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Diagnosis of an anaerobic pond treating temperate domestic wastewater: An alternative sludge strategy for small works

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Abstract

An anaerobic pond (AP) for treatment of temperate domestic wastewater has been studied as a small works sludge management strategy to challenge existing practice which comprises solids separation followed by open sludge storage, for up to 90 days. During the study, effluent temperature ranged between 0.1°C and 21.1°C. Soluble COD production was noted in the AP at effluent temperatures typically greater than 10°C and was coincident with an increase in effluent volatile fatty acid (VFA) concentration, which is indicative of anaerobic degradation. Analysis from ports sited along the APs length, demonstrated VFA to be primarily formed nearest the inlet where most solids deposition initially incurred, and confirmed the anaerobic reduction of sludge within this chamber. Importantly, the sludge accumulation rate was 0.06 m^3 $capita^{-1}$ y^{-1} which is in the range of APs operated at higher temperatures and suggests a de-sludge interval of 2.3 to 3.8 years, up to 10 times longer than current practice for small works. Coincident with the solids deposition profile, biogas production was predominantly noted in the initial AP section, though biogas production increased further along the APs length following start-up. A statistically significant increase in mean biogas production of greater than an order of magnitude was measured between winters ($t_{(n=19)} = 5.52$, $P < 0.001$) demonstrating continued acclimation. The maximum methane yield recorded was 2630 $mgCH_4$ PE d^{-1}, approximately fifty times greater than estimated from sludge storage (57 $mgCH_4$ PE^{-1} d^{-1}). Anaerobic ponds at small works can therefore enable sludge reduction and longer sludge holding times than present, offsetting tanker demand, can reduce fugitive methane emissions currently associated with sludge storage, and based on the enhanced yield noted, could provide a viable opportunity for local energy generation.

Keywords: psychrophilic; psychrotolerant; methane production; municipal wastewater

1. Introduction

Due to population growth and legislative drivers implemented to enhance wastewater effluent quality, the sludge volume generated on-site at wastewater treatment works (WwTW) has increased. To illustrate, across the EU-15 countries sludge volume has increased by 34% over the last 20 years (Kelessidis and Stasinakis, 2012). To stabilise this sludge prior to safe disposal/reuse, many additional mesophilic anaerobic digester (AD) assets have since been built. However, due to economies of scale, AD is only really practicable for centralised large scale facilities serving dense populations which does not reflect the size distribution of WwTW. Across the EU, 80% of WwTW serve population equivalents (PEs) less than 5,000 (Alexiou and Mara, 2003). In the UK only 148 of >9,000 WwTWs currently employ AD (DEFRA, 2002; Anaerobic digestion portal, 2012). Consequently, sludge produced at small works is tankered to centralised WwTWs comprised of AD for treatment. However, tankering costs for sludge transportation, coupled with small sludge yields from individual WwTWs and the high number of small WwTWs can prove economically prohibitive, leading to either alternate management routes for sludge (McAdam et al., 2012) or extended periods of on-site sludge storage (up to 90 days) to limit tankering frequency (Hobson, 2001). Extended residence time in holding tanks, causes the retained sludge to degrade, reducing calorific value and increasing the likelihood for the generation of local fugitive emissions (Werther and Ogada, 1999; Hobson, 2001). Whilst limited data on fugitive emissions is available, in a US study, a fugitive methane flux of 6.9 to 10.9 $gCH_4\ m^{-2}\ d^{-1}$ from a sludge holding tank used for storage of primary and secondary sludge was recorded (Czepiel et al., 1993). Based on collated experimental data, Hobson (2001) estimated a specific methane emission of 36 $kgCH_4\ tonne^{-1}$ of raw dry solids (RDS) stored over a 90 day holding period, which was equivalent to 25% of the total yield attainable via mesophilic AD. Consequently, extended open sludge storage reduces the potential energy yield from the sludge if tanrkered offsite to AD, but also increases the risk of local greenhouse gas emissions.

Anaerobic ponds (APs) have been traditionally implemented in warm climates as a passive roughing stage to reduce the organic load onto subsequent treatment stages. AP are typically dimensionalised similarly to rectangular primary sedimentation tanks (PSTs) in a European WwTW (3:1 Length:Width aspect ratio) to enabled effective solids capture (Guyer, 2011). However, APs are also specifically oversized to allow extended sludge residence times (therefore combining both primary sedimentation tank and sludge holding tank) which enables anaerobic conditions to develop providing in-situ sludge volume reduction and therefore a reduction in desludging frequency to once every several years. The translation of this technology to a European context could therefore provide a potentially significant solution for sludge management at small works. Whilst an established technology in warm countries (DeGarie et al., 2000), most APs reported in the literature have been left uncovered, losing the opportunity to recover produced methane either for energy recovery or to limit carbon footprint, since the primary purpose has been for sludge reduction and protection of downstream assets. Consequently, there is currently extremely limited gas production data for APs treating domestic wastewater. Furthermore, the significant body of literature is based on APs applied to treatment of wastewaters with temperatures ranging 18°C to 25°C (McAdam et al., 2012), with few studies on application in temperate climates (Picot et al., 2003) largely due to a general perception that Northern European domestic wastewater cannot be treated anaerobically due to low temperatures and low organic strength (Lester et al., In Press) since kinetic rates in anaerobic degradation decrease with temperature (Lettinga et al., 2001). However, Langenhoff and Stuckey (2000) found that the Arrhenius equation, often used to model temperature effects on kinetic rates, may overestimate this decrease. Craggs et al. (2008) suggested that the methane yield (and hence solids degradation) in low temperature APs could equal those of mesophilic ADs, provided solids retention time were doubled to compensate for the lower kinetic rate. The following study therefore seeks to understand the potential role of APs for the treatment of temperate domestic wastewater, specifically through: (1) Long term operation (>1 y) of an AP to establish treatment performance during start-up and through a full annual cycle to establish resilience to temperature and seasonal variation; (2) quantify sludge accumulation rates and biogas production rates in temperate conditions to estimate desludge frequency and local energy yields; and (3) compare methane production rates to emission rates generated from three sludge holding tanks based at small scale UK WwTW to benchmark comparative environmental performance.

2. Materials and Methods

2.1 Experimental reactor design

A pilot-scale horizontally baffled anaerobic pond (AP) was constructed of 12mm uPVC sheeting and sealed with PVC hot welding to form a hydraulic volume of 230 L. The AP was dimensioned using a 3:1 Length:Width ratio in accordance to current best practice (Mara and Pearson, 1998) (Figure 1). The AP contained two baffles, located at L/3 and 2L/3 along the reactor length, which extended to the height of the reactor and 85% of the reactor width (Peña et al., 2003), creating three 'chambers'. An additional baffle that extended from the top of the reactor down to below water level was located adjacent to the outlet, to prevent gas escape through the outlet. The reactor was sealed with a gas-tight lid that comprised three gas sampling ports located at each of the baffled sections to enable evaluation of gas production along the length of the pond. In addition to inlet/outlet, sampling ports were installed at 0.25m, 0.75m and 1.25m along the reactor length to aid diagnosis.

The reactor was initially seeded with 7% by volume anaerobic sludge (VS = 36 g L^{-1}) collected from a mesophilic AD. The AP was fed crude wastewater at a liquid flow rate of 75 L d^{-1}, yielding a theoretical HRT of 3.1 days, which is in agreement with previous full-scale AP studies (McAdam et al., 2012). Based on an average inlet crude wastewater total Chemical Oxygen Demand (tCOD) of 546 mg L^{-1}, this yielded an average organic loading rate (OLR) of 0.18 kgCOD m^{-3} d^{-1} which is also in the range of previous full-scale African and South American studies (De Oliveira, 1990; El-Deeb Ghazy et al., 2008; Peña, 2002). Influent and effluent were analysed three times a week in duplicate for total suspended solids (TSS), volatile suspended solids (VSS), tCOD, soluble COD (sCOD) and biochemical oxygen demand (BOD_5). Liquid samples were also collected and analysed once a month from the side ports. ANOVA tests were performed on all data sets to determine statistical significance of differences in means to 95% confidence. Data sets were first analysed for normal distribution, using normality probability plots with r^2 >0.95 assumed to be normally distributed, to determine the application of parametric or non-parametric ANOVA tools. Parametric data were examined for equal means using two-way student t-tests for equal variances or Welch's t-test for non-equal variances of the data sets. Non-parametric data were examined for equal medians using the Wilcoxon signed-rank test for paired samples sets and the Mann-Whitney U test for independent data sets.

2.2 Determination of sludge degradation from three full-scale STWs

Sludge samples were taken from three decentralised WwTWs in the UK, which contained a primary sedimentation tank (PST) and final sedimentation tank (FST), but with differing secondary treatments. The sites utilised a trickling filter (TF, dry weather flow (DWF)=36,000 m^3 d^{-1}, PE=112,289), an oxidation ditch (OD, DWF=1,320 m^3 d^{-1}, PE=5,533), and a rotating biological contactor (RBC, DWF=210m^3 d^{-1}, PE=765). Subsamples from sludge holding tanks on each site were collected and stored in sample vessels at room temperature (19.5^oC ± 2.0^oC) for 8 weeks. Sludge samples were setup in triplicate.

2.3 Analytical methods

Samples were analysed for BOD_5, COD, TSS and VSS according to standard methods (APHA, 1998). Measurement for sCOD was taken after filtering through a 1.2µm glass fibre filter (Whatman, Maidstone, UK) with the particulate COD fraction (pCOD) calculated by subtracting sCOD from tCOD. Calorific value (CV) of sludge samples was determined using bomb calorimetry according to CEN/TS 15400 (Marchwood Scientific Services, Southampton, UK). A range of six volatile fatty acids (VFA), acetic, propionic, butyric, n-butyric, i-valeric and n-valeric, were determined by high performance liquid chromatography (HPLC-UV) using a 1 mM H_2SO_4 mobile phase to elute through a fermentation separation column (Bio-Rad, California, USA). Particle size distribution (PSD) was measured using a laser diffraction particle sizer (Mastersizer 2000, Malvern Instruments, Malvern, UK). Biogas was captured in gas-tight sampling bags and analysed twice a week for total volume and gas composition. Gas volume was measured using a displacement method adapted from Mshandete et al. (2005). Gas composition was measured by gas chromatography with a thermal conductivity detector (CSi 200 Series, Cambridge Scientific Instruments Ltd, Cambridge, UK). Sludge depth was measured following 129 d and 534 d using a perspex tube graduated at 1mm intervals. To enhance spatial resolution, a grid of 0.1mx0.1m was used. Ambient and liquid temperatures were recorded at the time of sampling using a digital probe thermometer, with a sensitivity of ±0.05^oC.

3. Results

3.1 Impact of residence time on sludge degradation in sludge holding tanks

Sludge samples collected from on-site sludge holding tanks at three full-scale de-centralised WwTW were monitored for 8 weeks to measure sludge degradation and fugitive GHG emissions. Total solids concentrations of 40 kg m^{-3}, 7.5 kg m^{-3} and 40 kg m^{-3} were measured in sludge samples from the WwTWs comprising the TF, RBC and OD respectively. An initial increase is soluble COD was noted at the start of the trial which is indicative of hydrolysis and was characterised by a first-order relationship (Figure 2). During this period, the kinetic rates of hydrolysis (k_h) were calculated as 0.02 d^{-1} for the RBC sludge and 0.008 d^{-1} for the TF and OD sludges. However, following 6 weeks, 4 weeks and 2 weeks storage of the TF, RBC and OD sludge respectively, the residual sCOD in the sludge declined and was coincident with the production of methane. During the period monitored, average methane production rates of $2.1\text{x}10^{-6}$ $kgCH_4$ d^{-1}, $2.0\text{x}10^{-6}$ $kgCH_4$ d^{-1} and $4.8\text{x}10^{-5}$ $kgCH_4$ d^{-1} were recorded for the TF, RBC and OD respectively. As a consequence, following eight weeks storage, calorific value (CV) reduced from 13,781 kJ kg^{-1}, 13,361 kJ kg^{-1} and 13,767 kJ kg^{-1} for the TF, RBC and OD WwTW respectively to 12,432 kJ kg^{-1}, 12,056 kJ kg^{-1} and 11,990 kJ kg^{-1}, or equivalent to a reduction in mean calorific value of between 9.8% and 12.9%.

3.2 Characterisation of solids and organics removal within the anaerobic pond

Over the full study period (534 d), COD removal was characterised into three fractions (total, soluble and particulate) and average removals of 46±19% tCOD (n=93), 69±15% pCOD (n=93) and -17±40% sCOD (n=93) were recorded respectively. Fractionated COD data was also collated into monthly averages to discern the contribution of temperature on removal (Figure 3). For the particulate fraction, average monthly pCOD removal ranged from 51±19% (n=13) to 83±4% (n=5), with the minimum and maximum recorded during average monthly temperatures of 8.5°C and 17.9°C respectively. No statistical difference was observed ($t_{(n=42)}$=0.13, p=0.90) between mean pCOD removal rates recorded during winter and summer (Dec.-Feb. 74±10%, $T_{effluent}$=4.6°C; Jun.-Aug., 75±10%, $T_{effluent}$=16.7°C). However, the impact of temperature on sCOD removal was more evident ($U_{(n=44)}$=582, p=<0.001). To illustrate, during the summer period, negative sCOD removal of -26±33% was recorded (Jun.-Aug., $T_{effluent}$=16.7°C), whereas during winter, positive sCOD removal of 11±25% was determined (Dec.-Feb., $T_{effluent}$=4.6°C). The increase in sCOD with temperature, is indicative of volatile fatty acid (VFAs) formation (McAdam et al., In Press), which was supported by a weak positive correlation between effluent VFA concentration and effluent temperature (Figure 4). More specifically, at effluent temperatures above 12°C, VFA concentration markedly increased as a proportion of sCOD, whereas at effluent temperatures less than 15°C, VFA carbon contributed less than 25% of the effluent sCOD. Acetic acid was the dominant VFA identified, constituting on average 54% (*n*=45) of the total molar concentration.

3.3 Retention, accumulation and spatial distribution of solids in the anaerobic pond

Throughout the year, mean removal of 71±13% TSS was recorded (*n*=93). The consistency with which the AP retained particulate material was also assessed by developing resilience curves from the annual TSS influent and effluent data (Figure 5). The influent TSS profile generated from the annual data indicated an unstable TSS concentration profile within the influent (TSS range 91 mg L^{-1} to 1573 mg L^{-1}), as demonstrated by the positive skew above the 90^{th} percentile. Median particle size in the influent ranged from 35μm to 235μm. The effluent profile of the AP was characterised by a steep gradient and a limited tail in the upper quartile of the distribution, analogous to a leptokurtic distribution, and is indicative of limited instability. To illustrate, TSS effluent concentrations of 62 mg L^{-1}, 77 mg L^{-1} and 80 mg L^{-1} were recorded at the 50^{th}, 75^{th} and 90^{th} percentile, confirming the characteristic narrow distribution. A d_{50} median particle size of 20μm was measured in the effluent. The effluent profile was compared to the effluent TSS profile generated from a full-scale UK based primary sedimentation tank (PST) and a full-scale AP which is the only known AP to be currently treating domestic wastewater for the collection of methane. In both cases, the reference technologies were subject to higher average TSS concentrations, with 92% (*n*=32) and 37% (*n*=40) of the influent TSS samples >300 mgTSS L^{-1} for the full-scale AP and PST respectively versus only 29% (*n*=93) for the AP. However, similar effluent distribution profiles were evident when compared to the AP, which is of note since the reference AP was operated at a higher average operating temperature of 19.6 °C and the PST operated at a contrasting HRT approaching 0.1 d. Sludge volume distribution was initially assessed at day

219 which showed 67%, 13.5% and 19.5% of the sludge volume to be distributed between the first, second and third chambers respectively (Figure 6). Final analysis at 534 d measured 47% of the sludge volume distributed in the front chamber and 26.5% measured in chambers 2 and 3. The final total accumulated sludge volume was approximately 29 L or 13% of the total reactor volume which converts to a sludge accumulation rate of 0.06 m^3 $capita^{-1}$ $year^{-1}$. At the end of the study, the average VS content of the sludge layer was 55±13% (*n*=8), 46±9% (*n*=8) and 41±10% (*n*=8) for chambers 1, 2 and 3 respectively.

3.4 Temporal and spatial variations in biogas production and composition

Methane production was predominantly distributed into chamber one closest to the inlet, which coincides with where high pCOD removal was observed (Figure 7). A mean annual production rate of 3.69 LCH_4 m^{-3}WWT (*n*=57) was recorded in chamber 1, with 0.76 LCH_4 m^{-3}WWT (*n*=57) and 0.13 LCH_4 m^{-3}WWT (*n*=57) recorded in chambers 2 and 3 respectively. Methane production in each chamber was subject to temporal effects, with low production noted during the first two quarters of operation, followed by an increase in warmer temperatures to a maximum in summer (Q4), and a subsequent decline in the second winter period (Q5 and Q6). Whilst there was no statistical difference in median effluent temperatures between the two winter periods (Q2 = 4.9°C, Q6 = 6.6°C; $U_{(n=22)}$=197, *p*=0.42), mean biogas production was significantly higher in the second winter at 2.53 L CH_4 m^{-3} WWT (Q6, $t_{(n=19)}$=5.25, *p*=<0.001), compared to the initial winter period (Q2, 0.22 L CH_4 m^{-3} WWT;), indicating acclimation to have occurred over the study. Following start-up, biogas methane composition also progressively increased in chamber 1 from an initial 12% CH_4 in Q1 ($T_{effluent}$ 6.6°C) to 56% CH_4 in Q5 ($T_{effluent}$ 11.2°C) (Figure 8). A similar increase in methane composition was noted in chambers 2 and 3 with highest mean methane composition observed during Q5 at 45.3 % and 28.5 % respectively.

Total methane gas production ranged between 0.02 LCH_4 m^{-3} wastewater treated (WWT) and 19.89 LCH_4 m^{-3}WWT over the full study. Whilst no clear correlation with temperature was determined, a general increase in methane production with temperature was evident (Figure 9) and could be broadly differentiated into two datasets at around 8.8 °C (marked with a dashed line) which is equivalent to the minimum crude wastewater influent temperature measured during the study. In all, 96 % of gas production data below 1 LCH_4 m^{-3} WWT (n=23) and 92% of biogas composition data under 35% CH_4 v/v (n=25) were recorded for effluent temperatures below 8.8°C, yielding a mean production rate of 0.62 LCH_4 m^{-3}WWT. The heat loss necessary to achieve effluent temperatures from <8.8°C to below 0.5°C can be explained by the experimental positioning of the pilot-scale AP on an above ground support structure rather than buried below ground as with full-scale AP, which resulted in an effluent temperature profile more closely described by ambient air temperature than the influent wastewater ($T_{ambientair}$ -4.1°C to 22.7°C). For the full data set above 8.8°C, a mean production rate of 8.48 LCH_4 m^{-3}WWT was recorded, with the higher methane yield being commensurate with increased average methane gas composition of 49% CH_4 v/v.

4. Discussion

Data collected from this trial demonstrates that anaerobic ponds can be used to reduce methane emissions and desludge frequency from small works based in cold climates through replacing primary sedimentation tank and sludge holding tank assets as a single unit process. To illustrate, methane emission rates determined with sludge from three sludge holding tanks, demonstrated between 1.15 and 26.8 $kgCH_4$ $tonne^{-1}$ RDS would be released over a typical 90 day retention time, or 0.05 to 1.2 gCH_4 m^{-2} d^{-1}. Whilst lower than those recorded in the literature of approximately 36 $kgCH_4$ $tonne^{-1}$ RDS and 7 gCH_4 m^{-2} d^{-1} (Hobson, 2006; Czeipel et al., 1993), the data provides a conservative estimate of UK sludge holding tank methane emissions and importantly suggests that covered AP could omit this release (up to 57 $mgCH_4$ PE d^{-1}). Following continued AP operation without sludge withdrawal, it follows that there is an optimum loading rate after which effluent quality will decline due to washout (Peña and Mara, 2003; Toprak, 1994). However, the effluent TSS profile from the AP compared favourably to the effluent TSS profiles collected from a full-scale AP operated in Melbourne for domestic wastewater treatment and a full scale UK primary sedimentation tank which was characterised by a similar influent TSS profile. Spatial distribution of the resident sludge volume at 219d illustrated that 67% of retained sludge was in the first chamber (Figure 6) and is consistent with reports on full scale APs (Picot et al., 2005; Paing et al., 2000). This can be attributed to the reasonably coarse particle diameter of the influent wastewater biasing early sedimentation (d_{50} 35-235 µm), the low superficial velocity imposed by a 3 d HRT, and the inclusion of a baffle which dissipated momentum and local velocities (Shilton, 2003), enhancing sludge accumulation in the front chamber. The early physical separation of TSS within this standard AP design therefore enables consistent solids separation performance in colder temperatures despite the transient and continuous accumulation of a sludge layer, and so presents a suitable replacement for existing PSTs. Importantly, Daelman et al. (2012) reported methane emissions of 8 $kgCH_4\ hr^{-1}$ from a PST on a 360,000 PE WwTW (533 $mgCH_4\ PE^{-1}\ d^{-1}$), indicating that whilst short HRT are used, release of fugitive methane is also promoted in PSTs. Consequently, a fugitive methane emission of 590 $mgCH_4\ PE^{-1}\ d^{-1}$ could be avoided by using a covered AP to replace both the sludge holding tank and PST.

A sludge accumulation rate of 0.06 $m^3\ capita^{-1}\ year^{-1}$ was recorded based on data at the completion of the trial, which is in the range of earlier APs operated at higher temperatures (Nelson et al., 2004, Picot et al., 2005). At completion, only 47% of the total accumulated sludge was resident in the initial chamber, and the total sludge volume used accounted for 13% of available volume. Desludge frequency is commonly based on reaching 30 to 50% v/v (Mara and Pearson, 1998), which suggests an interval of 2.3 to 3.8 years. The volume redistribution noted was due to sludge accumulation local to the inlet, reducing channel area, which increases the local velocity profile, enabling extended particle transport along the path length of the AP. Sludge reduction in the first chamber over the warmer summer months is also expected to have influenced the observed sludge volume redistribution; an observation supported by the tendency for increased effluent VFA concentration and sCOD formation in the summer months and on average 81% of total methane production manifesting from the front chamber. Picot et al. (2003) similarly noted a sharp increase in biogas production after the winter period. The authors proposed that increased temperature initiated degradation of the carbon stored in the sludge layer during winter. However, methane activity did increase along the length of the AP, following a period of establishment. Biogas production recorded in the second winter period (Q6) was an order of magnitude higher than when compared to the first winter period (Q2), despite there being no statistical difference between effluent temperatures at both periods. Heubeck and Craggs (2010) reported on an AP treating pig slurry and found that the minimum temperature at which methane was formed decreased as the pond aged. It is therefore proposed that the higher biogas production exhibited in Q6 is indicative of an extended period of acclimatisation. The VFA formation observed in this study has also previously been considered an indication of acclimation, where VFAs have been observed in effluent for up to a year following start-up (Picot et al., 2003). However, VFA formation was noted at the end of the study period (>500d), despite the establishment of methane production. Lew et al. (2009) reported that at temperatures below 20^{o}C, anaerobic degradation of particulates was inhibited by temperature, whereas degradation of the soluble fraction was not. In this study, the dominant VFA formed was acetic acid, which is readily amenable and so it is suggested that the low superficial liquid velocities exhibited in the AP limited mixing (Peña et al., 2003) and thus limited contact between the soluble organic fraction (VFAs) formed in the first chamber and the sludge layer resident in the subsequent two chambers. Maximum methane production of 19.89 LCH_4 m^{-3}WWT was measured in Q4 which was coincident with the highest average effluent temperature; a mean of 4.92 LCH_4 m^{-3}WWT was recorded for the full study. Importantly, in this study, the AP was not insulated from the cold and so equilibrated to local air temperatures which at times approached 0°C. At full scale, the surrounding soil bank provides insulation such that the temperature profile would more closely resembles the influent wastewater, which in this study was consistently above 8.8°C (Safley Jr. and Westerman, 1989; Park and Craggs, 2007). Consequently, a mean of 8.48 LCH_4 m^{-3}WWT (mean recorded above 8.8°C) potentially more closely describes the expected yield. However, this does not take in to consideration the expected continued enhancement in methane yield following furthered acclimation.

5. Conclusions

The AP has been demonstrated to achieve extended sludge storage in temperate conditions, without compromising effluent quality, and based on the utilisation of methane collection, affords lower fugitive emission rates. To achieve extended sludge storage up to 10 times as proposed, an extended land area is demanded to support the 3 day HRT. Whilst potentially constraining for large-scale WwTWs in urbanised areas, their application at small-scale, rural works is considered realistic. Furthermore, since up to 80% of the solids separation occurred in the front third of the AP, scale could be considerably reduced, though this will inevitably present a trade off with desludge frequency. The methane emission rates estimated from sludge holding tanks in temperate conditions present compelling evidence for the need to capture fugitive emissions. However, utilisation of fugitive methane from sludge holding tanks in isolation (57 $mgCH_4$ PE d^{-1}) is unlikely to be economically viable, indicating gas capture and flaring to be best practice, which remains a more environmentally sound carbon management strategy than currently employed. Through replacing sludge holding tanks with the AP, the methane yield increased by around 50 times, and since biogas methane content remained >35% following start-up (even during winter), there is potential for small scale electrical production through combined heat and power (CHP). Economies of scale for biogas CHP systems are continually falling, with commercial units known to be available at 3-15 kW_e with a base cost of around £1045 kW_e (not installed). Based on the yield in this study, 0.25 kW_e of capacity is required per 100 PE, and assuming a feed-in-tariff of £0.14 kWh^{-1}, would deliver annual revenue of £307 y^{-1}, indicating payback of around three years. After ten years of operation, an AP in Melbourne, Australia, delivered a yield of 0.16 m^3CH_4 m^3WWT, around eight times higher than in this study, which would advantage the economics further. Whilst an equivalent yield cannot be expected due to the temperature differential (Melbourne sewage average temperature, 19.6°C; northern hemisphere, 12°C), the statistically significant increase in methane yield between winters, coupled with the continued production of VFA, is indicative of acclimation and suggests a higher yield is possible with longer operation. Further optimisation of AP design could also be considered to enhance methane yield. For example, driving contact between soluble substrate (VFA) and the active sludge layer in the latter pond section using engineering interventions such as vertical baffling could enhance production. The potential demonstrated in this study therefore warrants further examination into optimised design; the economic argument is further compounded if weighted against the cost of carbon associated with the existing fugitive emission from both holding tanks and PSTs.

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References

Alexiou, G.E., Mara, D.D., 2003. Anaerobic waste stabilization ponds: A low-cost contribution to a sustainable wastewater reuse cycle. Applied Biochem. Biotechnol. 109, 241-252.

Anaerobic digestion portal (29 June, 2012), Official biogas plant map, available at: http://biogas-info.co.uk/index.php/ad-map.html (accessed 27 July 2012)

British Standards Institution, 2006. Solids recovered fuels - Methods for the determination of calorific value, DD CEN/TS: 15400, British Standards Institution, London, UK

Craggs, R., Park, J., Heubeck, S., 2008. Methane emissions from anaerobic ponds on a piggery and a dairy farm in New Zealand. Aust. J. Exp. Agric. 48, 142-146.

De Oliveira, R., 1990. The performance of deep waste stabilization ponds in northeast Brazil (unpublished PhD. thesis), The University of Leeds, Leeds: UK.

DeGarie, C.J., Crapper, T., Howe, B.M., Burke, B.F., McCarthy, P.J., 2000. Floating geomembrane covers for odour control and biogas collection and utilization in municipal lagoons. Wat. Sci. Tech. 42, 291-298.

DEFRA, 2002. Sewage treatment in the UK, UK implementation of the EC urban wastewater treatment directive, DEFRA, London, UK.

El-Deeb Ghazy, M.M., El-Senousy, W.M., Abdel-Aatty, A.M., Kamel, M., 2008. Performance evaluation of a waste stabilization pond in a rural area in Egypt. Am. J. Environ. Sci. 4, 316-326.

Heubeck, S., Craggs, R.J., 2010. Biogas recovery from a temperate climate covered anaerobic pond, Wat. Sci. Tech. 61, 1019-1026.

Hobson, J., 2001. Emissions from wastewater handling, in Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories, IPCC, Geneva, Switzerland

Kelessidis, A., Stasinakis, A.S., 2012. Comparative study of the methods used for treatment and final disposal of sewage sludge in European countries. Waste Manage. 32, 1186-1195.

Langenhoff, A.A.M., Stuckey, D.C., 2000. Treatment of dilute wastewater using an anaerobic baffled reactor: Effect of low temperature. Water Res. 34, 3867-3875.

Lettinga, G., Rebac, S., Zeeman, G., 2001. Challenge of psychrophilic anaerobic wastewater treatment. Trends Biotechnol. 19, 363-370.

Mara, D.D., Pearson, H.W., 1998. Design Manual for Waste Stabilization Ponds in Mediterranean Countries, First ed, Lagoon Technology International, Leeds: England.

McAdam, E.J., Ansari, I., Cruddas, P.H., Martín-García, N., Lester, J.N., Pursell, N., Cartmell, E., Jefferson, B., 2012. Waste stabilisation ponds for anaerobic wastewater treatment. Proc. Inst. Civil. Eng. Eng. Sustainability 165, 201-213

Nelson, K.L., Cisneros, B.J., Tchobanoglous, G., Darby, J.L., 2004. Sludge accumulation, characteristics, and pathogen inactivation in four primary waste stabilization ponds in central Mexico. Water Res. 38, 111-127.

Noyola, A., Morgan-Sagastume, J.M., López-Hernández, J.E., 2006. Treatment of biogas produced in anaerobic reactors for domestic wastewater: Odor control and energy/resource recovery. Rev. Environ. Sci. Biotechnol., 5, 93-114.

Paing, J., Picot, B., Sambuco, J.P., Rambaud, A., 2000. Sludge accumulation and methanogenic activity in an anaerobic lagoon. Wat. Sci. Technol. 42, 247-255.

Park, J., Craggs, R.J., 2007. Biogas production from anaerobic waste stabilisation ponds treating dairy and piggery wastewater in New Zealand. Wat. Sci. Technol. 55, 257-264.

Peña, M.R., 2002. Advanced primary treatment of domestic wastewater in tropical countries: development of high-rate anaerobic ponds. PhD thesis, The University of Leeds, Leeds: UK.

Peña, M.R., Mara, D.D., 2003. High-rate anaerobic pond concept for domestic wastewater treatment: Results from pilot scale experience. Seminario Internacional sobre Métodos Naturales para el Tratamiento de Aguas Residuales, October 1-3, Cartagena, Colombia, pp. 68.

Peña, M.R., Mara, D.D., Piguet, J.M., 2003. Improvement of mixing patterns in pilot-scale anaerobic ponds treating domestic sewage. Wat. Sci. Technol. 48, 235-242.

Picot, B., Paing, J., Sambuco, J.P., Costa, R.H.R., Rambaud, A., 2003. Biogas production, sludge accumulation and mass balance of carbon in anaerobic ponds Wat. Sci. Technol. 48, 243-250.

Picot, B., Sambuco, J.P., Brouillet, J.L. and Riviere, Y., 2005. Wastewater stabilisation ponds: Sludge accumulation, technical and financial study on desludging and sludge disposal case studies in France. Wat. Sci. Technol. 51, 227-234.

Safley Jr., L.M., Westerman, P.W., 1989. Anaerobic lagoon biogas recovery systems. Biol. Wastes. 27, 43-62.

Shilton, A.N., Mara, D.D., Craggs, R.J., Powell, N., 2008. Solar-powered aeration and disinfection, anaerobic co-digestion, biological CO2 scrubbing and biofuel production: The energy and carbon management opportunities of waste stabilisation ponds. Wat. Sci. Technol. 58, 253-258.

Strauss, M., Larmie, S.A., Heinss, U., 1997. Treatment of sludges from on-site sanitation - Low-cost options. Wat. Sci. Technol. 35, 129-136.

Toprak, H., 1995. Temperature and organic loading dependency of methane and carbon dioxide emission rates of a full-scale anaerobic waste stabilization pond. Water Res. 29, 1111-1119.

Toprak, H., 1994. Empirical modelling of sedimentation which occurs in anaerobic waste stabilization ponds using a lab-scale semi-continuous reactor, Environ. Technol. 15, 125-134.

Water UK, 2011. Sustainability Indicators 2010/2011 http://www.water.org.uk/home/news/press-releases/indicators2010-11/water-uk---sustainability-report-2010-11.pdf (accessed 20/09/12).

Werther, J., Ogada, T., 1999. Sewage sludge combustion. Prog. Energy Combust. Sci. 25, 55-116.

Williams, A. (Ed.), 1994. Methane Emissions, Watt Committee Report No.28, The Watt Committee On Energy, London, UK.

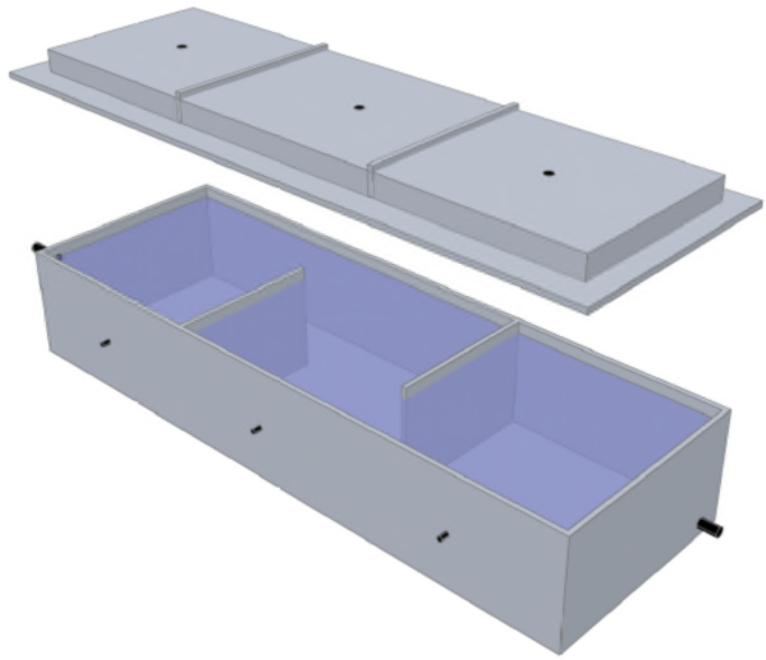

Figure 1. An illustration of the anaerobic pond design used. Baffles were located at L/3 and 2L/3 to limit short circuiting. The lid construction was divided into three sections that aligned with the baffles to enable tracking of gas production along the reactor length. Sampling ports were also sited along the length of the pond.

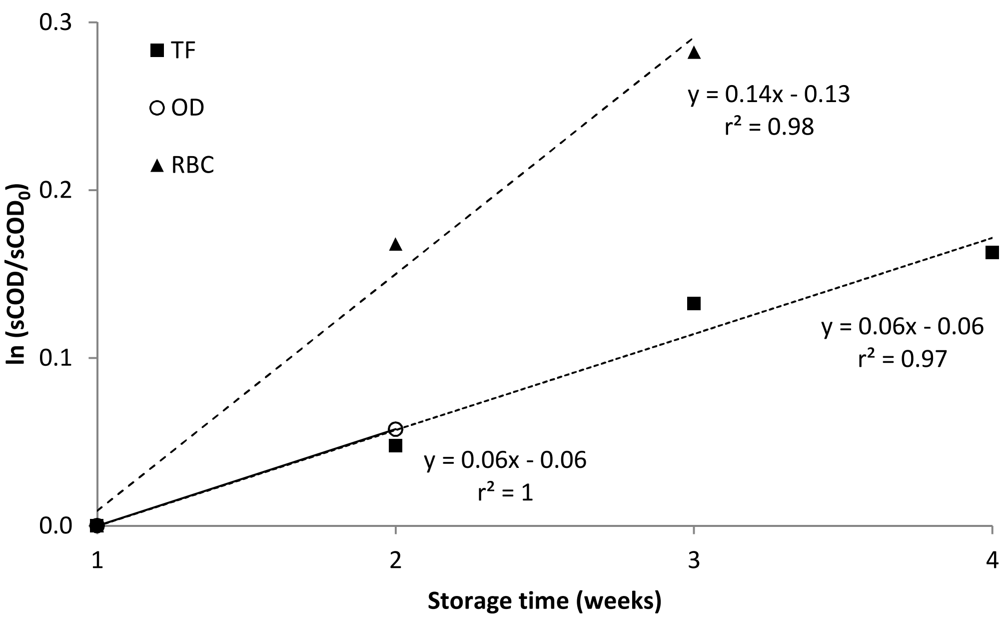

Figure 2. During the initial stage of sludge storage, soluble COD (sCOD) production followed a pseudo first order relationship. Sludge samples collected from holding tanks at three WwTW comprised of a trickling filter (TF), oxidation ditch (OD) or rotating biological contactor (RBC).

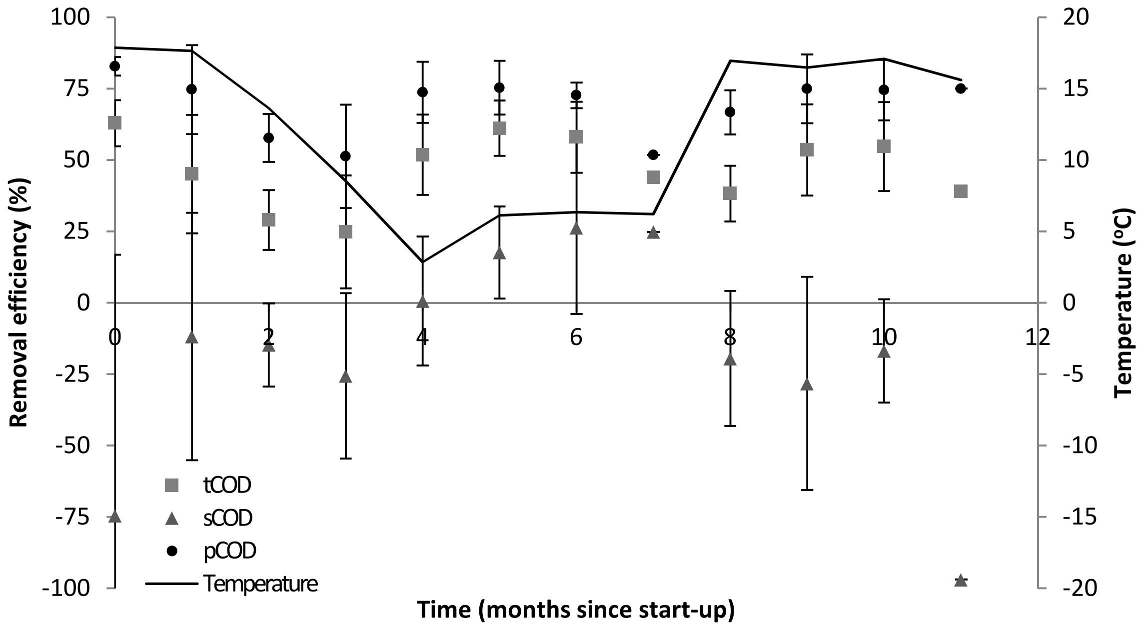

Figure 3. Removal efficiency determined for total COD (tCOD) and soluble (sCOD) and particulate COD (pCOD) fractions in a pilot-scale AP. Data presented comprises monthly average and standard deviation for a 12 month period. Monthly mean ranges: pCOD 51% to 83% (*n*=95, σ=16%); sCOD -75% to 26% (*n*=93, σ=40%). Temperature profile added comprised monthly average temperature.

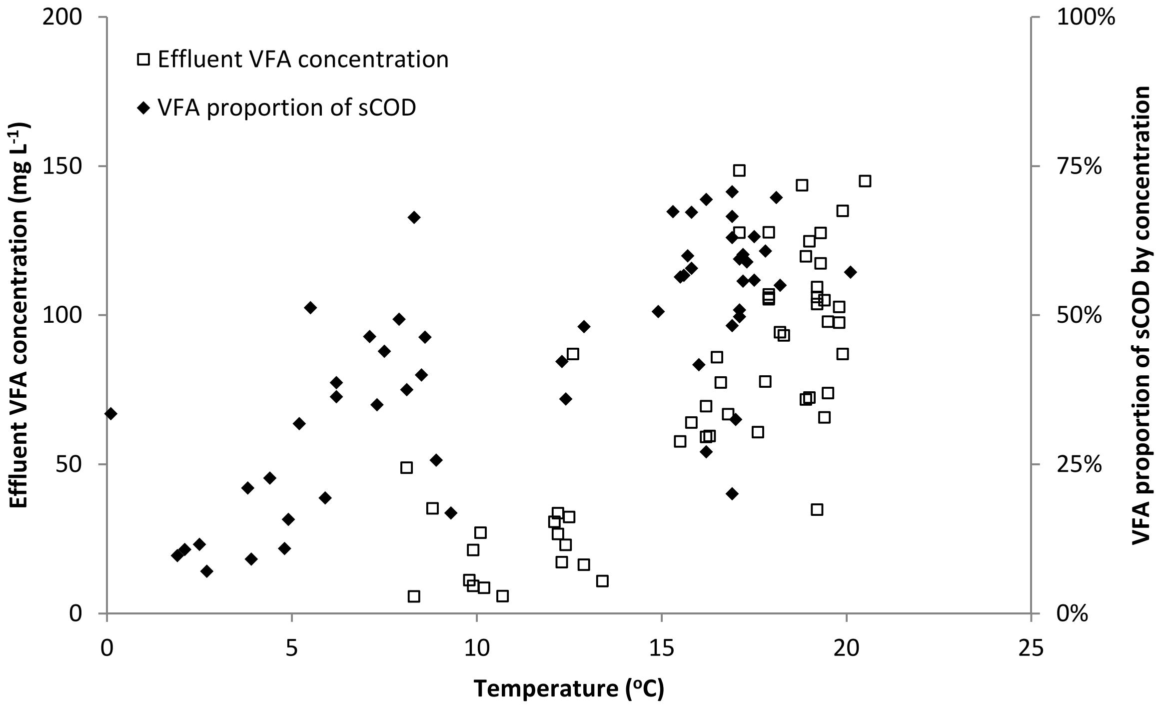

Figure 4. The effect of effluent temperature on effluent volatile fatty acid (VFA) concentration over the full study is presented (*n*=56), also as a proportion of effluent sCOD. Both datasets indicate a weak positive correlation with temperature.

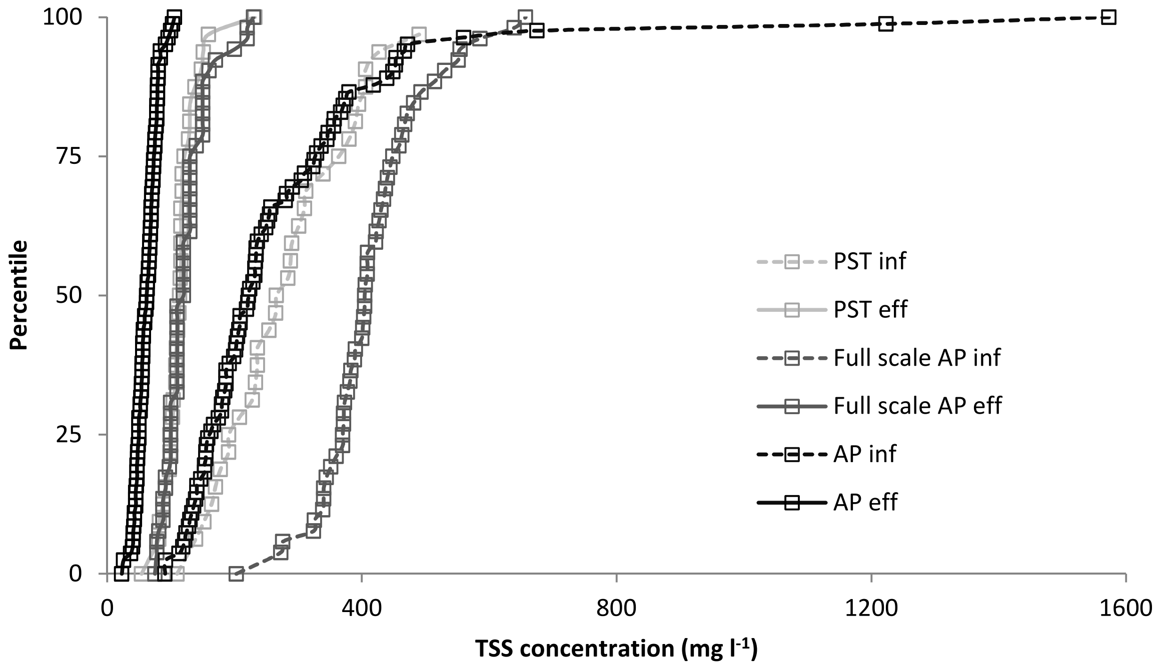

Figure 5. Resilience curves produced from total suspended solids data (TSS) influent and effluent data from this AP study, and compared to resilience curves from a full scale AP in Melbourne, Australia (*n*=52) and a full scale UK primary sedimentation tank PST (*n*=40), both treating domestic wastewater.

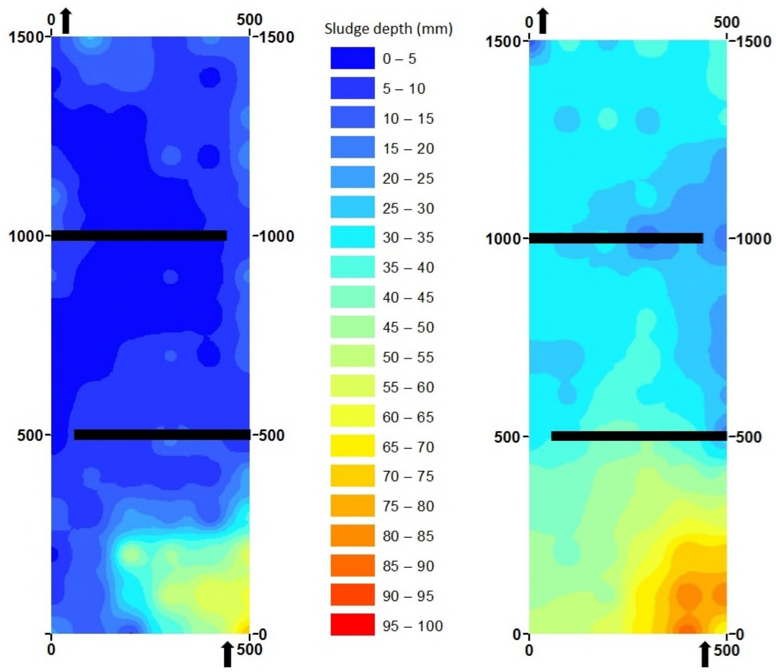

Figure 6. Sludge accumulation map at (left) 219d and (right) 534d, produced from 96 measurements on a 100 mm x 100 mm grid. Whilst high accumulation was observed at the front end of the AP after 219 days (67% of total sludge volume in front third of the AP), more proportionate distribution of sludge volume was note later in the study (47% of total sludge volume in front third after 534 d).

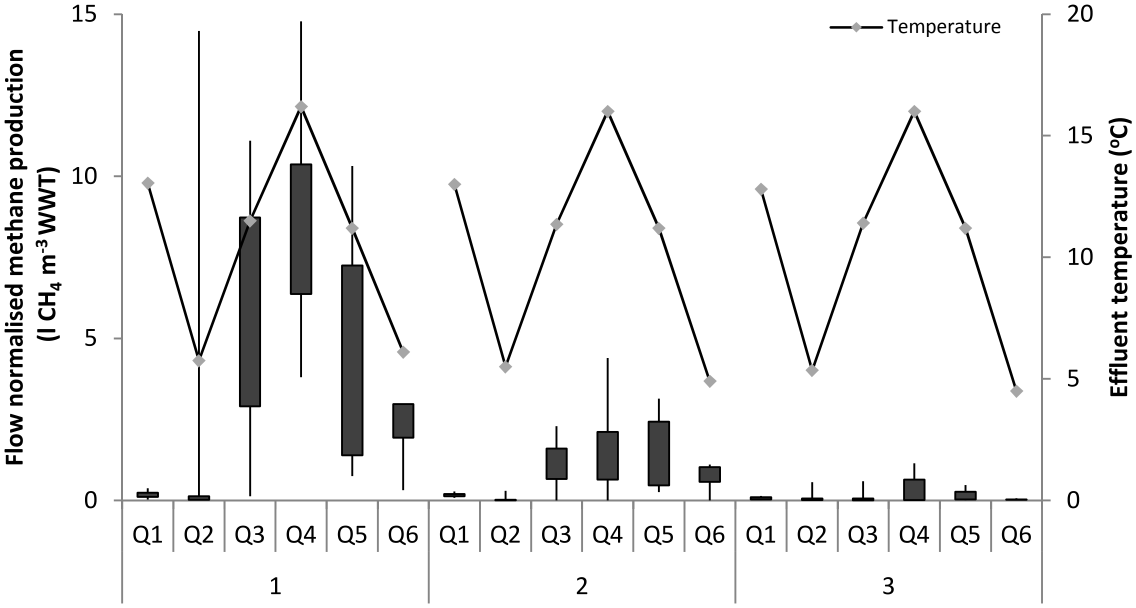

Figure 7. Average methane production recorded along the APs length from inlet (Chamber 1) to outlet (Chamber 3). The time series (Q1 to Q6, *n*=54) demonstrates the methane production per quarter (3 months) in each chamber where Q1 is start up, Q2 and Q6 are winters, and Q4 is the intervening summer. Upper and lower limits of the boxes are 25th and 75th percentiles whilst the whisker ends represent the minimum and maximum values recorded in that time period.

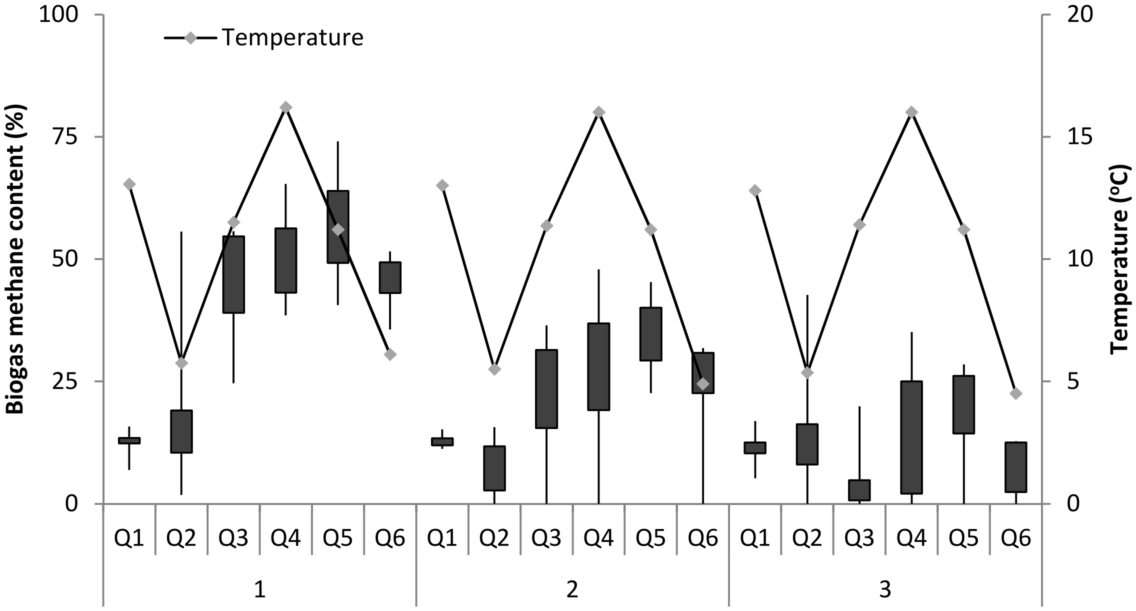

Figure 8. Average biogas methane composition recorded along the APs length from inlet (Chamber 1) to outlet (Chamber 3). The time series (Q1 to Q6, *n*=54) demonstrates the methane biogas composition per quarter (3 months) in each chamber where Q1 is start up, Q2 and Q6 are winters, and Q4 is the intervening summer. Upper and lower limits of the boxes are 25th and 75th percentiles whilst the whisker ends represent the minimum and maximum values recorded in that time period.

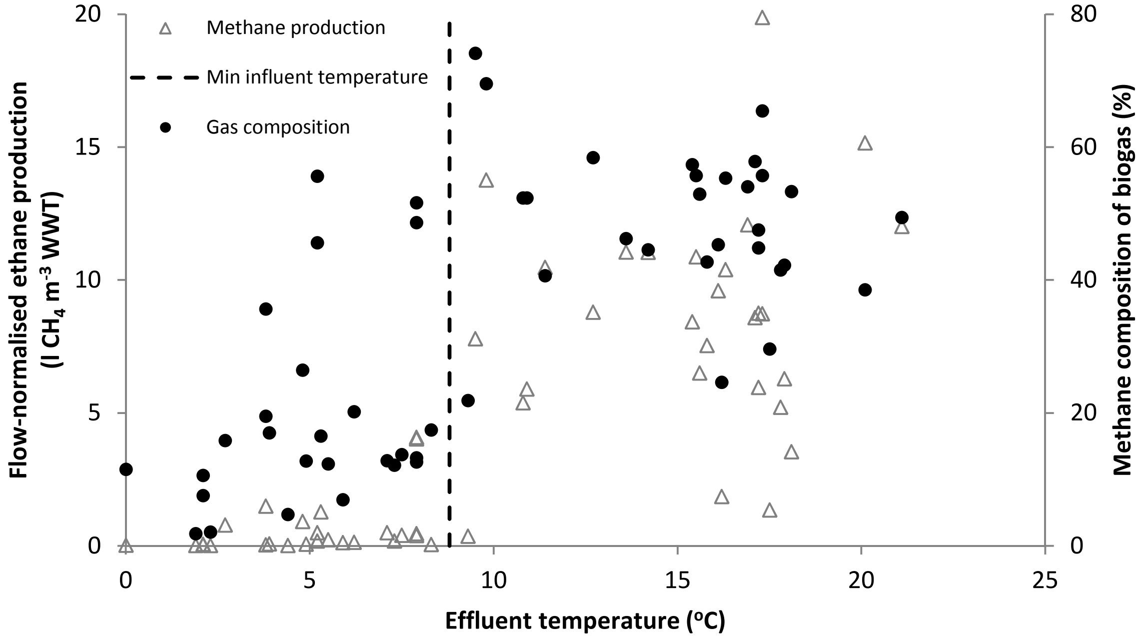

Figure 9. Effect of temperature on normalised methane production and biogas composition (*n*=54). Due to the exposed location of the AP, effluent temperatures were closer to air temperatures than the influent wastewater (minimum 8.8°C, represented by vertical dashed line). Below 8.8 °C mean methane production was 0.62 L CH_4 m^{-3}WWT and above 8.8 °C, 8.48 L CH_4 m^{-3}WWT.