; Mönch-Tegeger et al. 2015). Furthermore, Björn et al. (2012) demonstrated that similar total solid (TS) values do not necessarily correspond to similar viscosity values when different substrates, other than sewage sludge, were anaerobically digested in laboratory-scale reactors. Mönch-Tegeger et al. (2015) showed a large impact of the type of substrate, the TS content, and the mechanical disintegration of the substrate on the viscosity in a manure-based full-scale biogas process. However, information on the effects of OFMSW co-digestion with sewage sludge on rheological properties of the slurry in anaerobic digesters at WWTPs is scarce.

The aim of this study was to evaluate the feasibility of OFMSW at high mass proportion as a co-substrate for production of biogas in sewage sludge anaerobic digesters. For this purpose, the fraction of OFMSW in the influent feed was stepwise increased to 70% of VS in a mixture with sewage sludge, corresponding to a three-time increase in the process OLR. To assess the process performance, we monitored gas production and composition, pH, VFA, NH₄⁺-N, TS and VS, elemental composition of the digestate, dynamics of the methanogenic community, turnover kinetics of intermediate products, and viscosity of the reactor sludge.

Materials and methods

Reactor setup and operation

Two 5 l semi-continuous, stirred, tank reactors (R1 and R2) with working volume of 4 l were inoculated using anaerobic digester sludge from Henriksdal WWTP in Stockholm, Sweden. Both digesters operated at 37 °C for 200 days. Primary and waste-activated sewage sludge were collected every second week, stored at 4 °C, and used as a feed substrate for R1 and R2. The PWASS loading rate was 1.3–1.8 gVS l⁻¹ day⁻¹ depending on the TS and VS contents of the PWASS. The HRT was 20 days. From day 54, the OLR was increased stepwise to 5.0 gVS l⁻¹ day⁻¹ in R1

Table 1 Daily additions (ml) and average organic loading rates (gVS l⁻¹ day⁻¹) of primary sludge (PS), waste-activated sludge (WAS), and organic fraction of municipal solid waste (OFMSW) during different operational periods of R1 and the corresponding changes in hydraulic retention time (HRT) of the process

Operational period (days)	PS		WAS		OFMSW		HRT (days)
	Loading rate (gVS l ⁻¹ day ⁻¹)	Volume (ml)	Loading rate (gVS l ⁻¹ day ⁻¹)	Volume (ml)	Loading rate (gVS l ⁻¹ day ⁻¹)	Volume (ml)	
0–53	1.0 ± 0.1	148	0.39 ± 0.07	52	0.0	0.0	20
54–67	1.2 ± 0.1	148	0.36 ± 0.11	52	0.0 → 1.0	0 → 30	20 → 17
68–105	0.9 ± 0.1	148	0.44 ± 0.06	52	1.0	30	17
106–112	0.9 ± 0.0	74 (dewatered)	0.32 ± 0.00	52	1.0	30	17 → 26
113–140	1.1 ± 0.1	74 (dewatered)	0.34 ± 0.02	52	1.0 → 3.5	30 → 105	26 → 17
140–200	0.9 ± 0.1	74 (dewatered)	0.44 ± 0.04	52	3.5	105	17

by introducing a slurry of OFMSW (TS: 15 wt% and VS: 90% TS) which was collected from Åby biogas plant in Linköping, Sweden, and hygienized at 70 °C for at least 1 h. R2 was operated as a control without OFMSW. The addition of OFMSW reduced the HRT of reactor R1 from 20 to 17 days (Table 1; Fig. 1b). To increase the loading rate of the OFMSW, either the OLR of PWASS needed to be decreased or the substrate needed to be dewatered to maintain an HRT above 16 days. The limit was set according to earlier experiences and was also deemed suitable as the HRT of the full-scale plant in Henriksdal is between 16 and 20 days. A lower OLR of PWASS to the downstream anaerobic digester at the WWTP may limit the flow of sewage sludge through the upstream processes. To avoid a decrease of the OLR of PWASS, we applied dewatering of the primary sludge (Table 1), which had the highest water content. The dewatering was done by centrifugation at 2300 relative centrifugal force for 2 min using a Heraeus Megafuge 16 centrifuge (Thermo Scientific). Reactor sludge corresponding to the volume of the substrate was withdrawn every day prior to the feeding.

The gas production from R1 and R2 was monitored online using gas meters based on the principle of water displacement (MilliGas counters, Ritter, Germany). Once a week, biogas was collected during 24 h and its composition (i.e., CH₄, CO₂, O₂, and H₂S) was determined using a Biogas Check Analyzer (Geotechnical Instruments, UK). All methane gas volumes are given at the standard temperature and pressure (STP; 273 K and 1 atm; unit Nml, N = “normal”). pH was measured according to European Standard EN 12,176 by use of a pH meter (infoLab pH 7310, from WTW, Germany). The NH₄⁺-N concentration was measured spectrophotometrically using Hach kit LCK304 from Hach Lange, Germany, according to the manufacturer’s instruction. Concentrations of VFA, including acetate, propionate, butyrate, iso-butyrate, valerate, iso-valerate, capronate, and iso-capronate, were measured twice a week according to Jonsson and Borén

(2002). Total solid and VS were quantified once a week according to Swedish Standard method SS028113. The VS reduction was calculated using the mass balance method (Eq. 1), and the concentration of free ammonia in the reactor liquid was calculated using Eq. (2):

$$1 - (Q_{\text{out}} \times VS_{\text{out}} / Q_{\text{in}} \times VS_{\text{in}}) \quad (1)$$

$$[\text{NH}_3 - \text{N}] / \text{NH}_4^+ - \text{N} = 1 + \left(10^{-\text{pH}} / 10^{-(0.09018 + 2729.92/T(\text{K}))} \right) \quad (2)$$

Elemental composition of the sludge from R1 and R2 was determined by monthly analyses of the total concentration of selected macronutrients (N, P, K, Mg, Na, Ca, and S), micronutrients (Fe, Cu, Ni, Zn, Co, Mn, Mo, Se, and W), and heavy metals (Pb, Cr, and Cd). The elemental analyses were performed by Eurofins Environment Testing Sweden AB (Lidköping, Sweden), according to Swedish standard method SS0281502.

Analysis of the methanogenic community composition

Every week, 10-ml sludge samples from R1 and R2 were collected and centrifuged at 10,000×g for 10 min (4 °C). The supernatant was removed and the pellets stored at -20 °C until further analyses of the methanogenic community composition. Samples from days 50, 114, 136, and 188 were used for DNA extraction. DNA was isolated using the PowerSoil[®] RNA/DNA Isolation Kit (MO BIO, Inc. Carlsbad, California). The concentration of extracted DNA was measured with the Quant-IT dsDNA High Sensitivity Assay Kit with Qubit 2.0 fluorometer (Invitrogen, Carlsbad, California), according to the manufacturer’s instruction. Extracted DNA samples were stored in nuclease-free water at -20 °C before being analyzed by quantitative polymerase chain reaction (qPCR), in a LightCycler 480 system (Roche Diagnostics GmbH,

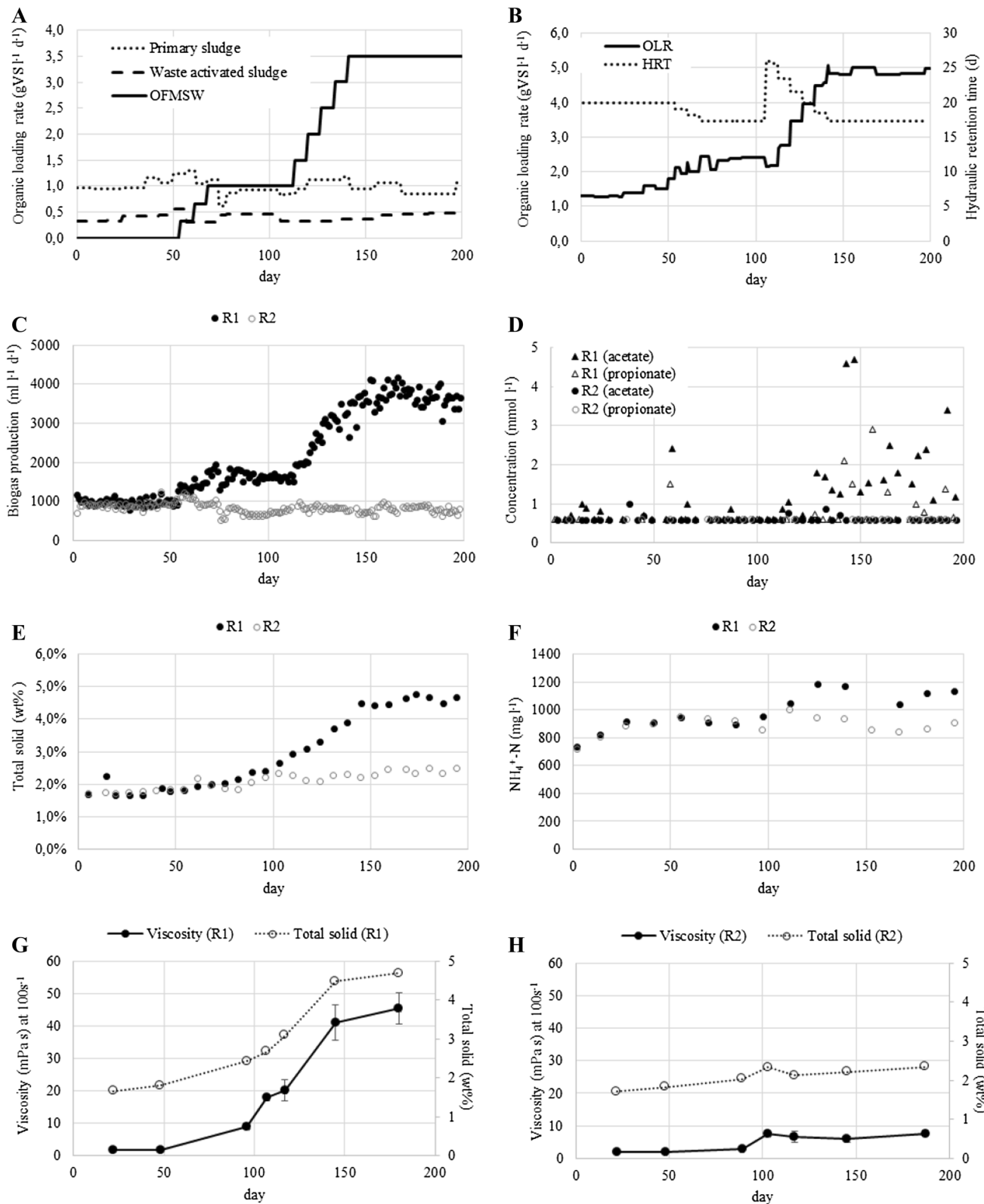


Fig. 1 **a** Loading rates of primary sludge, waste-activated sludge, and OFMSW for R1. **b** OLR and HRT in R1. **c** Biogas production per litre of reactor per day in R1 and R2. **d** Concentration of acetate and propionate in R1 and R2. **e** Total solid contents in R1 and R2. **f** Concentration of NH₄⁺-N in R1 and R2. **g** Apparent viscosity at

shear rate 100 s⁻¹ together with total solid content for R1 during increased OLR by OFMSW amendment. **h** Apparent viscosity at shear rate 100 s⁻¹ together with total solid content for R2 with no amendment of OFMSW

Mannheim, Germany). The qPCR targeted 16S rRNA gene segments of the methanogenic orders of *Methanomicrobiales*, *Methanobacteriales*, *Methanococcales*, and the methanogenic families of *Methanosarcinaceae* and *Methanosaetaceae*. Detailed descriptions of the qPCR reaction mix, primer, and probe sets, DNA amplification method, and calibration standards are given in Ziels et al. (2015). The DNA extraction and qPCR analysis were performed in triplicate for each sampling point.

Viscosity analysis

Samples, collected from R1 at days 22, 48, 96, 107, 117, 145, and 180, and from R2 at days 22, 48, 89, 103, 117, 145, and 187, were used for sludge viscosity analysis. Apparent viscosities were determined using a rotational rheometer (RheolabQC, Anton Paar, Holland) with a CC27-SN19237 measuring system and a C-LTD80/QC cell, coupled with the Rheoplus software (Anton Paar). Rheograms including viscosity and flow curves were obtained by a three-step protocol according to Björn et al. (2012). Measurements were performed in triplicates at 37 °C immediately after collecting the samples. A certified viscosity reference standard (Cannon® RT1000) was used for instrument quality control.

Analysis of microbial capacity to degrade intermediate products

Batch assays were run at 37 °C using the sludge from R1 and R2 as inoculum to study changes in the microbial capacity to degrade intermediate products, including acetate, propionate, and oleate (as model substance for LCFA), as well as phenyl acetate (as model substance for monoaromatics). The batch experiment was set up as reported by Karlsson et al. (2012). The methane potential of reactor sludge collected at days 47, 110, and 187 was determined in batch assays. In brief, each bottle was filled with 135 ml of growth medium containing essential macro- and micronutrients (Karlsson et al. 2012; Feng et al. 2010), purged with a gas mixture of 80% N₂ and 20% CO₂, and inoculated with 15 ml of reactor sludge before sealed with rubber stoppers and screw caps. Subsequently, 15 mmol l⁻¹ acetate, 15 mmol l⁻¹ propionate, 5 mmol l⁻¹ oleate, and 5 mmol l⁻¹ phenyl acetate were separately supplemented in triplicate bottles, all bottles were then incubated at 37 °C for 260 h. Triplicate control bottles with only inoculum from the reactors were also included in the experiment. To determine the biogas production, the gas pressure in each bottle was measured using a pressure meter (Testo 312, Testo Instruments, Germany). Gas samples were collected for headspace methane content analysis according to Karlsson et al. (2012). The accumulated methane

production is reported at atmospheric pressure and 37°C. The methane production rates, representing the conversion rate of the intermediate products to methane, were calculated by fitting a Gompertz function, $M = MP \times \exp[-\exp(Rm \times e \times (\lambda - t)/MP + 1)]$, to the accumulated methane production (M) data plotted against incubation time (t) (Nopharatana et al. 2007). In this equation, MP, Rm, and λ represent methane production potential, methane production rate, and lag time, respectively. Model parameters were estimated by nonlinear regressions using the SPSS software (IBM, USA).

Results and discussion

Process performance at elevated loading rate of OFMSW

Upon introduction of 1.0 gVS l⁻¹ day⁻¹ of OFMSW to R1, the HRT decreased from 20 to 17 days due to an increase in volume of the substrate mix (Table 1; Fig. 1a, b). Under these operational conditions, the average biogas production increased from 1.0 ± 0.1 l biogas l⁻¹ day⁻¹ (between days 0 and 54) to 1.7 ± 0.1 l biogas l⁻¹ day⁻¹ (between days 68 and 105), while the average specific methane production for these two periods was the same (390 ± 50 and 390 ± 37 Nml methane gVS⁻¹, respectively). After the dewatering performed to further increase the OFMSW loading, the TS content of the primary sludge increased from 3.4 ± 0.4 wt% (between days 0 and 105) to 6.6 ± 0.9 wt% (between days 106 and 200). The substrate mix with the dewatered primary sludge resulted in an HRT of 26 days at OLR of 2.1 gVS l⁻¹ day⁻¹ (Fig. 1b). Subsequently, the stepwise elevation of OFMSW from 1.0 to 3.5 gVS l⁻¹ day⁻¹ between days 113 and 140 resulted in a decrease in the HRT from 26 to 17 days (Fig. 1b). The reactors were operated for three further HRT to assess the stability of the process (days 140–200).

Increasing the OFMSW-loading rate to 3.5 gVS l⁻¹ day⁻¹ (corresponding to total OLR of 5.0 gVS l⁻¹ day⁻¹) yielded a daily biogas production of 3.8 ± 0.3 l biogas l⁻¹ day⁻¹ (days 140–200; Fig. 1c) and a specific methane production of 420 ± 30 Nml methane gVS⁻¹. At higher loading of OFMSW, a slight elevation in acetate and propionate levels was observed, but both stayed below 3 mmol l⁻¹ throughout the experiment (Fig. 1d). All other VFAs stayed below the detection limit for both R1 and R2 all times. pH remained stable at 7.3 ± 0.1 in both reactors over the experimental period (data not shown). The TS content of R1 increased from 2.0 to 4.5% (Fig. 1e) with a VS content of 70 ± 1.5% of TS. The VS reduction was significantly (t test, $p < 0.01$) higher in R1 (62 ± 2.0% of influent VS) at OLR of 5.0 gVS l⁻¹ day⁻¹ compared to the control reactor

($49 \pm 2.5\%$ of influent VS). No significant effect of co-digestion was observed on the H_2S content of the biogas, implying no need for additional measures to remove H_2S from the biogas for further applications. However, the $\text{NH}_4^+\text{-N}$ level increased from 0.9 to 1.2 g l^{-1} in R1 at the elevated OLR and was higher than the control reactor (Fig. 1f). The increase in $\text{NH}_4^+\text{-N}$ corresponds to an increase in free ammonia N from 25 to 40 mg l^{-1} (as calculated from Eq. 2) and was below the reported inhibition concentration ($>100 \text{ mg l}^{-1}$ depending on system; reviewed by Yenigüngen and Demirel 2013). Aichinger et al. (2015) reported a similar increase in ammonium/ammonia concentration, mainly due to the degradation of protein content of OFMSW. The results in this study show that a stable process could be obtained at OFMSW-loading rate of $3.5 \text{ g gVS l}^{-1} \text{ day}^{-1}$ and at an HRT of 17 days and that the elevated ammonia concentration did not affect the microbial activity negatively.

The addition of OFMSW and the corresponding increase in TS contributed to an increase in apparent viscosity of the reactor sludge in R1 from 1.8 ± 0.1 to $45 \pm 4.8 \text{ mPa}\cdot\text{s}$ (Fig. 1g). This would be expected, since an increase in TS often results in a higher fluid viscosity (Seyssiecq et al. 2003). This is in agreement with Battistoni et al. (1993; Battistoni 1997), who reported that the rheological characteristics of OFMSW specifically were correlated with the total TS and VS. The viscosity of the reactor sludge in the control reactor (R2) ranged between 2.0 ± 0.2 and $7.7 \pm 0.3 \text{ mPa}\cdot\text{s}$ throughout the experiment (Fig. 1h), which was in the same range as previously reported values for reactor liquids of processes digesting sewage sludge (Baudez et al. 2011, 2013). It should also be noted that in addition to TS, changes in parameters such as, e.g., particle size, polymeric substances, and ion concentrations, may have contributed to changes in viscosity of the R1 upon co-digestion of OFMSW as an additional substrate (Nges et al. 2012; Moestedt et al. 2016; Lindorfer and Demmig 2016). Accordingly, increased energy consumption for mixing of reactor material due to elevated OFMSW loading is expected as an effect of co-digestion of OFMSW with PWASS. However, no problems associated with foaming were experienced upon the inclusion of OFMSW in R1 nor was there any need for reinforced mixing.

Effects of co-digestion on the elemental composition of the sludge

Elemental composition of sludge samples from R1 and R2, including concentrations of elements important for both process performance and sludge quality as a biofertilizer (i. e., micronutrients, macronutrients, and heavy metals) is listed in Table 2. The concentrations of the elements decreased in R1 over time (except for Na and K) due to a

dilution caused by the addition of the OFMSW. In particular, a substantial decrease in the concentration of Fe (from 76 to 46 g kgTS^{-1}) and P (from 40 to 24 g kgTS^{-1}) was observed in R1 over time as compared to R2 (Table 2). Relatively high Fe content of PWASS sustains low H_2S levels in the biogas in sewage sludge digesters, by FeS (s) precipitation (Shakeri Yekta et al. 2014). Despite a decrease of the Fe content caused by the addition of OFMSW, the H_2S content of the biogas remained below 40 ppm over the course of the experiment. Thus, at the conditions applied in the present study, no extra precautions are needed to control the H_2S concentration of the biogas formed.

The sodium concentration increased from 6.1 to 14 g kgTS^{-1} in R1, while it varied between 3.2 and 5.7 g kgTS^{-1} in R2. Similarly, the K concentration increased from 11 to 19 g kgTS^{-1} in R1, while it ranged between 5.3 and 11 g kgTS^{-1} in R2. Furthermore, co-digestion of OFMSW with PWASS resulted in a lower concentration of metals, in particular Cu, Zn, and Pb in sludge from R1, compared to R2, and there was an indicative decrease for the trace elements Ni, Co, Se, and W (Table 2). The variations in macro- and micronutrient concentrations in R2 originate from varying levels in the PWASS substrate, received every second week from WWTP. The changes in the macro- and micronutrient contents of the reactor liquid did not influence the overall performance of the anaerobic digestion process at the investigated loading rate, but it might affect the quality of the digestate as a biofertilizer.

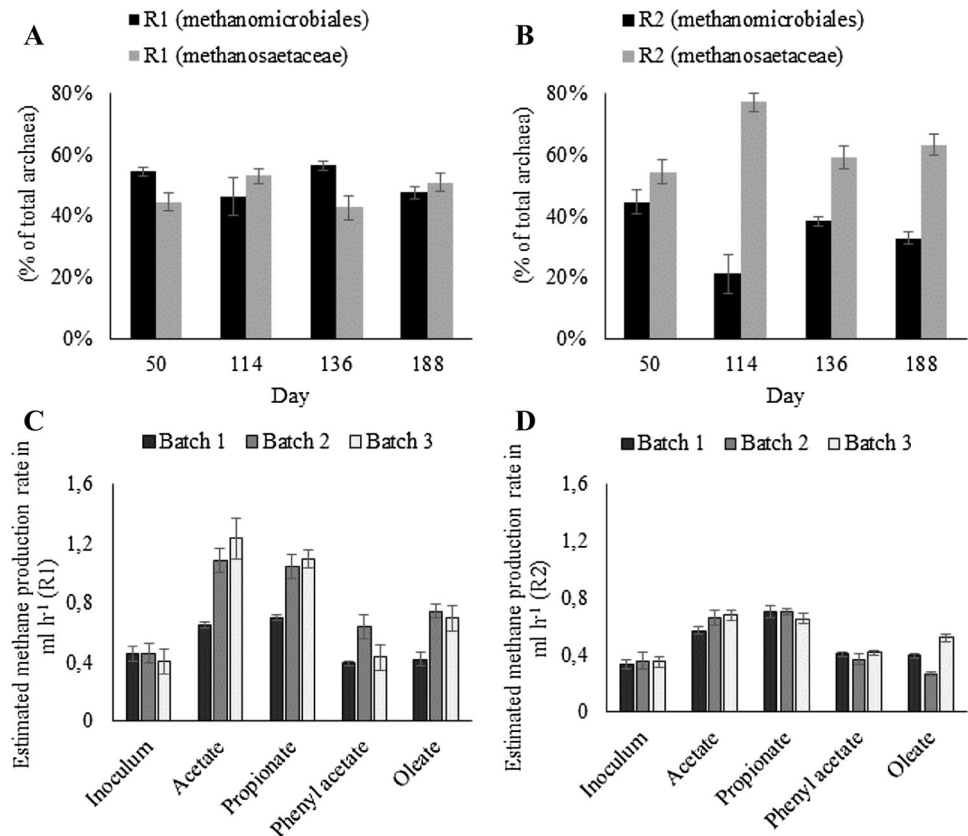
Effects of co-digestion on methanogenic community and methane production rates from intermediate products

The methanogenic community was initially dominated by hydrogenotrophic *Methanomicrobiales* (55 ± 1 and $45 \pm 3\%$ of the total archaea for R1 and R2, respectively) and acetoclastic *Methanosaetaceae* (45 ± 4 and $55 \pm 4\%$ of the total archaea for R1 and R2, respectively). The relative abundance of these major methanogenic groups in R1 remained relatively constant throughout the experiment, while the control reactor, R2, showed higher fractions of the acetoclastic *Methanosaetaceae* (Fig. 2). The observed distribution is in line with the results obtained by Sundberg et al. (2013) in a survey of Swedish WWTP biogas reactors, where the abundance of acetoclastic *Methanosaetaceae* ranged 68–87%, while hydrogenotrophic methanogens made up for the rest of the methanogenic community. Among these, *Methanoculleus*, *Methanospaera*, and *Methanobrevibacter* were the most common. Occurrence of the high proportion of hydrogen-utilizing methanogens of the order *Methanomicrobiales*, to

Table 2 Concentration of selected elements (in mg kgTS⁻¹) in the OFMSW used and in sludge samples regularly collected during the experiment from R1 (co-digester) and R2 (control) reactors. An.U stands for analytical uncertainty

Day	An.U (%)	OFMSW					R1					R2							
		26	51	98	124	166	198	26	51	98	124	166	198	26	51	98	124	166	198
N	10	28,000	94,000	115,000	80,000	69,000	76,000	83,000	84,000	90,000	86,000	71,000	92,000	83,000	84,000	90,000	86,000	71,000	92,000
P	15	5200	40,000	38,000	29,000	21,000	24,000	39,000	37,000	45,000	33,000	32,000	35,000	39,000	37,000	45,000	33,000	32,000	35,000
K	20	12,000	11,000	14,000	15,000	17,000	19,000	10,000	11,000	8700	6900	5500	5300	10,000	11,000	8700	6900	5500	5300
Mg	15	1500	5200	5200	4000	3700	3600	5100	5500	5500	4000	3900	3400	5100	5500	5500	4000	3900	3400
Na	20	7100	6100	7600	8800	11,000	14,000	5700	5400	3600	3800	3200	3500	5700	5400	3600	3800	3200	3500
Ca	15	20,000	36,000	30,000	19,000	18,000	18,000	36,000	38,000	30,000	19,000	24,000	23,000	36,000	38,000	30,000	19,000	24,000	23,000
S	20	2700	12,000	12,000	9100	7800	8700	12,000	11,000	13,000	9000	9300	10,000	12,000	11,000	13,000	9000	9300	10,000
Fe	15	340	76,000	77,000	55,000	43,000	45,000	77,000	67,000	98,000	91,000	110,000	110,000	77,000	67,000	98,000	91,000	110,000	110,000
Cu	15	11	380	360	270	240	220	370	410	460	370	410	390	370	410	460	370	410	390
Ni	15	<2.5	100	200	160	91	81	140	150	120	92	69	66	140	150	120	92	69	66
Zn	15	41	570	500	420	370	320	580	100	620	540	610	520	580	100	620	540	610	520
Co	20	<2.5	6.3	7.5	4.8	3.1	2.5	7.0	6.1	7.9	4.6	5.2	4.7	7.0	6.1	7.9	4.6	5.2	4.7
Mn	15	45	170	200	140	110	110	170	170	200	130	140	120	170	170	200	130	140	120
Mo	20	<1	20	39	32	18	16	9.9	25	22	17	13	12	9.9	25	22	17	13	12
Se	20	<1	2.9	2.3	1.7	1.1	1.3	2.3	2.9	3.2	1.9	2.1	2.0	2.3	2.9	3.2	1.9	2.1	2.0
W	25	<1	6.1	7.5	4.9	3.1	2.3	7.0	6.2	8.6	5.4	5.1	4.0	7.0	6.2	8.6	5.4	5.1	4.0
Pb	25	0.66	14	16	14	15	9.9	14	16	20	16	26	18	14	16	20	16	26	18
Cr	15	<2.5	160	320	260	150	130	290	250	190	140	100	99	290	250	190	140	100	99
Cd	15	0.1	0.70	0.74	0.62	0.54	0.56	0.73	0.80	0.81	0.65	0.76	0.62	0.73	0.80	0.81	0.65	0.76	0.62

Fig. 2 Relative abundance of the dominant methanogens, *Methanosaetaceae* and *Methanomicrobiales*, in R1 (a) and R2 (b) reactors, as determined using qPCR. Triplicate samples were analyzed at each day. Comparison of the methane production rate in batch assays inoculated with sludge collected from R1 (c) and R2 (d) reactors at days 47 (batch 1), 110 (batch 2), and 187 (batch 3)



which *Methanoculleus* belongs, may be a sign of the need for very low partial pressure of hydrogen for the syntrophic degradation of some intermediate compounds. Furthermore, it suggests that syntrophic acetate oxidation (SAO) may occur in parallel with the aceticlastic methanogenic pathway, as hydrogenotrophes such as *Methanoculleus* are often hydrogen scavenging organisms in conjunction with SAO (cf. Neubeck et al. 2016).

The 16S rRNA gene concentration of *Methanomicrobiales* was significantly higher (*t* test, $p < 0.01$) in R1 compared to R2 at the end of the experiment ($10,700 \pm 1200$ and 7400 ± 1200 16S rRNA gene copy number ng DNA⁻¹, respectively). Thus, the co-digestion of OFMSW seemed not to affect the initial composition of the methanogenic flora in R1. However, the methanogenic community changed in reactor R2, possibly influenced by differences in characteristics of the PWASS over time.

The introduction of OFMSW as a co-substrate and corresponding changes in the operational conditions seemed to favor the turnover kinetics of the degradation intermediate products in R1 (Fig. 2c, d). The rates of methane production from acetate, propionate, and oleate were higher in batch assays inoculated by sludge from R1 after initiating the co-digestion of OFMSW (Fig. 2c). However, this was not observed for phenyl acetate (Fig. 2c). The results are in line with other studies reporting

that co-digestion with sewage sludge often accelerates the digestion process and increases the biogas yield (Edelmann et al. 2000; Gupta et al. 2012; Aichinger et al. 2015). In addition, higher occurrence of the hydrogenotrophic methanogens in R1 compared to R2 (cf. Fig. 2) may have promoted the syntrophic interactions between hydrogen-generating bacteria and hydrogen-utilizing methanogens, which would in turn facilitate the turnover of intermediate degradation products such as VFA and LCFA.

Estimated potential for increased biogas production at WWTP in Sweden

In Sweden, 38% (472,130 tons) of the OFMSW, produced during 2015, was treated biologically (Swedish Waste Management Association 2016). The total Swedish OFMSW is thus around 1,240,000 tons, and if all these wastes were digested, it would give around 1.24 TWh biogas (Jörgen Ejlerstsson, Scandinavian Biogas Fuels AB, pers. comm.). Our study shows that the OLR of an anaerobic digestion process treating PWASS can be increased 3.3 times by co-digestion with OFMSW, which enables a fourfold increase of the biogas production. The total volume of digesters at Swedish wastewater treatment plants is 339,000 m³ volume (ES 2015:03; ISSN 1654-7543), and if the maximum load of OFMSW wet weight is set to 16 kg/m³ and day

(corresponding to 3.5 kgVS m^{-3} and day^{-1} at a TS of 25% and a VS of TS of 90%), this would require $218,000 \text{ m}^3$ of the digester volume, meaning that there is capacity for digesting all available OFMSW still leaving room for additional substrate from, e.g., industries, agricultural activities, etc.

Conclusions

The results clearly demonstrated that co-digestion of OFMSW with sewage sludge in municipal anaerobic digesters is favorable. Co-digestion of OFMSW with PWASS at a 3:1 ratio on the basis of VS content and at an OLR of $5.0 \text{ gVS l}^{-1} \text{ day}^{-1}$ has the potential to increase the biogas production approximately four times, without disturbing the process performance. Furthermore, co-digestion of OFMSW with PWASS seemed to favor a stable methanogenic community composition and increase turnover kinetics of the degradation intermediate products (acetate, propionate, and oleate) in the digester. Practical limitations potentially include the need for sludge dewatering to maintain a sufficient HRT (17 days in this study) at elevated OFMSW-loading rates, as well as additional energy consumption for mixing due to an increased sludge viscosity.

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Compliance with ethical standards

Conflict of interest On behalf of all authors, the corresponding author states that there is no conflict of interest.

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