Long term operation of a thermophilic anaerobic reactor: process stability and efficiency at decreasing sludge retention time

Ivet Ferrer[a,b,*], Felícitas Vázquez[c] and Xavier Font[d]

[a] Environmental Engineering Division
Department of Hydraulic, Maritime and Environmental Engineering
Technical University of Catalonia (UPC)
c/ Jordi Girona 1-3, E-08034 Barcelona, Spain
E-mail address: ivet.ferrer@upc.edu

[b] GIRO Technological Centre
Rambla Pompeu Fabra 1, 08100 Mollet del Vallès, Barcelona, Spain

[c] Institute of Biotechnology and Biomedicine
Universitat Autònoma de Barcelona
Bellaterra (Cerdanyola del Vallès, 08193-Barcelona, Spain)
E-mail address: felicitas.vazquez@uab.cat

[d] Composting Research Group
Department of Chemical Engineering
Escola Tècnica Superior d'Enginyeria
Universitat Autònoma de Barcelona
Bellaterra (Cerdanyola del Vallès, 08193-Barcelona, Spain)
E-mail address: xavier.font@uab.cat

* Corresponding author:
Tel.: (+34) 93 4016463
Fax: (+34) 93 4016579
E-mail: ivet.ferrer@upc.edu

Abstract

The aim of this study was to evaluate the performance of thermophilic sludge digestion at decreasing sludge retention time (SRT) and increasing organic loading rate (OLR), in terms of methane production, effluent stabilisation, hygienisation and dewaterability. Focus was put on determining indicators to help prevent process failure. To this end, a lab-scale reactor was operated for nearly two years at 55 ºC. Methane production rate was increased (from 0.2 to 0.4-0.6 m^3_{CH4} $m^{-3}_{reactor}$ d^{-1}) by decreasing the SRT from 30 to 15-10 days, while increasing the OLR from 0.5 to 2.5-3.5 kg VS $m^{-3}_{reactor}$ d^{-1}. Sludge dewaterability was worsened at SRT below 15 days; while pathogen destruction was always successful. The following concentrations might be used to prevent process failure: VFA C2-C5 (3.7 g COD L^{-1}), acetate (0.6 g L^{-1}), acetate/propionate (0.5), intermediate alkalinity (1.8 g $CaCO_3$ L^{-1}), intermediate/partial alkalinity (0.9), intermediate /total alkalinity (0.5), CH_4 in biogas (55 %).

Keywords: Biogas; Biosolids; Dewaterability; Hygienisation; Wastewater

1. Introduction

In anaerobic digesters, biogas production depends on the amount of organic matter biodegraded by anaerobic microorganisms. Thus, it depends on the composition of the substrate, and presence and equilibrium between anaerobic consortia in the reactor. Design and operation parameters of the process include sludge retention time (SRT), organic loading rate (OLR), temperature and reactor flow, amongst others.

Sludge hydrolysis is often regarded as the rate limiting stage of the overall process (Vavilin *et al*., 2007); it affects the total amount of solids converted into soluble compounds and ultimately to biogas. However, soluble substrates utilization rates for fermentation and methanogenesis play a key role on process stability. The concentration of intermediate products like volatile fatty acids (VFA) is a common indicator of process unbalance (Marchaim and Krause, 1993; Pind *et al*. 2002). An accumulation of VFA in the digester may result from problems in the synthrophic bacterial relationships between the H_2-producing and the H_2-consuming bacteria; insufficient methanogenic population to utilize all VFA produced or insufficient retention time for this process to take place.

Since growth rates of methanogenic archaea are lower than those of fermentative bacteria, they determine the minimum (or washout) SRT for the process. At 20, 25 and 35 ºC the washout SRT are 7.8, 5.9 and 3.2 days, respectively, which turn into 40, 30 and 15 days design values by taking a safety factor of 5 for suspended growth processes (Metcalf and Eddy, 2003). Because growth rates of thermophilic methanogens are 2-3 times higher than those of mesophilic homologues (Van Lier *et al.*, 1993), the minimum and design SRT would be in the range of 1-2 and 5-8 days, respectively.

Process temperature not only affects the reaction rate and required SRT to achieve a certain process efficiency (i.e. solids removal and methane production), but also plays a key role in the stability of the process. Methanogenic archaea are especially sensitive to temperature fluctuations, even to changes around 1 ºC d^{-1} (Metcalf and Eddy, 2003). This can be particularly critical for thermophilic processes, since they are reported to be less stable than mesophilic ones (Buhr and Andrews, 1977). For this reason, a number of studies have focused on the effect of temperature fluctuations on thermophilic anaerobic digestion (Van Lier *et al.*, 1993; Ahring *et al.,* 2001; El-Mashad *et al.*, 2004).

Both temperature and SRT have direct influence on treatment costs, with respect to initial capital investment (i.e. digester volume depends on the SRT), as well as operation and maintenance costs (i.e. digester heating, mixing and pumping). Hence, interest has also been put on studying the effect of the SRT on process performance (Lin *et al.*, 1986; Zhang and Noike, 1994; Miron *et al.*, 2000; De la Rubia *et al.*, 2006a; De la Rubia *et al.*, 2006b; Ponsá *et al.*, 2008). From an economical point of view, it would be most effective to operate at a minimum SRT allowing optimising methane production and solids removal, whilst assuring process stability.

Considering the whole sludge treatment line in wastewater treatment plants (WWTP), sludge stabilisation in anaerobic digesters is followed by sludge conditioning and dewatering steps. Since solids dewatering may account for some 7 % of the energy requirements in conventional activated sludge WWTP (Metcalf and Eddy, 2003), the reduction of dewatering costs by enhancing sludge dewaterability is of major importance. However, from the literature it is not clear whether the anaerobic process improves or degrades sludge dewaterability; and how mesophilic and thermophilic effluents compare in terms of dewaterability is not clear either (Houghton *et al.*, 2000; Houghton and Stephenson, 2002; Neyens and Baeyens, 2002; Novak *et al.*, 2003).

According to the 3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000), thermophilic digestion should enable effluent hygienisation for its use on land, which is strongly recommended whenever it is possible to recycle the nutrients contained in the sludge, improving soil fertility and minimising the amount of waste going to incineration or landfill. Consequently, there has been a growing interest upon this technology.

The aim of this study was to assess the impact of SRT and OLR on the anaerobic thermophilic digestion of sewage sludge. Process performance was monitored at decreasing SRT, while the influence of the solid content in the feed sludge, hence the OLR and its variability, were evaluated. The combined effect of all these process parameters on biogas and methane production, as well as effluent stabilisation, hygienisation and dewaterability, were assessed. Focus was put on seeking alert values for parameters which may be used to prevent process failure.

2. Materials and methods

2.1 Sewage sludge

The sludge used for this work was obtained from two municipal WWTP near Barcelona (Spain), which serve an equivalent population of 130,000 equivalent inhabitants (EI) each. The conventional wastewater treatment used in these plants consists of preliminary and primary treatment and secondary treatment in the activated sludge unit. Primary sludge (PS) and secondary waste activated sludge (WAS) are thickened and mixed (sampling point), before undergoing mesophilic (38 ºC) anaerobic digestion at 40 days SRT. Finally, digested sludge is dewatered in a centrifuge.

The inoculum used to seed the digester was mesophilic digested sludge (since no thermophilic sludge digestion plant operates in the Barcelona area). The substrate was the mixture of thickened PS and WAS (75 / 25 % v/v), which was collected weekly and stored at 4 ºC until use. Low-solids sludge (total solids (TS) < 40 g L^{-1}) was used for the first 14 months; whereas high-solids sludge (TS > 40 g L^{-1}) was used thereafter. Volatile solids (VS) contents were 14-24 g L^{-1} (68-77 % VS/TS) and 30-35 g L^{-1} (58-75 % VS/TS) in the low-solids and high-solids sludge, respectively. In general, the values are typical of sludge from conventional activated sludge WWTP entering digestion, with TS below 50 g L^{-1} and VS/TS around 70 % (Speece, 1988).

2.2 Experimental set-up

The experimental set-up used in this work consists of a 5 L continuous stirred tank reactor (CSTR) with automated semi-continuous feeding, temperature control (35-55 ºC) and on-line biogas measurement. It is described in detail elsewhere (Ferrer *et al.*, 2008).

2.3 Experimental procedures

The reactor was seeded with 5 L of mesophilic digested sludge from a full-scale reactor. The temperature was switched from 38 to 55 ºC in a single step, while stopping the organic loading for a few days. The OLR was thereafter increased by decreasing the SRT to 30, 25, 20, 15, 12.5, 10, 8, 7 and 6 days. The OLR was also increased by changing from low-solids to high-solids sludge. Each subsequent SRT decrease was made once the digester had reached stable operation (i.e. fairly constant performance in terms of biogas production, VFA concentration and pH in the reactor) as proposed by other authors (Angelidaki and Ahring, 1994; El-Mashad *et al*., 2004). This digester was operated for 21 months, under the conditions summarised in Table 1.

2.4 Analytical methods

The solids content of sludge was determined according to the Standard Methods procedure 2540G (APHA, 1999). TS and VS were determined directly from sludge samples, whereas total dissolved solids (TDS) and volatile dissolved solids (VDS) were determined from the supernatant of samples centrifuged at 7000 rpm. Supernatants underwent filtration through 1.2 µm nominal pore size glass fibber filters (Albet FVC047, Spain). The particulate fractions, total suspended solids (TSS) and volatile suspended solids (VSS) were subsequently deduced. pH, alkalinity and VFA (acetic, propionic, iso-butyric, n-butyric, iso-valeric and n-valeric acids) were also analysed from the filtrate supernatant. Samples for VFA analysis were further filtered through a 0.45 µm nylon syringe filter. VFA and biogas composition were determined by gas chromatography (Perkin-Elmer AutoSystem XL Gas Chromatograph), as described in Ferrer *et al*. (2008).

Total, partial and intermediate alkalinities were determined as proposed by Ripley *et al.* (1986). The method consists of a two step titration: a first one down to pH 5.75, which is due to HCO_3^- species and is known as partial alkalinity (PA); and a second one down to pH 4.3, which corresponds to the total alkalinity (TA). The intermediate alkalinity (IA), which is related to VFA concentration, is then estimated as the difference between TA and PA. It can be used as an indirect measurement of VFA concentration. The alkalinity ratio (AR), defined as the ratio between intermediate and total alkalinity (IA/TA), or between intermediate and partial alkalinity (IA/PA); may also be a useful indicator of the concentration of VFA in the sample.

Sludge dewaterability was determined using the Capillary Suction Time (CST) test, according to the Standard Methods procedure 2710G (APHA, 1999). The CST used was a Triton CST filterability tester, model 200, Triton Electronics Ltd., Essex, UK. Standard filter papers (Part No. 815095) were supplied by Triton Electronics.

Sludge hygienisation was evaluated by the concentration of *Escherichia coli* and *Salmonella sp*. in digested effluents. *E. coli* were quantified by the methodology ISO 16649:2000 and the results were expressed as colony forming units per mL ($CFU \cdot mL^{-1}$). In the case of *Salmonella sp*., only presence or absence was determined by the methodology NF-V08-052 and the results were presence or absence per 50 mL of sample.

3. Results and discussion

3.1 Thermophilic anaerobic sludge digestion at decreasing SRT

Process performance during the long term operation of the reactor (654 days) is illustrated in Figures 1 and 2. Mean values of operating and efficiency parameters during stable periods under each condition assayed are summarised in Table 2.

The process was start-up by seeding the digester with mesophilic sludge and rising process temperature from 38 to 55º C in a single-step. Working at 30 days SRT, the OLR reached 0.69 kg VS $m^{-3}_{reactor}$ d^{-1}, leading to methane production rates around 0.22 m^3_{CH4} $m^{-3}_{reactor}$ d^{-1} and 40-50 % VS destruction (Table 2, period II). Such values are in the range of those reported in the literature for thermophilic digestion of sewage sludge at high SRT. For instance, De la Rubia *et al.* (2006a) obtained around 0.19 m^3_{CH4} $m^{-3}_{reactor}$ d^{-1} and 53 % VS removal working at 27 days SRT.

The SRT was subsequently reduced to 25 and 20 days (OLR ~ 1 kg VS $m^{-3}_{reactor}$ d^{-1}), leading to biogas production rates between 0.35-0.42 m^3_{biogas} $m^{-3}_{reactor}$ d^{-1}, with 62-68 % CH_4 in biogas (Table 2, periods III and IV). VS destruction was lower at 20 days SRT (40 % vs. 53 %) due to fluctuations in influent VS concentration. Other authors have obtained similar results at 20 days SRT (~ 0.4 m^3_{biogas} $m^{-3}_{reactor}$ d^{-1}, 60-65 % CH_4 in biogas, ~ 53 % VS destruction) (De la Rubia *et al.*, 2002; Gavala *et al.*, 2003).

The best results were obtained at the lowest SRTs (15 and 10 days), with OLR of 1-1.6 and 1.5-2 kg VS $m^{-3}_{reactor}$ d^{-1}, respectively. In particular, the highest biogas and methane production rates (up to 0.56 and 0.36 m^3 m^{-3} d^{-1}, respectively) correspond to 10 days SRT (Table 2, period VI).

After switching from low-solids to high-solids sludge (40-60 g TS L^{-1}; 30-35 g VS L^{-1}), OLR as high as 3-4 kg VS $m^{-3}_{reactor}$ d^{-1} were maintained (Figure 1). Biogas production rate was almost doubled from 0.5 to 1 m^3_{biogas} $m^{-3}_{reactor}$ d^{-1} at 10 days SRT feeding low- and high-solids sludge, respectively (Table 2, periods VI and VII). However, higher effluent VFA (> 4 g COD L^{-1}) were detected (Figure 3(a)).

The SRT was gradually decreased to 6 days with OLR ranging from 4.5 to 6.5 kg VS $m^{-3}_{reactor}$ d^{-1} (Figure 1), which are amongst the highest OLR and lowest SRT reported for single stage sludge digestion (Buhr and Andrews, 1977; Speece, 1988; De la Rubia *et al.,* 2006a; De la Rubia *et al.,* 2006b). Initially, biogas production reached its highest rates (~ 1.5 m^3_{biogas} $m^{-3}_{reactor}$ d^{-1}), with 58-69 % CH_4 in biogas. However, these operating conditions prompted VFA accumulation (VFA C2-C5 increase from 4 to 10 g

COD L^{-1}), as shown in Figure 3(a). Methane content in biogas drop below 50 % (Figure 2) and VS removal to 13 %. To avoid digester failure, the SRT was set back to 10 days.

3.2 Process stability

During almost two years of experimental work, process stability was disturbed whenever the OLR increased, either as a result of decreasing the SRT or due to fluctuations in the solids content of feed sludge. Additionally, the process was unsteady after temperature fluctuations episodes (caused by occasional operating problems), especially when they happened together with organic overloading. In all cases, the immediate response of the system was a decrease in methane content in biogas from around 60 % to below 50 % and VFA accumulation (Figures 2-3) as a result of decreased methanogenic activity.

Based on this study, limit concentrations to detect and prevent digester failure during thermophilic sludge digestion are proposed (Table 3) and discussed a follows.

3.2.1 Volatile fatty acids

Although the concentration of all VFA increased during the instability episodes, the rise in acetate concentration was perhaps the most accentuated. Throughout the whole experimental period, acetate fluctuated within a wider range of concentrations, compared to other major VFA like propionate, iso-butyrate and iso-valerate. Figure 3(b) shows that these three VFA followed parallel trends, propionate concentration always being the highest. On the other hand, acetate concentration ranged from almost 0 to nearly 1 g L^{-1}. This clearly indicates that occasional temperature fluctuations and organic overloading affected methanogens to a higher extent than acidogens, with subsequent accumulations of acetate in the liquor. Since changes in propionate concentration were less pronounced, the trend followed by the acetate to propionate ratio (A/P ratio) was similar to that of acetate, as can be seen from Figure 3(b).

As well as individual and total VFA, some authors have proposed acetate concentration and A/P ratio as valuable indicators to predict process failure (Marchaim and Krause, 1993; Pind *et al*., 2002). For manure, an acetic acid concentration of 0.8 g L^{-1} and an A/P ratio of 1.4 have been proposed as limit values (Hill *et al.,* 1987; cited in Marchaim and Krause, 1993). To our knowledge, such limit values for thermophilic sewage sludge digestion have not yet been proposed. In the present study, acetate concentration was usually below 0.6 g L^{-1} (Table 2, all periods) and only in cases of organic overloading or temperature fluctuations (due to operating problems) did this value rise above 0.6 g L^{-1} and up to 2 g L^{-1}. Furthermore, concentrations above 1 g L^{-1} were only reached when the SRT was reduced to 6 days, with OLR greater than 5 kg VS $m^{-3}_{reactor}$ d^{-1}, as shown in Figure 3. Therefore, a limit concentration of 0.6 g L^{-1} of acetic acid would seem more appropriate to predict digester failure in the case of thermophilic sludge digestion. Similarly, during stability periods the A/P ratio was below 0.5 (Table 2, all periods); hence the limit A/P ratio to predict digester failure ought to be reduced to around 0.5. From our experimental results, it might be hypothesized that the total VFA (C2-C5) concentration corresponding to these values would be around 3.7 g COD L^{-1}, depending on the individual VFA concentration. Such a high concentration would be detrimental to the quality of the effluent sludge, meaning that subsequent post-treatments ought to be considered.

3.2.2 Alkalinity

According to Ripley *et al.* (1986), the total alkalinity (TA) of a sample is a result of HCO_3^- species, which is known as partial alkalinity (PA); and VFA, which is known as intermediate alkalinity (IA). The latter is estimated as the difference between the TA and PA. For this reason, the IA consists of an indirect measurement of VFA, and the alkalinity ratios between intermediate and total (IA/TA) or partial (IA/PA) alkalinities are alternative process indicators. In the present study, the profile of the IA/PA ratio was indeed very similar to that of total VFA, acetate concentration and A/P ratio in Figure 3; while variations in the IA/TA ratio were less pronounced. In general, the IA/PA ratio was more sensible to variations in the VFA concentration than the IA/TA.

The correlation between total VFA (C2-C5) concentration, acetate concentration or A/P ratio; and alkalinity ratios or intermediate alkalinity was further analysed (Figure 4). Obviously, the best correlated parameter was intermediate alkalinity, followed by IA/PA and IA/TA ratios. The best correlations were obtained with respect to total VFA concentration ($R^2 \leq 0.79$); while the correlations with acetate concentration were very poor ($R^2 \leq 0.65$) and no correlations were found with the A/P ratio ($R^2 \sim 0$).

If threshold values were to be set in order to predict process failure based on alkalinity measurements (which is common practise at industrial scale); the values corresponding to the aforementioned VFA C2-C5 concentration of 3.7 g COD L^{-1} would be around 0.9 for IA/PA ratio, 0.5 for IA/TA ratio and 1.8 g $CaCO_3$ L^{-1} for intermediate alkalinity.

3.2.3 Methane content in biogas

With regards to the methane content in biogas, during stable periods this value always ranged between 60-70 % (Table 2, all periods), which is typically reported in the literature for thermophilic sludge digestion (Krugel *et al*. 1998; Zábranská *et al*. 2000; De la Rubia *et al.* 2006a; De la Rubia *et al.* 2006b; Ferrer *et al*., 2008; Palatsi *et al*., 2009). It only fell below 55 % in cases of organic overloading or temperature fluctuation, which suggests a warning concentration of 55 % for thermophilic sludge digestion. It should be noticed that such a value would be within the common range for other processes; for instance in digesters treating the organic fraction of municipal solid wastes methane content in biogas ranges from 50-60 %.

3.2.4. pH

In terms of pH, this parameter was fairly constant and remarkably high (around 8). Even working at 6 days SRT, with the highest OLR (> 5 kg VS $m^{-3}_{reactor}$ d^{-1}), when all other indicator parameters were above the limit values proposed, the pH was still 8. The reason for this is that the alkalinity of the system was also the highest; hence the buffer capacity of the system prevented from pH drop resulting from VFA accumulation. In sewage sludge digesters, sufficient alkalinity is generally found (3-5 g $CaCO_3$ L^{-1}) to prevent the pH from falling below the limit for methanogenesis inhibition (Metcalf and Eddy, 2003). Studies with high-solids sludge (4-10 % TS) have shown that the optimum pH range for high rate digestion is 6.6-7.8, while the acceptable pH range is 6.1-8.3; meaning that below 6.1 the process may fail due to an excessively low methanogenesis rate compared to acidogenesis rate, while above 8.3 the process might be inhibited by free ammonia (Lay *et al.,* 1997). Ammonia inhibition is favoured by high process temperature (Angelidaki and Ahring, 1994) and is pointed out as a major cause for low biogas production treating pig slurries (Bonmatí and Flotats, 2003).

3.3. Effect of SRT and OLR on process efficiency and stability

The main objective of decreasing the SRT was to determine the minimum SRT allowing a stable anaerobic process performance at 55 ºC. Bearing in mind that the minimum design SRT is around 15 days at 35 ºC (Metcalf and Eddy, 2003), and that the growth rates of thermophilic methanogens are 2-3 times higher than those of mesophilic homologues, (Van Lier *et al.*, 1993), the theoretical SRT may be reduced to 5-8 days at 55 ºC. However, such a reduction is likely to deteriorate process efficiency, especially regarding the quality of the effluent which is generally poorer in thermophilic digesters (Buhr and Andrews, 1977). Digested sludge dewaterability might consequently be degraded.

For the purposes of this study, the SRT was gradually reduced from 30 to 6 days. However, because the feeding sludge was collected weekly from the WWTP, seasonal variations and operational changes affected its composition and organic content. Furthermore, low-solids and high-solids sludge were used. Whilst operating under a fixed SRT, the OLR was affected by the sludge organic content; thus it was also necessary to assess the effect of OLR on the thermophilic sludge digestion.

Figure 5 shows methane production rate, effluent VFA and effluent VS as a function of the OLR. In general, high correlations were obtained for methane production rate and VFA (R^2=0.96). This means that daily methane production, hence methanogenic activity, was very much dependant on the OLR, regardless of the SRT. Similarly, acidogenesis increased with the OLR (Figure 5), but short SRT were not enough to convert all VFA to methane, which means that a portion of hydrolysed organic compounds did not end up yielding methane.

De la Rubia *et al.* (2006a) found a similar dependence of methane production rate on OLR and SRT over the range of 15-75 days during thermophilic anaerobic digestion of PS and WAS. COD mass balances indicated that the amount of COD used for methane generation increased at decreasing SRT or increasing OLR. The results obtained by these authors suggest that higher OLR (> 2.2 kg VS m^{-3} d^{-1}) or lower SRT (< 15 days) might have resulted in further methane production improvement (> 0.4 m^3_{CH4} m^{-3} d^{-1}).

Miron *et al.* (2000) reported that, during psychrophilic digestion of PS, SRT of 10 days were enough to obtain methanogenic conditions in the reactor, while lower SRT (8 days) resulted in acidogenic conditions. Taking into account that reaction rates are higher under thermophilic conditions, it might be speculated that the homologues SRT for a thermophilic process would be lower.

In the present study, the minimum SRT assayed was 6 days, but the minimum SRT ensuring a stable performance was also 10 days. Methane production under thermophilic conditions was improved by decreasing the SRT from 30 to 10 days. It was further enhanced at 6 days SRT with an OLR higher than 5 kg VS m^{-3} d^{-1}, feeding high-solids sludge. However, when the OLR eventually increased (> 6 kg VS m^{-3} d^{-1}) as a result of fluctuations in the solids content of the feed sludge, methanogenic activity was severely affected; as indicated by decreased biogas production, with methane content below 50 %, and a sudden accumulation of VFA, with a total concentration higher than 6 g L^{-1}. Furthermore, the quality of the effluent in terms of VS content was worsened.

On the other hand, working at SRT of 10 days still with high OLR (3-4 kg VS $m^{-3}_{reactor}$ d^{-1}), the process was more stable. Biogas and methane production rates (0.55-0.6 and 0.35-0.4 m^3 $m^{-3}_{reactor}$ d^{-1}) were increased by 50 % compared previous results at higher SRT. Gas production at 10 days SRT was within the range obtained by other authors at 15 days SRT (De la Rubia *et al*., 2006a; Benabdallah *et al.,* 2006); but clearly higher than that obtained at 20 days SRT (De la Rubia *et al*., 2002; Gavala *et al*., 2003). In practise, this means that the sludge daily flow rate could be doubled or the digester volume reduced, while producing the same amount of methane (i.e. energy). However, higher effluent VS and especially higher VFA, ought to be expected at this reduced SRT; which might deteriorate subsequent sludge dewatering.

3.4 Sludge dewaterability

Sludge dewaterability was measured by determining the capillary suction time (CST) of digested sludge samples obtained during each stability period. Figure 6(a) shows that CST values increased proportionally to the OLR (R^2=0.92). The trends are similar when the CST is expressed per g TS or g VS.

A clear dependence of CST on the solids concentration in the sludge is shown in Figure 6(b): the higher the solids concentration, the higher the CST. Hence, it may be speculated that any increase in effluent VS and TS resulting from changing the OLR and/or SRT may ultimately affect digested sludge dewaterability. From the results of this study, it seems that digested sludge dewaterability was deteriorated with TS higher than 26 g L^{-1} and VS higher than 17 g L^{-1}; which corresponded to OLR above 3 kg VS $m^{-3}_{reactor}$ d^{-1} and SRT below 10 days.

According to the work by Miron *et al.* (2000), the dewaterability of PS worsened under acidogenic conditions (SRT ≤ 8 days), while it improved under methanogenic conditions (SRT ≥ 10 days). This was related to a decrease in the mean particle size, thus an increase in the total surface area, under acidogenic conditions. Moreover, only at high SRT of 15 days was digested sludge dewaterability improved compared to that of influent sludge. The results of the present study are quite consistent with those findings, since only at SRT above 15 days was the CST value (60-160 s) below that of influent sludge (437 s). Sludge dewaterability was worsened (CST ~ 630-1370 s) at shorter SRT (10-6 days), which were typically associated to higher effluent VFA, thus higher soluble VS. Indeed, an increasing trend was followed by CST with respect to effluent VFA (Figure 6(c)).

Some controversy exists in the literature regarding the effect of anaerobic digestion on sludge dewaterability, and it is still not clear whether mesophilic and thermophilic digestion has any effect in sludge dewaterability. It has been shown that sludge dewaterability, as well as the amount of chemicals required for sludge conditioning, are directly dependant on the concentration of biopolymer in the solution (Novak *et al*., 2003). Houghton *et al*. (2000) and Houghton and Stephenson (2002) reported that the composition of microbial extracellular polymer (ECP) varied after sludge digestion and was also affected by the feed composition; attributing excess ECP production to acidogenic bacteria. This might also explain higher CST values obtained in the present study in samples with higher VFA concentration, in which the presence of acidogenic bacteria should be higher.

3.5. Effluent hygienisation

Sludge hygienisation was assessed by quantifying pathogen indicators *Escherichia coli* and *Salmonella* spp. from digested sludge samples obtained during each stability period, and comparing them to the values obtained from influent sludge samples. While *Salmonella* spp. was never detected; the concentration of *E. coli* in the influent sludge was in the range of 10^6 CFU mL^{-1}. A complete destruction of *E. coli* was achieved at SRT higher than 20 days, but concentrations in the range of 10^1 and 10^2 CFU mL^{-1} were found at SRT of 10-15 days and 6 days, respectively (Table 4). *E. co*li concentration in the effluent seemed to be depended on the OLR hence, on the influent characteristics.

Hygienisation of thermophilic effluent sludge in laboratory and full-scale reactors working at a range of SRT is reported in the literature (Zábranská *et al*., 2000; Laffite-Trouqué and Forster, 2002; Lu *et al.,* 2007). It is in fact a major advantage of thermophilic anaerobic digestion, compared to mesophilic operation. In this study, *E. coli* and *Salmonella* spp. concentrations in all effluent samples were below the limits proposed in the 3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000) for unrestricted land application of digested sludge; which suggests that a minimum SRT of 6 days at 55 ºC might be sufficient to prevent the spread of pathogens in the environment upon land application of digestates.

4. Conclusions

This long term study showed that the minimum SRT for a stable thermophilic sludge digestion was 10 days. Methane production was increased to 0.4-0.6 $m^3_{CH4}\ m^{-3}_{reactor}\ d^{-1}$ by decreasing the SRT to 15-10 days, but VS removal and sludge dewaterability were worsened below 15 days SRT, with high effluent VFA. Besides, the concentrations of pathogens were always below the limits proposed for unrestricted land application. The following indicators may be useful to prevent digester failure: VFA C2-C5 (3.7 g COD L^{-1}), acetate (0.6 g L^{-1}), A/P (0.5), IA (1.8 g $CaCO_3$ L^{-1}), IA/PA (0.9), IA/TA (0.5), CH_4 in biogas (55 %).

Acknowledgements

The authors wish to thank the financial support provided by the Spanish Ministry of Science and Technology and FEDER (REN2002-00926/TECNO). Eva Romero from GIRO Technological Centre is kindly acknowledged for her contribution.

References

Angelidaki, I., Ahring B.K., 1994. Anaerobic thermophilic digestion of manure at different ammonia loads: effect of temperature. Water Research 28(3), 727-731.

Ahring, B.K., Ibrahim, A.A., Mladenovska, Z., 2001. Effect of temperature increase from 55 to 65 ºC on performance and microbial population dynamics of an anaerobic reactor treating cattle manure. Water Research 35(10), 2446-2452.

APHA, 1999. Standard Methods for the Examination of Water and Wastewater 20th edition, American Public Health Association/American Water Works Association/Water Environment Federation, Washington DC, USA.

Benabdallah, El-Hadj T., Dosta, J., Márquez-Serrano, J., Mata-Álvarez, J., 2006. Effect of ultrasound pretreatment in mesophilic and thermophilic anaerobic digestion with emphasis to naphthaleneand pyrene removal. Water Research 41(1), 87-94.

Bonmatí, A., Flotats, X., 2003. Air stripping of ammonia from pig slurry:

characterisation and feasibility as a pre- or post-treatment to mesophilic anaerobic digestion. Waste Management 23, 261-272.

Buhr, H.O., Andrews, J.F., 1977. The thermophilic anaerobic digestion process. Water Research 11, 129-143.

De la Rubia, M.A., Perez, M., Romero, L.I., Sales, D., 2002. Anaerobic mesophilic and thermophilic municipal sludge digestion. Chemical and Biochemical Engineering Quarterly 16(3), 119-124.

De la Rubia, M.A., Perez, M., Romero, L.I., Sales, D., 2006a. Effect of solids retention time (SRT) on pilot scale anaerobic thermophilic sludge digestion. Process Biochemistry 41, 79-86.

De la Rubia, M.A., Romero, L.I., Sales, D., Perez, M., 2006b. Pilot-scale anaerobic thermophilic digester treating municipal sludge. AIChE Journal 52(1), 402-407.

El-Mashad, H.M., Zeeman, G., Van Loon, W.K.P., Bot, G.P.A., Lettinga, G., 2004. Effect of temperature and temperature fluctuations on thermophilic anaerobic digestion of cattle manure. Bioresource Technology 95, 191-201.

Environment DG, EU., 2000. Working Document on Sludge 3rd Draft. URL: http://ec.europa.eu/environment/waste/sludge/pdf/sludge_en.pdf (July 2003)

Ferrer, I., Ponsá, S., Vázquez, F., Font, X., 2008. Increasing biogas production by thermal (70 ºC) sludge pre-treatment prior to thermophilic anaerobic digestion. Biochemical Engineering Journal 42(2), 186-192.

Gavala, H.N., Yenal, U., Skiadas, I.V., Westermann, P. Ahring, B.K., 2003. Mesophilic and thermophilic anaerobic digestion of primary and secondary sludge. Effect of pre-treatment at elevated temperature. Water Research 37, 4561-4572.

Hill, D.T., Cobb, S.A., Bolte, J.P., 1987. Using volatile fatty acid relationships to predict anaerobic digester failure. Transactions of the ASAE 30(2), 486-501.

Houghton, J.I., Quarmby, J., Stephenson, T., 2000. The impact of digestion on sludge dewaterability. Trans IChemE 78 B, 153-159.

Houghton, J.I., Stephenson, T., 2002. Effect of influent organic content on digested sludge extracellular polymer content and dewaterability. Water Research 36, 3620-3628.

Krugel, S., Nemeth, L., Peddie, C., 1998. Extending thermophilic anaerobic digestion for producing Class A biosolids at the Greater Vancouver Regional District Annacis Island wastewater treatment plant. Water Science and Technology 38, 409-416.

Lafitte-Trouqué, S., Forster, C.F., 2002. The use of ultrasound and γ-irradiation as pretreatments for the anaerobic digestion of waste activated sludge at mesophilic and thermophilic temperatures. Bioresource Technology 84, 113-118.

Lay, J-J., Li, Y-Y., Noike, T., 1997. Influences of pH and moisture content on the methane production in high-solids sludge digestion. Water Research 31(6), 1518-1524.

Lin, C-Y., Sato, K., Noike, T., Matsumoto, J., 1986. Methanogenic digestion using mixed substrate of acetic, propionic and butyric acids. Water Research 20(3), 385-394.

Lu, J., Gavala, H.N., Skiadas, I.V., Mladenovska, Z., Ahring, B.K., 2007. Improving anaerobic sewage sludge digestion by implementation of a hyperthermophilic prehydrolisis step. Journal of Environmental Management 88(4), 881-889.

Marchaim, U., Krause, C., 1993. Propionic to acetic ratios in overloaded anaerobic digestion. Bioresource Technology 43, 195-203.

Metcalf and Eddy, 2003. Wastewater Engineering: Treatment and Reuse. 4th edition, Mc Graw-Hill Higher Education, Boston, USA.

Miron, Y., Zeeman, G., Van Lier, J.B., Lettinga, G., 2000. The role of sludge retention time in the hydrolysis and acidification of lipids, carbohydrates and proteins during digestion of primary sludge in CSTR systems. Water Research 34(5), 1705-1713.

Neyens, E., Baeyens, J., 2002. A review of thermal sludge pretreatment processes to improve dewaterability. Journal of Hazardous Materials 98(1-3), 51-67.

Novak, J.T., Sadler, M.E., Murthy, S.N., 2003. Mechanisms of floc destruction during anaerobic and aerobic sludge digestion and the effect on conditioning and dewatering biosolids. Water Research 37(13), 3136-3144.

Palatsi, J., Gimenez-Lorang, A., Ferrer, I., Flotats, X., 2009. Start-up strategies of thermophilic anaerobic digestion of sewage sludge. Water Science and Technology 59(9), 1777-1784.

Pind, P.F., Angelidaki, I., Ahring, B.K., 2002. Dynamics of the anaerobic process: effects of volatile fatty acids. Biotechnology and Bioengineering 82(7), 791-801.

Ponsá, S., Ferrer, I., Vázquez, F., Font, X., 2008. Optimization of the hydrolitic-acidogenic anaerobic digestion stage (55 ºC) of sewage sludge: influence of pH and solid content. Water Research 42, 3972-3980.

Ripley, L.E., Boyle, W.C., Converse, J.C., 1986. Improved alkalimetric monitoring for anaerobic digestion of high-strength wastes. Journal of the Water Pollution Control Federation 58(5), 406-411.

Speece, R.E., 1988. A survey of municipal anaerobic sludge digesters and diagnostic activity assays. Water Research 22 (3), 365-372.

Van Lier, J.B., Hulsbeek, J., Stams, A.J., Lettinga, G., 1993. Temperature susceptibility of thermophilic methanogenic sludge: implications for reactor start-up and operation. Bioresource Technology 43, 227-235.

Vavilin, V.A., Fernandez, B., Palatsi, J., Flotats, X., 2007. Hydrolysis kinetics in anaerobic degradation of particulate organic material: an overview. Waste Management 28(6), 939-951.

Zábranská, J., Dohányos, M., Jeníček, P., Kutil, J., 2000. Thermophilic process enhancement of excess activated sludge degradability – two ways of intensification sludge treatment in Prague central wastewater treatment plant. Water Science and Technology 41(9), 265-272.

Zhang T.C., Noike, T., 1994. Influence of retention time on reactor performance and bacterial trophic populations in anaerobic digestion processes. Water Research 28(1), 27-36.

Table 1. Operating conditions

Period	Time (days)	Temperature (ºC)	SRT (d)	Solids content in feed sludge *
I	1-79	55	> 30	low-solids
II	80-161	55	30	low-solids
III	162-203	55	25	low-solids
IV	204-256	55	20	low-solids
V	257-331	55	15	low-solids
VI	332-437	55	10	low-solids
VII	484-529	55	10	high-solids
VIII	569-606	55	6	high-solids
IX	607-653	55	10	high-solids

* low-solids: total solids < 4 %; high-solids: total solids > 4 %

Note: Transition periods have not been included

Table 2. Average feed and digested sludge characteristics and operational parameters during anaerobic digestion of low-solids (periods I-VI) and high-solids (periods VII-IX) sludge

Parameter	**Period**					
	I	II	III	IV	V	VI
Working conditions						
T (ºC)	55.3 ± 1.2	55.4 ± 1.3	55.4 ± 0.5	55.3 ± 0.2	54.7 ± 0.4	54.2 ± 1.7
SRT (d)	29.1 ± 1.5	30.3 ± 3.3	25.4 ± 4.4	20.4 ± 2.8	16.0 ± 1.7	10.4 ± 0.5
OLR (kg VS m$^{-3}_{reactor}$ d^{-1})	0.47 ± 0.1	0.69 ± 0.1	0.97 ± 0.5	1.05 ± 0.2	1.38 ± 0.3	1.65 ± 0.3
Feed composition						
TS (g L^{-1})	19.63 ± 1.67	32.77 ± 8.04	31.48 ± 10.84	30.34 ± 7.38	28.86 ± 6.86	23.22 ± 5.17
VS (g L^{-1})	13.30 ± 0.85	22.16 ± 4.91	23.25 ± 7.70	21.34 ± 4.12	21.01 ± 5.14	17.93 ± 3.85
VS/TS	68.90 ± 4.67	68.21 ± 0.74	74.23 ± 1.79	70.59 ± 2.20	74.78 ± 1.80	77.52 ± 2.00
VFA C2-C5 (g COD L^{-1})	1.68 ± 0.32	4.30 ± 0.69	3.59 ± 0.55	2.72 ± 0.55	4.51 ± 0.79	3.68 ± 0.76
pH	6.97 ± 0.57	6.04 ± 0.11	5.75 ± 0.18	6.25 ± 0.12	5.92 ± 0.07	6.13 ± 0.29
Effluent composition						
TS (g L^{-1})	13.09 ± 1.74	17.60 ± 1.58	14.92 ± 1.15	20.11 ± 2.80	17.59 ± 0.94	18.90 ± 4.63
VS (g L^{-1})	7.90 ± 0.92	11.15 ± 1.18	9.55 ± 0.87	13.50 ± 0.78	11.62 ± 0.68	14.00 ± 2.31
VS/TS	61.76 ± 0.98	63.19 ± 1.68	63.94 ± 1.14	64.81 ± 1.27	66.39 ± 2.34	70.06 ± 0.86
VFA C2-C5 (g COD L^{-1})	0.94 ± 0.53	2.16 ± 0.52	1.60 ± 0.81	2.49 ± 0.48	2.60 ± 0.22	2.31 ± 0.63
Acetate (g L^{-1})	0.12 ± 0.17	0.31 ± 0.13	0.17 ± 0.15	0.17 ± 0.05	0.03 ± 0.04	0.22 ± 0.12
Propionate (g L^{-1})	0.29 ± 0.12	0.69 ± 0.13	0.51 ± 0.24	0.79 ± 0.15	0.92 ± 0.07	0.99 ± 0.10
iso-Butyrate (g L^{-1}1)	0.07 ± 0.05	0.19 ± 0.04	0.12 ± 0.09	0.24 ± 0.06	0.27 ± 0.02	0.29 ± 0.03
Butyrate (g L^{-1})	0.01 ± 0.02	0.00	0.00	0.00	0.34 ± 0.03	0.08 ± 0.10
iso-Valerate (g L^{-1})	0.12 ± 0.06	0.21 ± 0.08	0.20 ± 0.10	0.34 ± 0.07	0.34 ± 0.03	0.42 ± 0.07
Valerate (g L^{-1})	0.00	0.00	0.00	0.00	0.00	0.03 ± 0.04
A/P ratio	0.46 ± 0.39	0.44 ± 0.17	0.39 ± 0.33	0.21 ± 0.07	0.08 ± 0.03	0.22 ± 0.09
IA (g $CaCO_3$ L^{-1})	0.88 ± 0.08	1.26 ± 0.18	1.09 ± 0.23	1.32 ± 0.13	1.40 ± 0.12	1.47 ± 0.19
IA/TA ratio	0.31 ± 0.03	0.43 ± 0.04	0.37 ± 0.06	0.39 ± 0.04	0.41 ± 0.02	0.46 ± 0.03
IA/PA ratio	0.45 ± 0.07	0.75 ± 0.12	0.59 ± 0.16	0.65 ± 0.09	0.71 ± 0.07	0.86 ± 0.12
pH	8.18 ± 0.11	8.03 ± 0.09	8.15 ± 0.17	8.08 ± 0.11	7.86 ± 0.12	7.91 ± 0.09
Removal efficiency						
TS removal (%)	30.7 ± 10.9	39.7 ± 15.9	50.1 ± 14.2	36.1 ± 17.1	35.0 ± 17.7	27.5 ± 20.9
VS removal (%)	42.2 ± 5.9	44.1 ± 5.9	53.4 ± 3.0	40.5 ± 9.1	43.2 ± 4.	22.7 ± 4.5
Biogas characteristics						
Biogas prod. rate (m^3 m$^{-3}_{reactor}$ d^{-1})	0.18 ± 0.06	0.28 ± 0.07	0.35 ± 0.12	0.41 ± 0.14	0.36 ± 0.11	0.56 ± 0.14
Specific biogas prod. (m^3 kg VS$_{fed}^{-1}$)	0.37 ± 0.11	0.36 ± 0.07	0.42 ± 0.12	0.43 ± 0.08	0.29 ± 0.10	0.37 ± 0.10
Biogas yield (m^3 kg VS$_{removed}^{-1}$)	0.63 ± 0.09	0.70 ± 0.10	0.90 ± 0.43	0.99 ± 0.47	0.81 ± 0.68	1.15 ± 0.20
Methane prod. rate (m^3 m$^{-3}_{reactor}$ d^{-1})	0.08 ± 0.02	0.22 ± 0.04	0.20 ± 0.04	0.30 ± 0.07	0.24 ± 0.03	0.36 ± 0.11
Specific methane prod. (m^3 kg VS$_{fed}^{-1}$)	0.17 ± 0.03	0.26 ± 0.03	0.28 ± 0.08	0.29 ± 0.08	0.19 ± 0.04	0.23 ± 0.06
Methane yield (m^3 kg VS$_{removed}^{-1}$)	0.40 ± 0.05	0.47 ± 0.05	0.61 ± 0.29	0.70 ± 0.31	0.59 ± 0.43	0.71 ± 0.13
Methane content (%)	63.64 ± 3.03	64.57 ± 4.86	65.07 ± 2.58	66.21 ± 1.20	64.02 ± 1.37	61.78 ± 1.49

Table 2 (cont). Average feed and digested sludge characteristics and operational parameters during anaerobic digestion of low-solids (periods I-VI) and high-solids (periods VII-IX) sludge

Parameter	**Period**		
	VII	VIII	IX
Working conditions			
T (ºC)	53.2 ± 0.3	53.6 ± 1.1	52.3 ± 1.5
SRT (d)	9.4 ± 0.8	6.2 ± 1.3	10.1 ± 1.1
OLR (kg VS $m^{-3}_{reactor}$ d^{-1})	3.7 ± 0.4	5.2 ± 0.5	2.4 ± 0.3
Feed composition			
TS (g L^{-1})	45.39 ± 3.52	54.61 ± 7.65	40.60 ± 10.93
VS (g L^{-1})	34.86 ± 2.34	31.21 ± 3.60	24.23 ± 2.70
VS/TS	75.71 ± 0.59	58.08 ± 10.29	62.02 ± 9.11
VFA C2-C5 (g COD L^{-1})	4.37 ± 1.83	2.49 ± 0.99	1.40 ± 0.22
pH	6.61 ± 0.12	6.81 ± 0.31	7.05 ± 0.25
Effluent composition			
TS (g L^{-1})	21.91 ± 2.34	37.97 ± 9.69	24.33 ± 6.40
VS (g L^{-1})	14.94 ± 1.72	18.49 ± 4.02	14.39 ± 2.76
VS/TS	68.08 ± 0.79	49.07 ± 2.82	60.18 ± 4.78
VFA C2-C5 (g COD L^{-1})	5.48 ± 0.74	2.62 ± 0.87	3.48 ± 0.70
Acetate (g L^{-1})	0.58 ± 0.18	0.18 ± 0.27	0.52 ± 0.20
Propionate (g L^{-1})	1.43 ± 0.07	1.03 ± 0.12	1.17 ± 0.15
iso-Butyrate (g L^{-1}1)	0.52 ± 0.03	0.10 ± 0.09	0.18 ± 0.10
Butyrate (g L^{-1})	0.06 ± 0.08	0.01 ± 0.03	0.01 ± 0.01
iso-Valerate (g L^{-1})	0.78 ± 0.10	0.33 ± 0.16	0.40 ± 0.13
Valerate (g L^{-1})	0.02 ± 0.03	0.00	0.00
A/P ratio	0.40 ± 0.11	0.16 ± 0.22	0.45 ± 0.19
IA (g $CaCO_3$ L^{-1})	2.09 ± 0.17	1.64 ± 0.30	2.18 ± 0.13
IA/TA ratio	0.44 ± 0.03	0.39 ± 0.03	0.40 ± 0.02
IA/PA ratio	0.79 ± 0.11	0.63 ± 0.07	0.66 ± 0.07
pH	8.03 ± 0.11	8.13 ± 0.04	8.18 ± 0.07
Removal efficiency			
TS removal (%)	50.2 ± 7.5	39.8 ± 11.1	37.2 ± 19.0
VS removal (%)	57.3 ± 4.2	40.6 ± 10.1	38.6 ± 10.6
Biogas characteristics			
Biogas prod. rate (m^3 $m^{-3}_{reactor}$ d^{-1})	1.07 ± 0.15	1.46 ± 0.14	0.61 ± 0.14
Biogas yield (m^3 kg VS_{fed}^{-1})	0.30 ± 0.03	0.28 ± 0.03	0.27 ± 0.04
Specific biogas prod. (m^3 kg $VS_{removed}^{-1}$)	0.51 ± 0.20	0.71 ± 0.21	0.59 ± 0.14
Methane prod. rate (m^3 $m^{-3}_{reactor}$ d^{-1})	0.62 ± 0.13	0.86 ± 0.12	0.38 ± 0.08
Methane yield (m^3 kg VS_{fed}^{-1})	0.18 ± 0.04	0.17 ± 0.03	0.16 ± 0.03
Specific methane prod. (m^3 kg $VS_{removed}^{-1}$)	0.35 ± 0.11	0.43 ± 0.11	0.38 ± 0.09
Methane content (%)	62.13 ± 3.46	64.33 ± 7.50	63.81 ± 3.75

Table 3. Limit values proposed to prevent process failure during the anaerobic thermophilic digestion of sludge

Parameter	**Limit value**
Acetate concentration (g L^{-1})	0.6
A/P ratio	0.5
VFA C2-C5 (g COD L^{-1})	3.7
Intermediate alkalinity (g $CaCO_3$ L^{-1})	1.8
IA/PA ratio	0.9
IA/TA ratio	0.5
Methane content in biogas (% CH_4)	55

Table 4. Microbiological analyses of influent and effluent sludge samples

Pathogens	**Influent (PS+WAS)**	**Effluent (SRT)**					
		30 d	**25 d**	**20 d**	**15 d**	**10 d**	**6 d**
E .coli (CFU mL^{-1})	1.0×10^6	Absence	Absence	Absence	1.0×10^1	1.0×10^1	1.1×10^2
Salmonella spp. (in 50 mL)	Absence	Absence	Absence	Absence	Absence	Absence	Absence

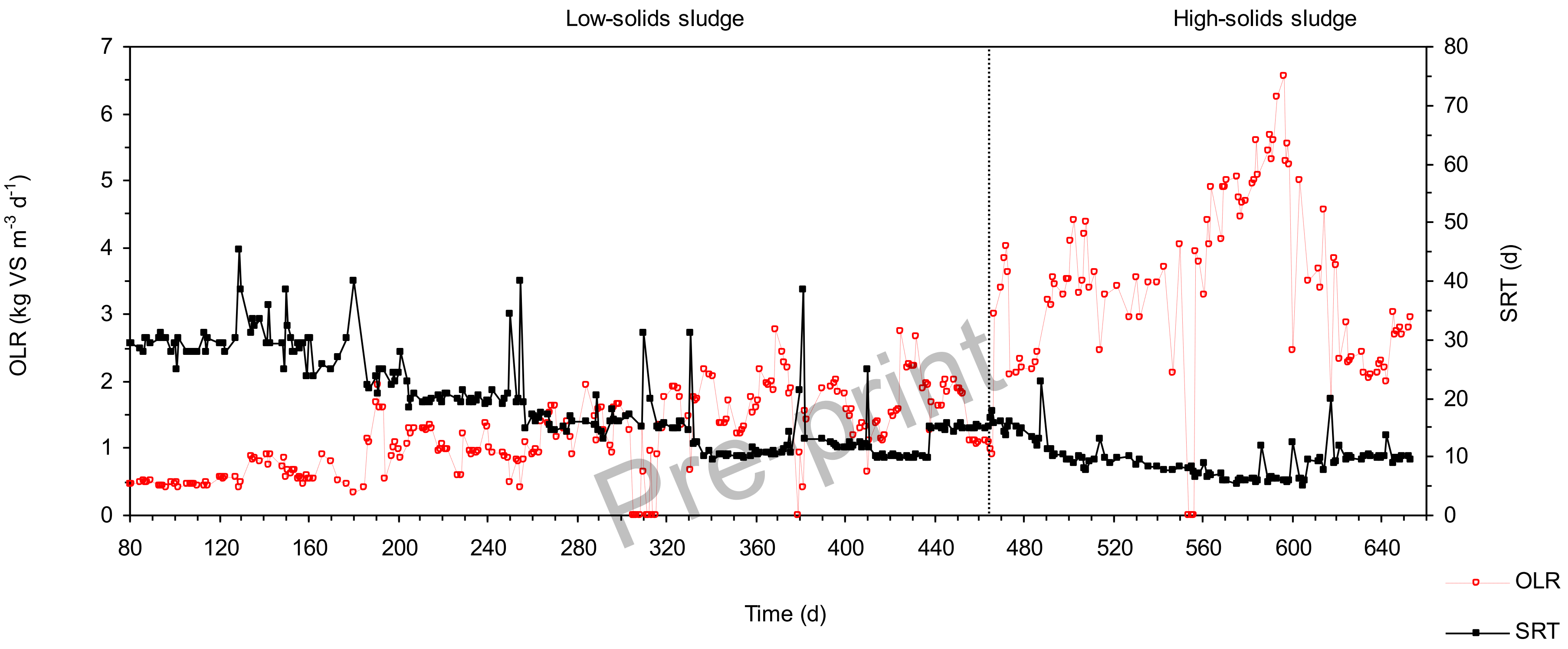

Figure 1. Sludge retention time (SRT) and organic loading rate (OLR) and during thermophilic (55 ºC) sludge digestion

Note: The start-up period has not been included

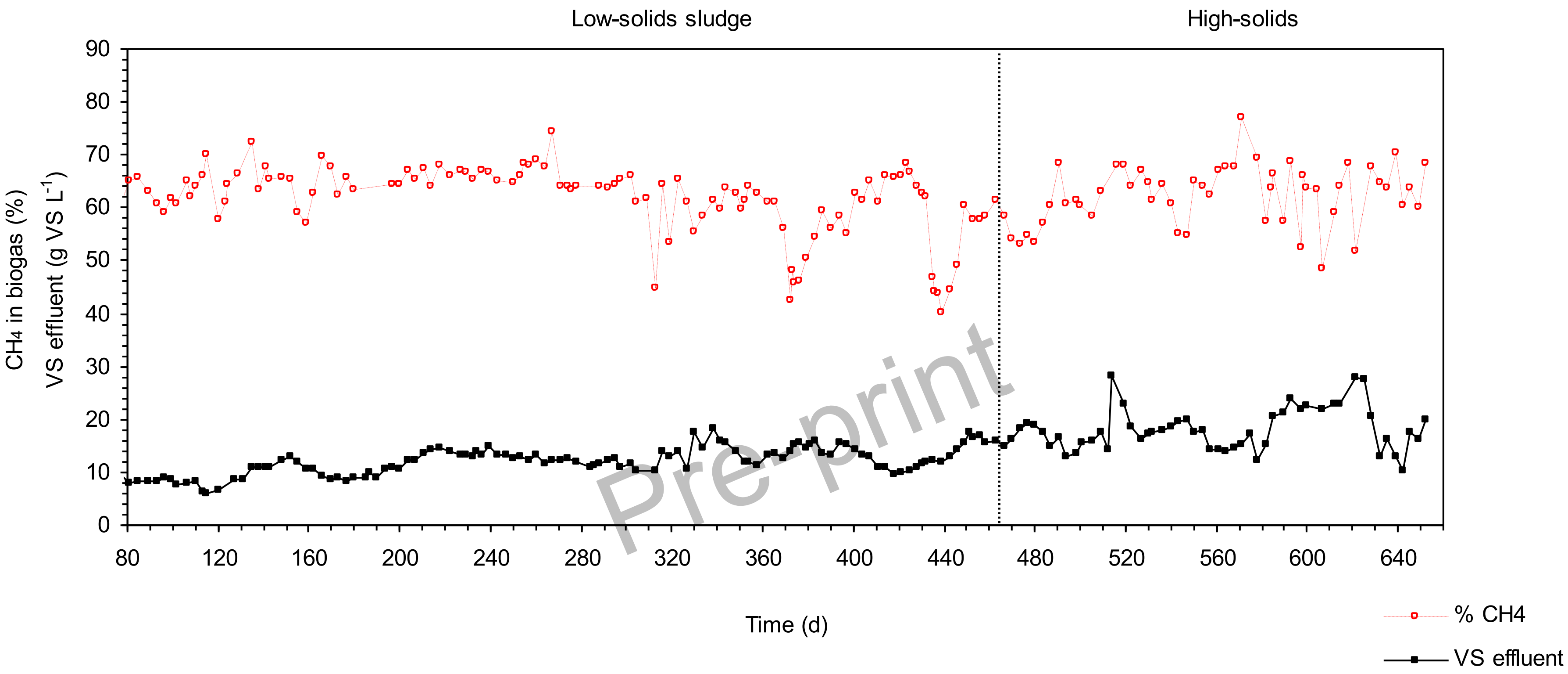

Figure 2. Methane content in biogas (% CH4) and effluent VS during thermophilic (55 ºC) sludge digestion

Note: The start-up period has not been included

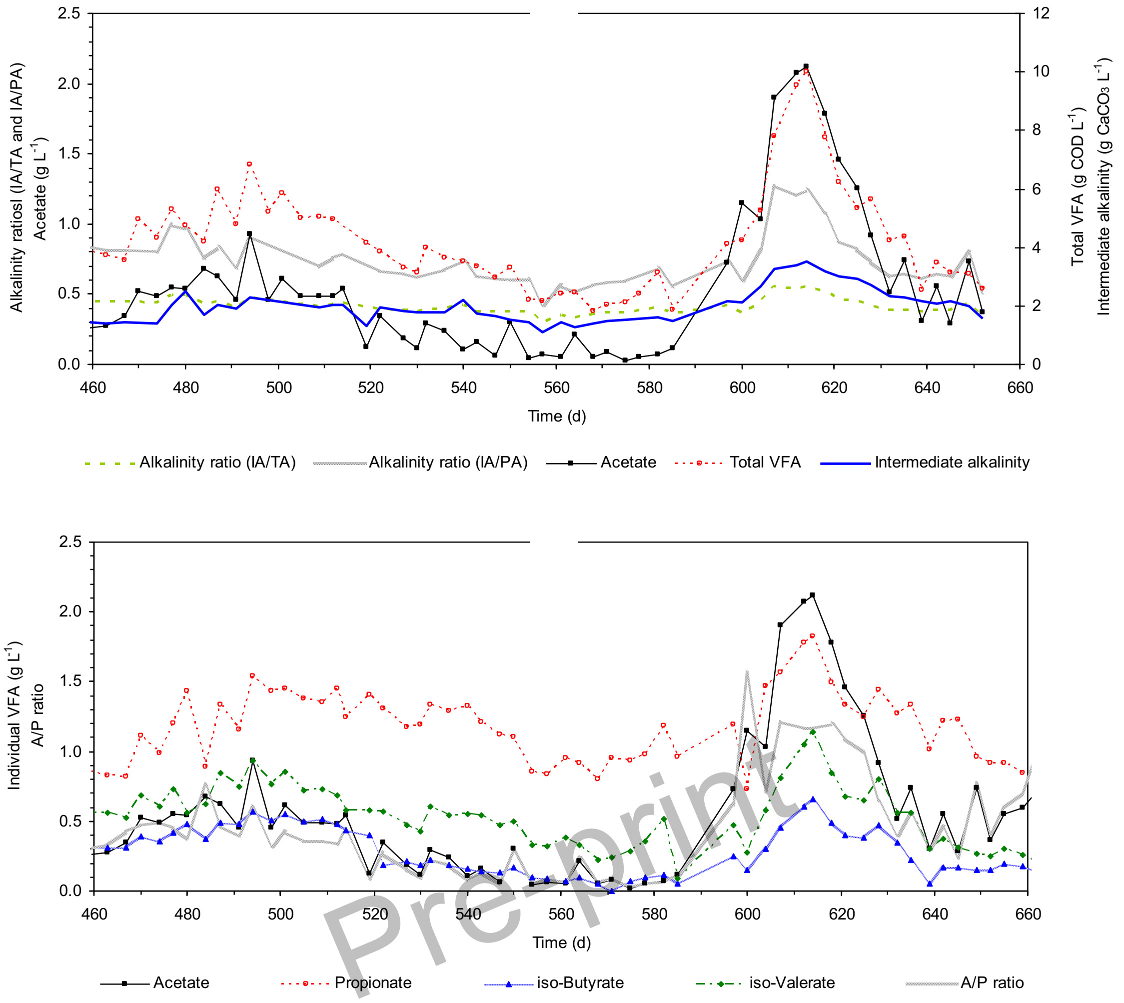

Figure 3. Volatile fatty acids (VFA) and alkalinity during thermophilic digestion of high-solids sludge: (a) Total VFA (VFA C2-C5), acetate concentration, intermediate alkalinity (IA), intermediate to total alkalinity ratio (IA/TA) and intermediate to partial alkalinity ratio (IA/TA); (b) individual VFA concentration and acetate to propionate ratio (A/P)

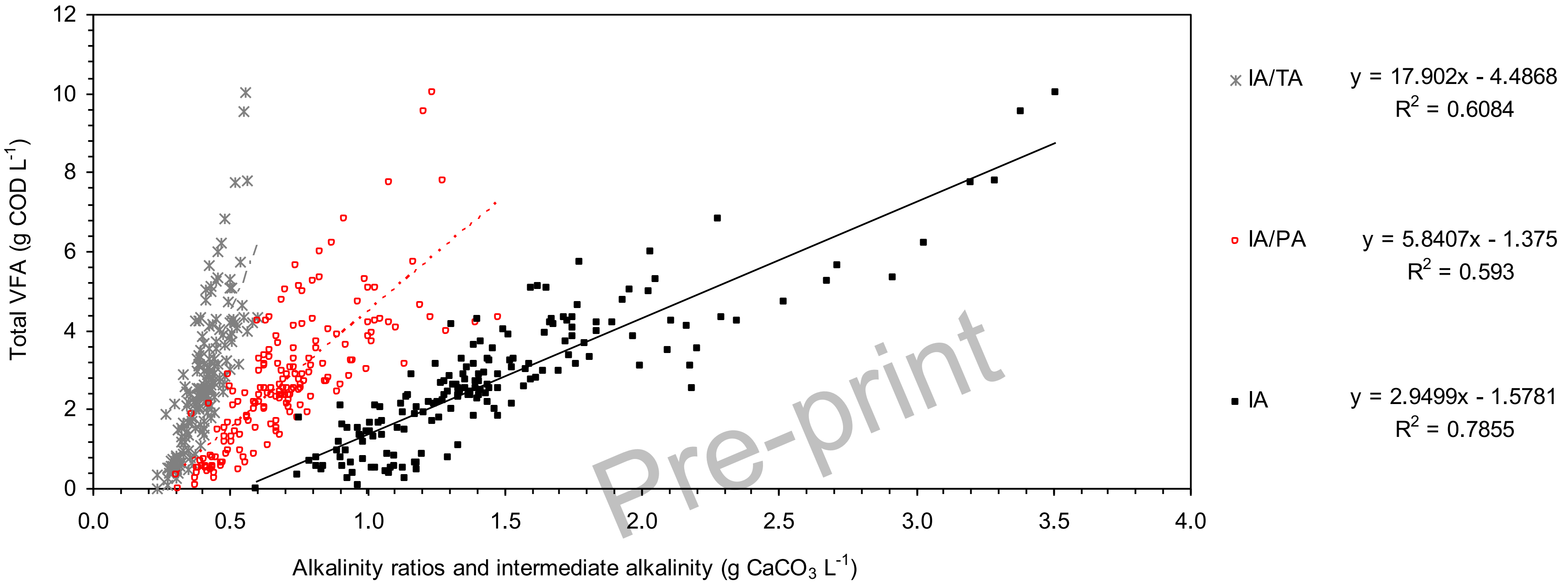

Figure 4. Correlation between total volatile fatty acids (VFA C2-C5) concentration and intermediate alkalinity (IA), IA to total alkalinity (IA/TA) and IA to partial alkalinity (IA/PA) ratios

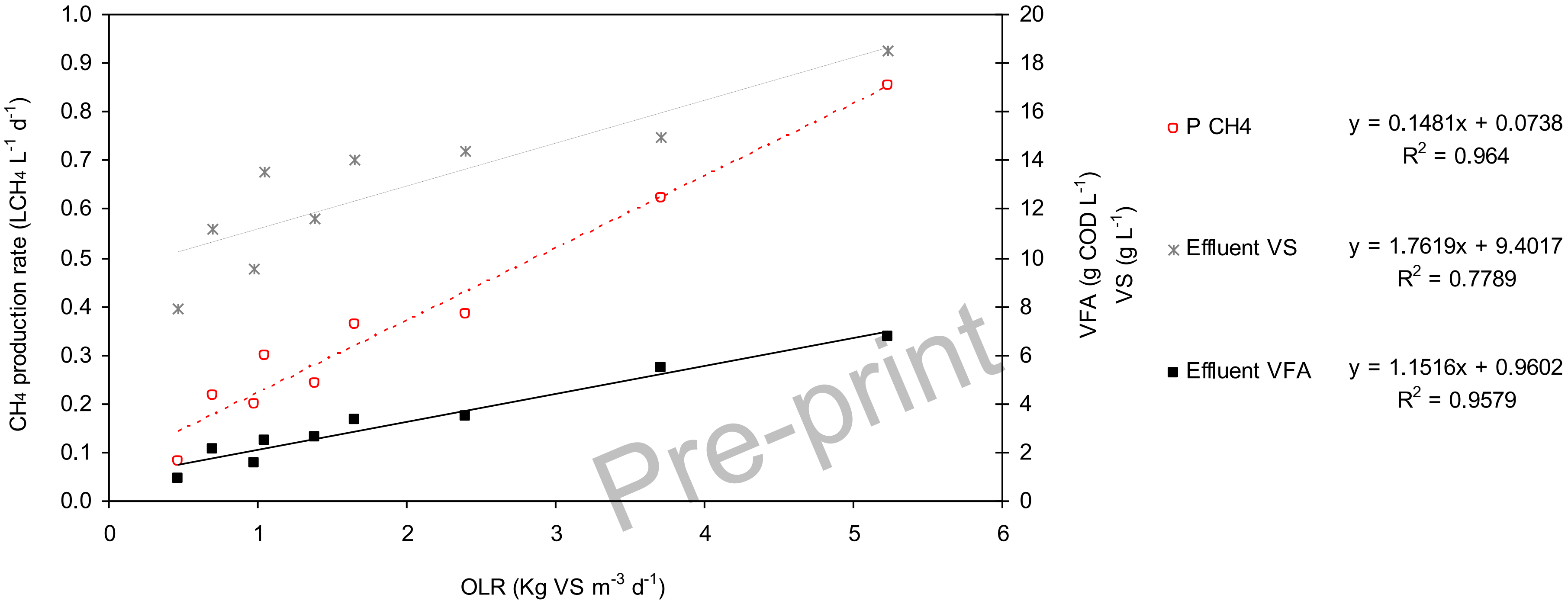

Figure 5. Methane production rate (P_{CH4}), effluent volatile fatty acids (VFA C2-C5) and volatile solids (VS) as a function of the organic loading rate (OLR), during thermophilic sludge digestion

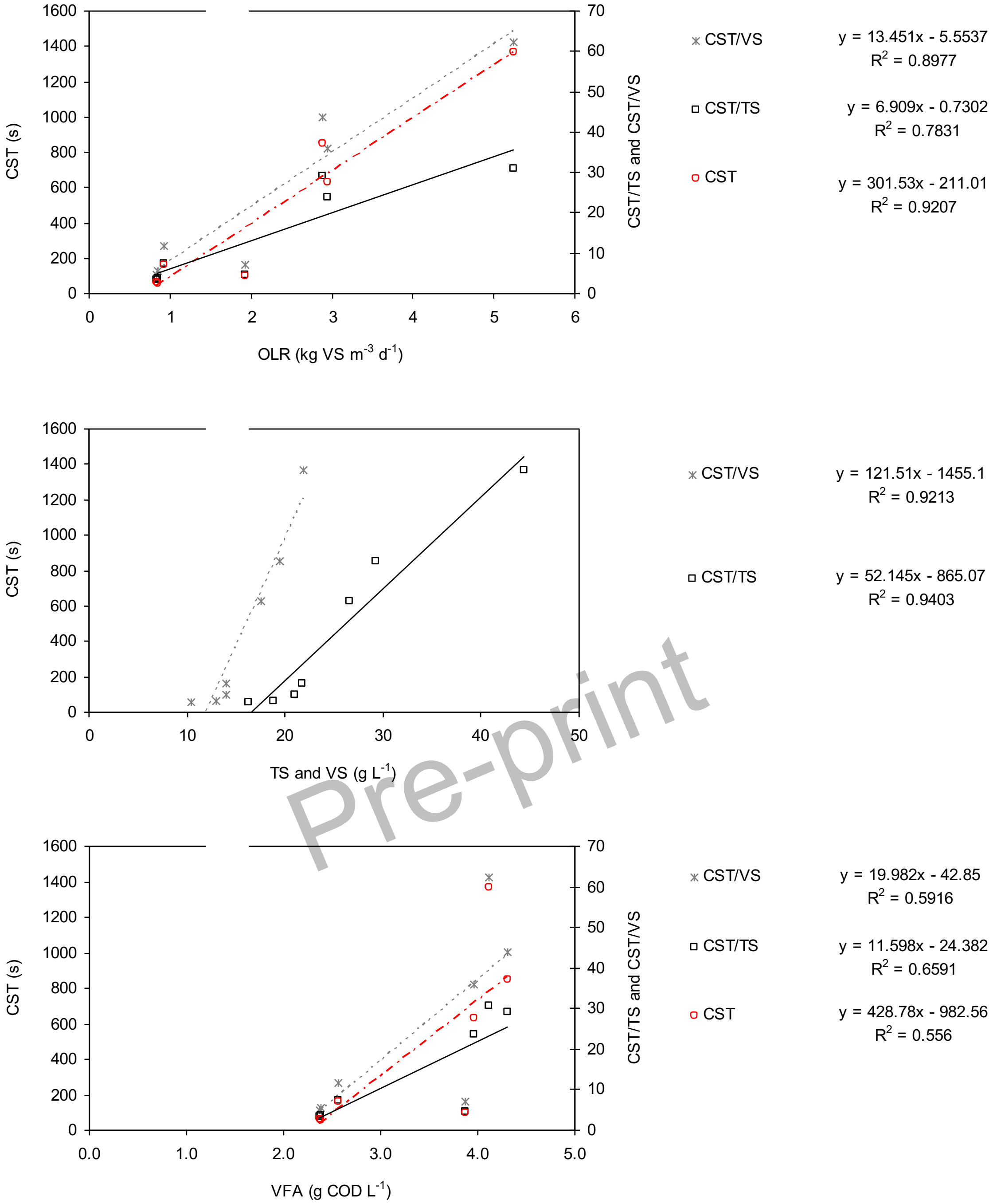

Figure 6. Capillary suction time (CST) of thermophilic digested sludge: (a) CST, CST per total solids (CST/TS) and CST per volatile solids (CST/VS) vs. organic loading rate (OLR); (b) CST vs. TS and VS; (c) CST, CST/TS and CST/VS vs. total volatile fatty acids (VFA C2-C5)

APPENDIX:

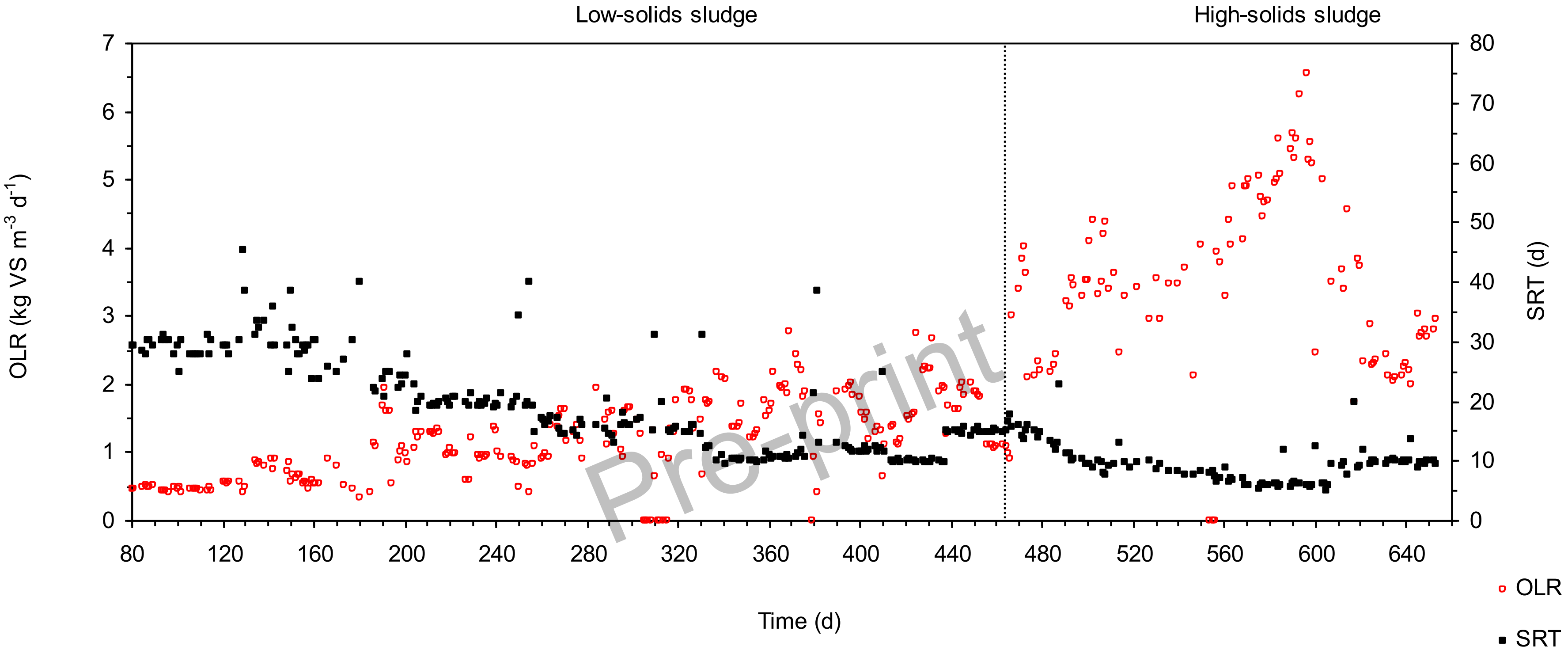

Figure 1. Sludge retention time (SRT) and organic loading rate (OLR) and during thermophilic (55 ºC) sludge digestion

Note: The start-up period has not been included