CRANFIELD UNIVERSITY

Peter Cruddas

Anaerobic ponds for domestic wastewater treatment in temperate climates

School of Applied Sciences

PhD Thesis

Supervisors: Dr. Ewan McAdam and Prof. Elise Cartmell
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ABSTRACT

Energy demand, greenhouse gas emissions, and operational costs are continuing to rise year on year in the wastewater treatment sector, with traditional treatment options unable to provide sustainable solutions to increasing volumes and tightening quality standards. Current processes produce inherent fugitive greenhouse gas (GHG) emissions, whilst also generating large quantities of sludge for disposal. Anaerobic ponds (APs) are natural wastewater treatment processes that have traditionally been confined to a pre-treatment stage of larger stabilisation pond systems. Consequently, current standard guidelines are not suited for low temperature, weak strength wastewaters, or for the emerging usage of APs for energy recovery and enhanced organic breakdown. To establish effective guidelines for adapting AP design for this purpose, this thesis explores the fundamental mechanisms with APs, in order to provide design alterations to enhance AP performance for full flow domestic wastewater treatment with a focus on the UK water sector.

Initially, a literature review of current AP design guidelines was conducted to determine the current state of the art and understand the fundamental design processes currently adopted. The review found that most APs are currently underloaded, largely to avoid malodour emissions, but this leads to unnecessarily large footprints and inhibits the digestion process through restricting biomass/substrate contact. It was concluded that the current design guidelines are not suitable for recent AP developments and application, such as covering to prevent odour escape, and the use of baffling to improve mixing and enhance organic degradation.

A pilot scale study was conducted on UK domestic wastewater to gain insight into the limitations of current AP design for this application and identify areas for optimisation. The pilot trial demonstrated the efficacy of AP usage for low temperature, weak strength wastewaters, even with unoptimised design. Decoupling hydraulic and solids retention time lead to biomass retention and subsequent acclimatisation, and was able to compensate for the low temperatures and weak wastewater. It was concluded that APs can provide an attractive alternative to current primary treatment options, through reducing GHG emissions and providing less frequent desludging requirements.

To optimise AP design, the effect of baffle configuration on AP hydrodynamics and the subsequent impact on treatment efficiency was investigated, in order to develop structural designs specifically targeting enhanced anaerobic degradation. Advantages found in baffling APs included improving mixing patterns between baffles, enhancing biomass/substrate contact, and creating an overall plug flow effect through the entire pond enabling the retention of biomass. Furthermore, the removal mechanism with the pond can be manipulated with use of baffles, with different orientations generating different flow patterns and therefore creating conditions preferential for greater solids settlement and capture, or mixing and contact. Following trials on single stage alternate baffling configurations, the development of a novel two stage AP design was trialled, applying knowledge gained from trials of differing baffle orientations to target separate stages of organic breakdown.

Further trials were conducted on the staged AP to establish optimal loading rates to be applied to APs in order to maximise performance and reduce physical footprint. These trials led to recommended design improvements including shorter hydraulic retention times (HRTs) to enhance mixing and decrease physical footprint, and improvements to the staged AP design to greater separate the stages of anaerobic digestion and provide optimal conditions for the stages at different points in the AP.

Finally, the knowledge gained from experimental work was used to present evidence for the inclusion of APs into decentralised WWT through flowsheet modelling of a proposed AP treatment works compared to a current base case. Advantages were found in decreasing sludge management requirements whilst providing suitable primary treatment, with additional potential benefits in renewable energy generation, which could increase both with improved biogas yields and the option of combining with other renewable technologies. In some circumstances, it may be possible for an AP flowsheet to operate entirely off-grid, eliminating the need for costly infrastructure such as permanent access roads and national electrical grid connection.

Keywords:

Waste stabilisation lagoons, methane, biogas, sludge, decentralised works

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First and foremost I would like to acknowledge the Herculean efforts of my long suffering yet eternally patient supervisor, Doctor Ewan McAdam, who whilst accepting that low temperature anaerobic treatment would take substantially longer than standard practices, was surely not anticipating such a similar trend in the production of this thesis. Many thanks for your constant dedication, unwavering support, persistent guidance and occasional brow-beatings to ensure this work was actually completed.

Additional thanks for academic support go to Professor Bruce “the finisher” Jefferson, whose timely intervention was much appreciated in getting over the line, and to Professor Elise Cartmell for ongoing assistance throughout the project. When I was wandering in the darkness of bad CFD practice, Doctors Nick Asproulis and Antonios Antoniadis brought me into the light, as well as the experience of Doctor Andrés Alvarado of the Universiteit Gent/Universidad de Cuenca. A heartfelt thanks to Doctors Gavin Collins and Estefania Porca of the National University of Ireland, Galway for introducing me to the fascinating world of microbial ecology, and to Doctor Miguel Peña of la Universidad del Valle, Colombia, for continued correspondence on the subject of high-rate anaerobic ponds.

For the day to day grind of building, operating, and analysing the pilot plants many thanks to Nigel Janes, Rukhsana Ormesher, Jane Hubble, and Alan Nelson for technical support, and to Laura Borea, Alessandra Marra, and Emilie Pauvret for assistance during their placement projects – I hope your experiences on these pilot plants have not deterred you from the glamour of anaerobic wastewater treatment. Thanks to my industrial sponsors from Halcrow, Nick Pursell, John Martin, and David Best for their valuable insights from the ‘real world’, and to Doctor Catherine Rees at Melbourne Water for collaboration and data sharing, as well as inviting me to visit the Godzilla of anaerobic ponds at Werribee.

Finally, many thanks to all my family and friends who have suffered through this tortuous process alongside me, in particular my partner Ellie who has passed her test in patience, love and support with only minor corrections.

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LIST OF ABBREVIATIONS

ABR	Anaerobic baffled reactor
AD	Anaerobic digestion
anMBR	Anaerobic membrane bioreactor
ANOVA	Analysis of variance
AP	Anaerobic ponds
ASP	Activated sludge process
BOD	Biochemical oxygen demand
CAP	Control anaerobic pond
CAPEX	Capital expenditure
CFD	Computational Fluid Dynamics
CHP	Combined heat and power
CV	Calorific value
DWF	Dry weather flow
EGSB	Expanded sludge blanket reactors
FST	Final sedimentation tank
GHG	Greenhouse gas
HBAP	Horizontally baffled anaerobic pond
HPLC	High performance liquid chromatography
HRT	Hydraulic retention time
HT	Humus tank
LCCA	Life-cycle cost assessment
M&E	Mechanical and electrical
OD	Oxidation ditch
OLR	Organic loading rate
OPEX	Operational expenditure
pCOD	Particulate chemical oxygen demand
PE	Population equivalent
PSD	Particle size distribution
PST	Primary sedimentation tank
qPCR	Quantitative polymerase chain reaction

RBC	Rotating biological contactor
RDS	Raw dry solids
RTD	Residence time distribution
SAP	Staged anaerobic pond
sCOD	Soluble chemical oxygen demand
SRT	Solids retention time
tCOD	Total chemical oxygen demand
TF	Trickling filter
TSS	Total suspended solids
UAP	Unbaffled anaerobic pond
UASB	Upflow anaerobic sludge blanket reactor
uPVC	Unplasticised polyvinyl chloride
SHT	Sludge holding tank
SMA	Specific methanogenic activity
VBAP	Vertically baffled anaerobic pond
VFA	Volatile fatty acid
VS	Volatile solids
VSS	Volatile suspended solids
WSP	Waste stabilisation pond
WWT	Wastewater treatment
WWTW	Wastewater treatment works

1 Introduction

1.1 Sustainability drivers in the wastewater treatment sector

Energy conservation, reduction in greenhouse gas (GHG) emissions and the promotion of renewable energies are key drivers for the water industry (Water UK, 2012). Energy use in the water sector has been increasing year on year (Figure 1-1) as the industry faces challenges in treating higher volumes, meeting tighter quality standards, and safe guarding against the effects of climate change (Water UK, 2012), and UK water utilities consume approximately 3 % of net UK electricity (Environment Agency, 2009). In wastewater treatment (WWT), energy is predominantly utilised in aeration systems (*c.* 55 %) to facilitate organic carbon biodegradation in the activated sludge process to CO_2 (Tchobanoglous et al., 2003). Whilst renewable energy generation has been increasing within the industry in recent years (Figure 1-1), more must be done to increase this proportion from the current 10 % of energy used to meet the UK target of 20 % by 2020 (Water UK, 2012).

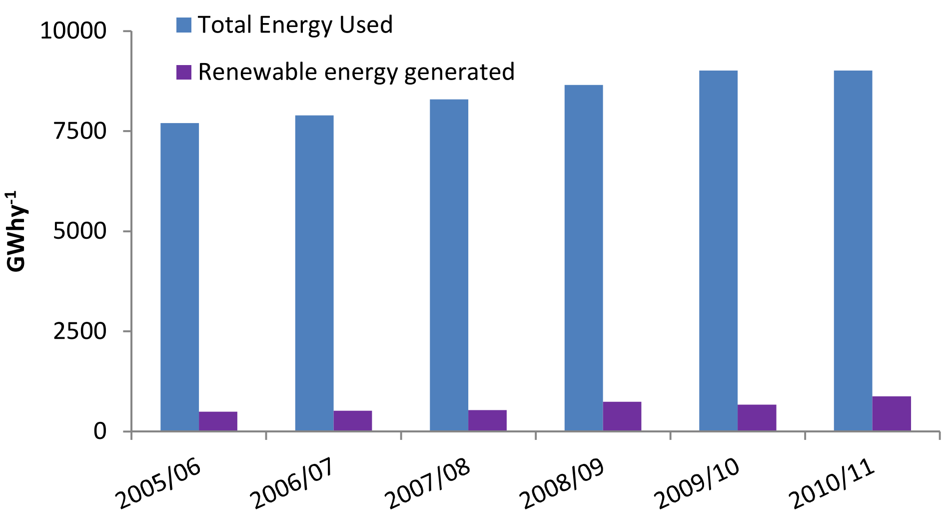

Figure 1-1 Total energy used and renewable energy generated by the UK water industry in the reporting period 2005 – 2011 (most recent data, source: Water UK)

Increased energy demand is connected to a substantial rise in GHG emissions, against the overall UK trend of GHG reductions (Figure 1-2). In particular, wastewater treatment has seen a significant rise in GHG emissions per ML WWT treated, which has been attributed to the need for increasingly energy-intensive treatment technologies to meet more stringent quality standards, whilst it has been recognised that compliance to the Water Framework Directive may have the potential to exacerbate emissions rises in the future (Water UK, 2009). The UK became the first country in the world to set legally binding GHG emissions cuts in the Climate Change Act 2008, with an ambitious target of 80 % reductions from 1990 levels by 2050, and many UK water companies are setting emissions reductions targets to contribute towards the national goal (OFWAT, 2010). A key area in reducing the energy demand and GHG emissions from WWT is at small, decentralised works. Approximately 79 % of WWT works in the UK treat population equivalents (PEs) of less than 2,000 (DEFRA, 2012), whilst across the EU 80 % of WWT works are <5,000 PE (Alexiou and Mara, 2003). Aeration comprises around 55 % of the electrical demand of wastewater treatment (Tchobanoglous et al., 2003), and due to inefficiencies at small scale, even technologies such as aerated wetlands are comparable to the traditional activated sludge process in electrical demand per PE treated (Pearce, 2013).

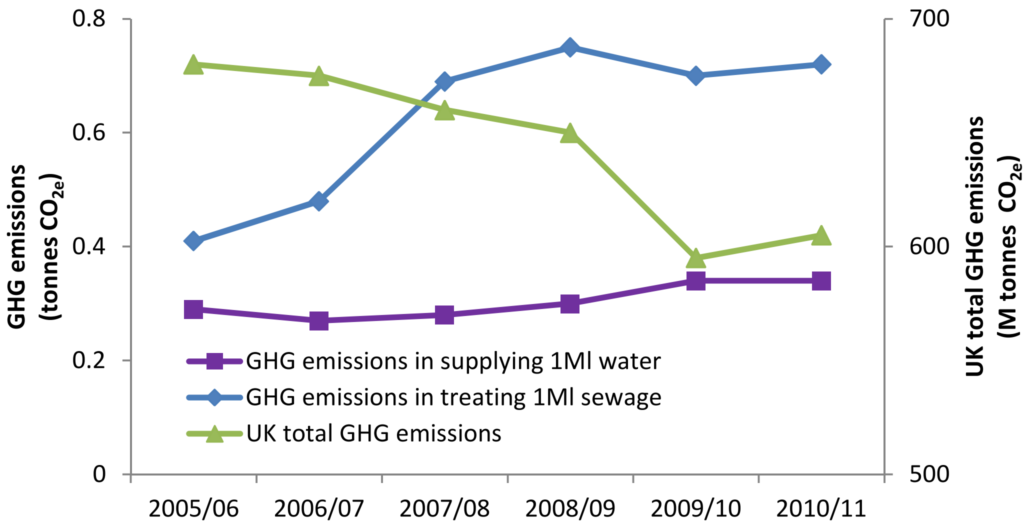

Figure 1-2 Greenhouse gas (GHG) emissions from the UK water sector and the UK as a whole in the reporting period 2005-2011 (most current data, source: Water UK)

The use of anaerobic digestion (AD) has been identified as the principle source of renewable energy generation and GHG emissions reductions, and in 2009 over 90 % of the renewable energy generation within the sector was from sludge combustion and digestion (Environment Agency, 2009). However, AD is currently only practiced at larger scale sites, and only 148 of the >9,000 WWT works in the UK operate anaerobic digesters (Anaerobic digestion portal, 2013). For small decentralised works, tankering sludge to centralised AD plants incur emissions in transportation, whilst fugitive emissions from the sludge storage before transport increase air pollution whilst decreasing the energy value of the sludge (McAdam et al., 2012). Therefore, on-site sludge management and renewable energy at decentralised WWT works could make a substantial contribution to achieving the water industry's sustainable development targets.

1.2 Anaerobic ponds

Anaerobic ponds (APs) are natural wastewater treatment processes that induce anaerobic conditions through loading rates that preclude aerobic activity, and provide primary treatment through degrading particulate carbon to methane. Whilst APs have traditionally been confined to a pre-treatment stage of a larger stabilisation pond system (Alexiou and Mara, 2003), recently they have been combined with other post-treatment process such as trickling filters (Broome et al., 2003). Their main advantages are low capital and maintenance costs, limited requirement for skilled personnel, and the ability to withstand hydraulic and organic shock loading (Alexiou and Mara, 2003). In the early 2000s, both research activity and full scale AP systems saw significant growth, as the water industry recognises the value of both reducing energy demand through utilising natural treatment processes where practical, and capturing the methane rich biogas produced by ponds as a source of renewable energy. However, this was almost exclusively focused on tropical climates, with few studies reported at low temperature. Furthermore, research activity in APs has slowed in the last decade, whilst over the same time period full-flow anaerobic treatment of domestic wastewaters has continued to gain momentum, and systems such as anaerobic MBRs are now accepted as a feasible technology, even at low liquid temperatures (Martin Garcia et al., 2013). For APs to join the growing portfolio of low temperature, full flow anaerobic treatment processes, design enhancements are required to both intensify the process to increase organic strength, and to optimise anaerobic degradation to compensate for the low temperature.

Whilst APs are already an established technology in countries such as New Zealand (Archer and Mara, 2003), India (Sato et al., 2007) and France (Racault and Boutin, 2005), they have never been used on domestic wastewater in the UK. This can be largely attributed to the perception that land requirements and poor performance at low temperature prohibit their use, and advancements in both these areas have been subdued by this historical preconception. The introduction of APs to decentralised WWT works would significantly reduce the energy demands of these facilities, through low inherent process energy demand, and small quantities of sludge generated. If APs are used as a primary treatment stage to a passive aerobic treatment process, such as a trickling filter or constructed wetland, only a small amount of biogas would have to be recovered from the AP in order to make the entire works energy-neutral.

1.3 Aim and objectives

This thesis investigates the fundamental mechanisms within APs, in order to provide design alterations to enhance AP performance for full flow domestic wastewater treatment with a focus on the UK water sector. Domestic UK wastewaters are characterised by a dilute organic concentration due to combined sewerage, and low temperature (mean *ca.* 12°C). These present a potentially significant barrier to effective AP treatment since both organic substrate concentration and temperature can be directly correlated to anaerobic microbial growth and the kinetic rate of anaerobic organic biodegradation (Weiland and Rozzi, 1991; Lettinga et al., 2001; Lew et al., 2009). A better understanding of the processes inside APs is needed in order to develop an engineering approach that will produce a design to optimise the anaerobic degradation process in these conditions. For temperate conditions, specific AP design principles require investigation since effective operation at low temperature is more heavily dependent upon avoiding bacterial washout and maximising organic retention to assure anaerobic microbial consortia are capable of effective methanogenesis at the lower kinetic rates.

The aim of this thesis is to establish effective guidelines for AP design to deliver enhanced methane recovery and sludge management on low temperature domestic wastewater (Figure 1-3).

To achieve this aim a series of objectives were identified:

1. A comprehensive literature review of current AP design guidelines to determine the current state of the art and understand the fundamental design processes currently adopted
2. A pilot scale study of an AP operating on UK domestic wastewater to gain insight into the limitations of current AP design for this application and identify potential areas for optimisation
3. Determine the effect of baffle configuration on AP hydrodynamics and the subsequent impact on treatment efficiency, to develop structural designs specifically targeting enhanced anaerobic degradation
4. Establish optimal loading rates to be applied to APs in order to maximise performance and reduce physical footprint
5. Utilise the knowledge gained from experimental work to present evidence for the inclusion of APs into decentralised WWT through flowsheet modelling of a proposed AP treatment works compared to a current base case

1.4 Thesis structure

The thesis takes the form of a series of chapters formatted in the style of journal papers (Table 1-1). All chapters were written by Peter Cruddas, and have been edited by Dr. Ewan McAdam. All pilot scale trials, associated laboratory analyses, and computational fluid dynamics (CFD) modelling were carried out at Cranfield University by Peter Cruddas, with support in the sampling and analysis during the pilot trials from Laura Borea, Alessandra Mara, and Emilie Pauvret as part of their placement requirements. Specific methanogenic activity tests and quantitative Polymerase Chain Reaction (qPCR) assays were carried out by Peter Cruddas with assistance from Dr

Gavin Collins and Dr Estefania Porca during visitation to the National University of Ireland, Galway.

Table 1-1 Thesis structure and journal submission plan for each chapter

Chapter	Objective addressed	Title	Target journal	Status
2	1	Anaerobic waste stabilisation ponds: The need for a fresh design approach	Environmental Technology	In preparation
3	2	Diagnosis of an anaerobic pond treating temperate domestic wastewater: An alternative sludge strategy for small works	Ecological Engineering	Published
4	3	Development of a staged anaerobic pond design through pilot trials and computational fluid dynamics	Environmental Engineering	In preparation
5	4	Performance of a two stage anaerobic pond at four hydraulic retention times	Water Research	In preparation
6	5	Incorporating anaerobic ponds into decentralised wastewater treatment	Ecological Engineering	In preparation

A literature review was conducted to assess the current state of the art of APs, identify standard design methods and practices and highlight knowledge gaps and areas for development in order to improve AP design. This literature review challenges the current design assumptions of APs and provides the basis for further investigation of some of the key AP processes that are currently not well understood. This review, entitled *Anaerobic waste stabilisation ponds: The need for a fresh design approach*, comprises chapter 2 of this thesis and is in preparation for submission to the journal Environmental Technology.

Chapter 3 is an assessment of the efficacy of a pilot scale AP treating UK domestic wastewater, as APs had not previously been trialled under these conditions. In addition to providing a comprehensive data set for AP operation in this environment, diagnosis of AP processes was also conducted to inform design decisions later in the project, and support the findings and assumptions of Chapter 2. This chapter has been published in Ecological Engineering under the title *Diagnosis of an anaerobic pond treating temperate domestic wastewater: An alternative sludge strategy for small works*.

Chapter 4 explores differing baffle configurations in order to investigate whether the manipulation of flow characteristics through baffle design significantly affects the performance and nature of an AP. The findings were used to develop a two stage AP design, with support from literature describing high rate anaerobic WWT processes that have also developed two stage configurations. The baffle designs are assessed both through the use of CFD modelling with validation from experimental tracer studies, and through pilot trials on domestic wastewater. This chapter is being prepared for submission to Environmental Engineering under the title *Development of a staged anaerobic pond design through pilot trials and computational fluid dynamics*.

Chapter 5 uses the two stage AP design developed in Chapter 4 to investigate the influence of hydraulic retention time on AP operation, with comparison made between the two stage AP and a control AP identical in design to the baseline study in Chapter 3. Flow characteristics at the increasing flow rates induced by shorter HRTs are studied through the use of CFD modelling, whilst pilot scale trials studied key AP performance indicators, such as biogas production, sludge accumulation, and removal efficiency of sanitary parameters. Specific methanogenic activity and qPCR assays were conducted on sludge samples taken from throughout both APs at the end of the study to determine any changes in microbial community profiles and activity with spatial change within the APs, both between stages in the two stage design and within the individual stages. This chapter is being prepared for submission to Water Research under the title *Performance of a two stage anaerobic pond at four hydraulic retention times*.

The overall implications of the research are presented in Chapter 6, and are then contextualised through comparison of model flowsheets typical for a decentralised WWT works. A flowsheet incorporating an AP, using performance data from the project, is compared with a current standard design flowsheet, to highlight the suitability of APs for use in decentralised WWT, and comment on their relative strengths and limitations within this scenario. Concluding remarks and recommendations for further work are provided in Chapter 7.

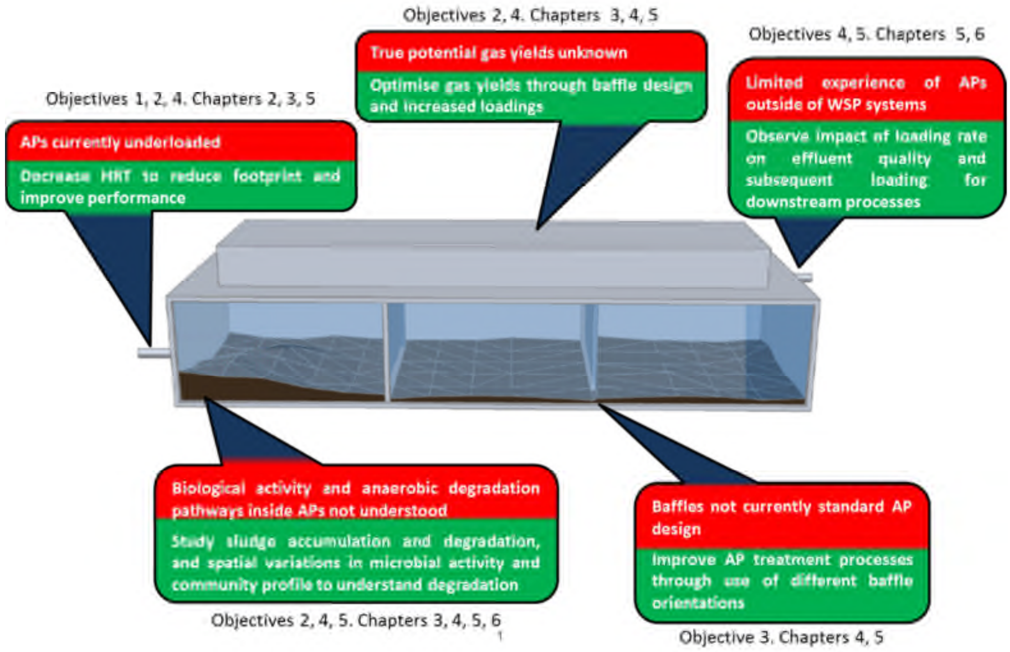

Figure 1-3 Conceptual diagram of the thesis structure

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2 Anaerobic waste stabilisation ponds: The need for a fresh design approach

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Abstract

Anaerobic ponds (APs) are an attractive wastewater treatment option due to their low energy requirement and potential for renewable energy generation through biogas production. Increased research output in the last two decades has led to many advances in AP process understanding and physical design, although these have not always been reflected in design guidelines. It is identified that most APs are loaded below their optimal rates, primarily due to concerns over malodours, and that current design practice of pond sizing by temperature is inaccurate and excessively conservative. Furthermore, whilst seasonal variations in accumulated sludge and biogas bubbling in the pond are likely to have a significant effect on pond performance, these aspects are not currently considered at the design stage. It is proposed that new guidelines are developed, recommending loading rates by process performance rather than odour avoidance, and incorporating dynamic pond processes such as sludge accumulation and biogas production which are currently ignored.

Keywords: Lagoons, anaerobic digestion, methane, sustainable technology

2.1 Introduction

Rising fossil fuel prices and the apparent transition toward process carbon accounting has encouraged the water industry to prioritise process sustainability when managing existing and new assets (Muga and Mihelcic, 2008). Accordingly, the development of low energy wastewater flowsheets is an emerging priority area (Brookes, 2013), with particular emphasis on extensive treatment processes such as waste stabilisation ponds (WSPs) (Shilton et al., 2008) and constructed wetlands (Moir, 2013). This has been combined with the emergence of intensified anaerobic processes, such as the upflow anaerobic sludge blanket reactor (UASB) and anaerobic membrane bioreactor (anMBR) to support full-flow ambient temperature anaerobic treatment as a feasible alternative flowsheet to energy intensive aerobic processes (Lettinga et al., 2001). By contrast, anaerobic waste stabilisation ponds (or anaerobic ponds, APs) complement both aspects of the above proposed technologies as they are an extensive process characterised by a low energy demand, which also affords the potential for methane generation and hence energy recovery. Furthermore, the degradation of solids within the process generates little excess sludge, meaning desludging operations are only required every two to four years (Alexiou and Mara, 2003; Papadopoulos et al., 2003; Konate et al., 2010). Coupled with low maintenance requirements, APs offer substantial reductions in both operational costs and carbon emissions from current treatment options, especially for remote facilities (McAdam et al., 2012). Anaerobic ponds have been widely used as a pre-treatment stage in full-flow wastewater treatment since the 1950s (Pescod, 1996). However, as an extensive technology, APs have often been overlooked as the perception is that of a ‘low-tech’ solution (Pearson, 1996) characterised by a prohibitively large footprint (Agunwamba, 2001), which can partly be attributed to unnecessarily low loading rates. Whilst empirical guidance was developed for the design of APs, the authors recognised in their development that the proposed bounds were conservative, which was in part to reduce the odour emissions, as the early system design did not consider gaseous recovery (Mara and Pearson, 1998).

In the early 2000s, it was recognised that many fundamental design parameters had not been sufficiently investigated (Picot et al., 2003; Alexiou and Mara, 2003; Peña et al., 2000). Subsequently, developmental research was undertaken with particular emphasis on identifying appropriate design geometry to maximise removal performance and reduce process scale (Vega et al., 2003; Agunwamba, 2006). Soon after, the covering of APs was recommended for environmental protection (Noyola et al., 2006) and energy capture in various industrial applications (Park and Craggs, 2007). These advancements, and the preliminary energy balances demonstrated at full scale for the inclusion of this technology, stimulated interest in this technology as a significant contributor to sustainable wastewater treatment (Shilton et al., 2008). However, despite the increase in research activity, there remains no consensus on appropriate design boundaries, and work towards this has slowed considerably in recent times. The aim of this review is to critically evaluate the current state–of-the-art in AP design, considering the applicability of current design guidance and finally positing a new set of design guidance for application to energy recovery in wastewater treatment based on research conducted to date.

2.2 Evaluating current anaerobic pond design with supporting empirical approaches

2.2.1 The influence of organic loading rates for APs

Mara and Pearson (1998) specified an empirical approach for AP design based on sizing the physical dimensions of the pond to attain the desired removal performance within a set of limiting environmental conditions. Limiting environmental factors include raw wastewater quality and mean ambient temperature (which is fixed at the coldest month). Consequently, volumetric organic loading rate (OLR) is applied as the design metric, which is a function of the raw wastewater organic strength and the empty pond volume (Table 2-1).

Table 2-1 Relation of temperature to BOD removal in APs, from Mara and Pearson (1998)

Temperature (°C)	Volumetric BOD loading (g m^{-3} d^{-1})	BOD removal (%)
<10	100	40
10-20	20T-100	2T+20
20-25	10T-100	2T+20
>25	350	70

where T is the temperature, in °C

Following data analysis of uncovered APs at full-scale, the authors suggested OLR boundaries to reside between a lower OLR of 100 g biochemical oxygen demand (BOD) m^{-3} d^{-1} and an upper OLR of 350 gBOD m^{-3} d^{-1} (Mara and Pearson, 1998). The lower limit was prescribed to ensure anaerobic conditions whilst the upper limit was assigned to minimise the diffuse release of odorous gases. Organic loading rate data, based on BOD, was collated from all published AP studies where BOD data was reported (Figure 2-1).

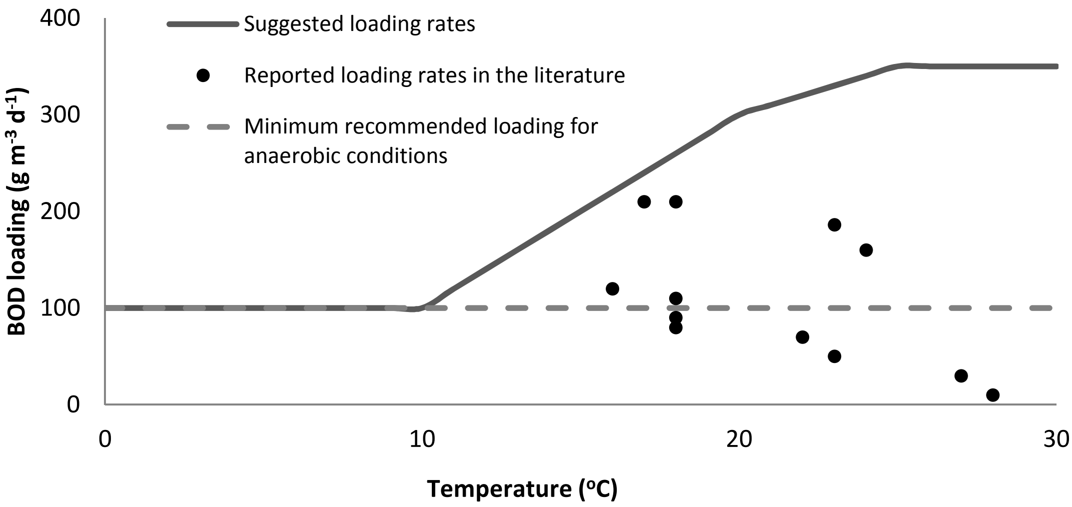

Figure 2-1 Reported BOD loading rates from the literature with respect to temperature, including the recommended design values from Mara and Pearson (1998)

Interestingly, when compared to the proposed empirical design criteria, all of the APs surveyed were below the recommended design value, whilst half were below the 100 gBOD m^{-3} d^{-1} loading rate recommended for anaerobic conditions. Odour release is cited most frequently as the principal operational problem with APs (Pearson et al., 1996; Picot et al., 2005a; Archer and Mara, 2003; Alexiou and Mara, 2003).

Table 2-2 Removal performance data reported for anaerobic ponds for the treatment of domestic wastewater

Reference	Location	Vol	Q	HRT	T	tCOD					BOD					TSS				
						OLR	Inf.	Eff.	Removal		OLR	Inf.	Eff.	Removal		OLR	Inf.	Eff.	Removal	
		m^3	$m^3\,d^{-1}$	d	°C	$kg\,m^{-3}\,d^{-1}$	$mg\,l^{-1}$	$mg\,l^{-1}$	%	$g\,m^3_V\,d^{-1}$	$kg\,m^{-3}\,d^{-1}$	$mg\,l^{-1}$	$mg\,l^{-1}$	%	$g\,m^3_V\,d^{-1}$	$kg\,m^{-3}\,d^{-1}$	$mg\,l^{-1}$	$mg\,l^{-1}$	%	$g\,m^3_V\,d^{-1}$
Alabaster et al. (1991) [1]	Kenya	4256	7056	0.6	24	3.59	2165	1162	46	1663	1.82	1100	553	50	907	N/a	N/a	N/a	N/a	N/a
Alabaster et al. (1991) [1]	Kenya	2128	5177-5596	0.4	24	1.68-3.26	691-1239	590-853	15-31	246-1015	0.85-1.65	351-629	280-406	20-35	173-586	N/a	N/a	N/a	N/a	N/a
Peña (2002)	Colombia	82	78-156	0.5-1.1	25-27	0.54-1.18	590-600	197-312	48-67	187-572	N/a	N/a	N/a	N/a	N/a	0.24-0.64	267-321	N/a	N/a	N/a
Peña (2002)[2]	Colombia	104	95-199	0.5-1.1	25-27	0.54-1.18	590-600	124-137	77-79	125-237	N/a	N/a	N/a	N/a	N/a	0.24-0.64	267-321	N/a	N/a	N/a
Peña (2002)[3]	Colombia	88	86-173	0.5-1.0	25-26	0.60-1.18	590-600	198-289	51-67	204-568	N/a	N/a	N/a	N/a	N/a	0.27-0.64	267-321	N/a	N/a	N/a
Alexiou (2003)	Greece	1	1	1.0	18	0.83	832	430	48	402	0.21	213	136	36	77	0.48	484	213	56	271
De Oliveira et al. (1996)	Brazil	3.24	2.16-3.24	1.0-1.5	23-24	0.34-0.50	502-508	183-298	41-64	140-319	0.16-0.19	186-240	35-92	62-81	99-151	0.20-0.28	283-298	58-74	75-80	149-225
Pearson et al. (1996)	Kenya	11475	4590	2.5	17	0.30	745	153	79	237	0.21	537	95	82	177	0.14	347	71	79	110
Hodgson and Paspaliaris (1996)[1]	Australia	90000	60000	1.5	N/a	N/a	N/a	N/a	N/a	N/a	0.30	450	170	62	187	N/a	N/a	N/a	N/a	N/a
Papadopoulos et al. (2004)	Greece	570	150	3.8	N/a	0.24	930	400	57	139	0.10	384	177	54	54	0.16	603	192	68	108
Papadopoulos et al. (2003)	Greece	570	120-150	3.8-4.8	18	0.20-0.24	907-947	365-461	51-60	102-143	0.09-0.11	427-444	191-241	44-57	39-67	0.12-0.13	504-594	170-217	63-66	79-89
Alabaster et al. (1991)[1]	Kenya	32396	4908-6072	5.3-6.6	24	0.13-0.20	844-1249	413-578	48-54	95-102	0.06-0.10	421-623	146-204	63-67	62-64	N/a	N/a	N/a	N/a	N/a
Parissopoulos et al. (2003)	Greece	570	120	4.8	N/a	0.18	860-870	403-501	42-53	78-96	0.08-0.10	387-456	213-330	28-45	27-37	0.12-0.14	565-657	161-275	58-72	80-85
Toprak (1995)	Portugal	6080	1337	5.1	N/a	0.17	699	299	58	88	N/a	N/a	N/a	N/a	N/a	0.07	375	N/a	N/a	N/a
Broome et al. (2003)	Zimbabwe	22500	5626	4.0	N/a	0.15	603	179	70	106	0.10	400	120	70	70	N/a	N/a	N/a	N/a	N/a
McAdam et al. (2012)[4]	U.K.	0.17	0.07	2.3	22	0.14	318	212	33	44	0.07	152	99	35	22	0.07	154	94	39	25
Paing et al. (2000)	France	5108	884	5.0	20	0.12	685	N/a	N/a	N/a	N/a	N/a	N/a	N/a	N/a	0.06	291	N/a	N/a	N/a
Picot et al. (2003)	France	5000	1,087	4.6	18	0.12	589	462	22	28	0.08	400	280	30	26	0.06	256	114	55	31
Paing et al. (2003)	France	5000	1036	4.6	18	0.12	589	462	22	26	N/a	N/a	N/a	N/a	N/a	0.06	256	114	55	29
Picot et al. (2005a)	France	14260	3127	4.6	N/a	0.12	557	369	34	41	0.08	347	187	46	35	0.06	256	158	38	21
El-Deeb Ghazy (2008)	Egypt	1400	225	6.2	23	0.09	556	403	29	26	0.05	294	229	22	10	0.05	283	214	24	11
De Oliveira (1990)	Brazil	74	9-15	5.0-8.0	27-28	0.04-0.08	362-407	187-202	48-50	21-42	0.01-0.03	119-154	59-75	50-51	7-16	0.02-0.05	172-242	40-43	75-83	16-41
De Garie et al. (2000)[1]	Australia	1440000	240000	6.0	N/a	N/a	N/a	N/a	N/a	N/a	0.07	400	120	70	47	N/a	N/a	N/a	N/a	N/a
Dhariwal and Soni (2008)	India	N/a	5000	N/a	24	N/a	N/a	N/a	N/a	N/a	N/a	151	76	50	N/a	N/a	335	134	60	N/a

Vol – Pond volume. Q – Liquid flow. HRT – Hydraulic retention time. T – Temperature. tCOD – total chemical oxygen demand. BOD – biochemical oxygen demand. TSS – total suspended solids. OLR – Organic loading rate. Inf. – Influent concentration. Eff. – Effluent concentration. N/a – Not available. [1]Domestic and industrial feedwater. [2]Mixing pit. [3]Horizontally baffled. [4]Vertically baffled

Consequently, in practice, the prescribed OLR is expected to have been adopted primarily to minimise odour promulgation rather than to optimise removal (Mara and Pearson, 1998; Papadopoulos et al., 2004; Pearson et al., 1996). Odour from APs is generated by the production of hydrogen sulphide and VFAs, which are by-products in the AD process that increase with higher activity (Noyola et al., 2006). Therefore, the reduction of OLR to mitigate odour release intentionally limits the AD process in order to limit these by-products. Importantly, the motivation for installing APs today is very different, with energy recovery being a primary driver for uptake. Consequently, where ponds are covered for energy recovery, the release of diffuse odour emissions is abated and where necessary, odour can be treated at a point source (Craggs et al., 2008; DeGarie et al., 2000; Hudson et al., 2008).

Analytical data for BOD and total suspended solids (TSS) from the same studies established a positive correlation between loading rate and mass removed (Figure 2-2). Specifically, as OLR is increased, higher mass removal rates are achieved, with the higher removal rates approaching the proposed upper limit of 350 gBOD m^{-3} d^{-1}. Whilst there will inevitably be an upper limit to loading rate, at which point overloading will cause a decline in removal rate (Toprak, 1994), these data suggest that maximum removal performance has not been achieved within the current operational envelope, thus greater removal performance may be realised at higher loading rates.

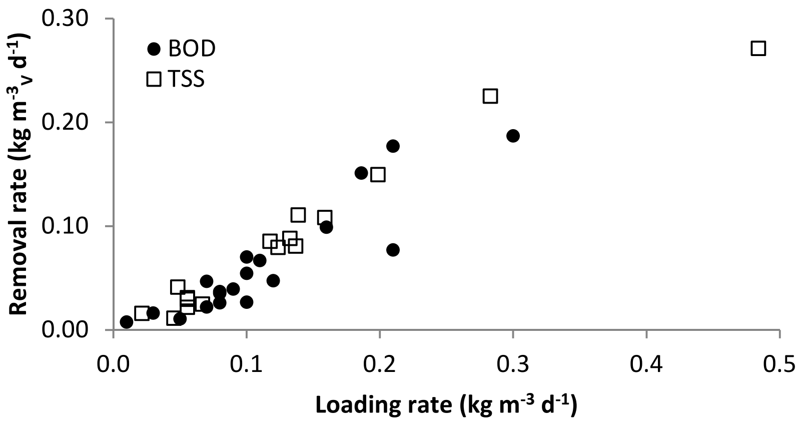

Figure 2-2 Reported BOD and TSS removal rates, normalised for pond volume, against loading rates

2.2.2 The influence of hydraulic retention time on anaerobic pond removal efficiencies

Once the OLR has been determined, the hydraulic retention time (HRT) can be specified since HRT is intrinsically linked to loading rate (adapted from Mara and Pearson, 1998):

$$\lambda_V = \frac{L_i}{\Theta} \qquad \textbf{Equation 2-1}$$

where λ_V is the volumetric organic loading rate (OLR), in g m^{-3} d^{-1}; Q is the flow rate, in m^{-3} d^{-1}; L_i is the influent concentration, in g m^{-3}; and, Θ is the HRT, in d. Anaerobic ponds are primarily used as a primary treatment stage, thus removal efficiency is the most common indicator of performance. Mara and Pearson (1998) include design values for approximate BOD removal efficiency in APs of between 40 % and 70 % for operating temperatures of between 10 °C and 25 °C (Table 2-1). As APs are a passive system and do not actively introduce mixing, the HRT provides the main parameter that governs contact between the raw wastewater and the active biomass (Shilton et al., 2000). Consequently, with the traditional design perspective, a minimum HRT of one day has been suggested (Mara and Pearson, 1998). However, in a comparison of published data on organic removal rates with loading rate (Figure 2-2), higher removal rates are observed at lower HRTs with the maximum removal rates recorded at 1 d for TSS and 1.5 d for BOD. This is intuitive based on the previous correlation established between OLR and removal rate (Table 2-2) since HRT is ostensibly proportional to OLR (Equation 2-1). Interestingly, in a pilot scale study, Peña and Mara (2003) reported 79 % total chemical oxygen demand (tCOD) removal efficiency for an AP with an HRT of 0.5 d, which supports the promise of a lower HRT; however, the authors also attributed the high removal to the hydraulic design, which incorporated a mixing pit at the inlet for enhanced biomass-substrate contact.

2.2.3 Relationship of temperature to anaerobic pond removal efficiency

The link between operational performance and temperature has been commonly reported (Dhariwal and Soni, 2008; Mara and Pearson, 1998; Peña and Mara, 2004;

Gloyna, 1971; Sáenz Forero, 1993). Toprak (1995b) established a linear regression comprised of two empirical constants:

$$E = -1.384 + 2.688(T_{l,e}) \qquad \textbf{Equation 2-2}$$

where E is the tCOD removal efficiency, in %; and, $T_{l,e}$ is the effluent liquid temperature, in ^{o}C. This model assumes that below 12 °C, sedimentation is the primary removal mechanism, yielding a consistent 30 % tCOD removal. Similarly, Mara and Pearson's (1998) model also assumed limits; a 40 % BOD removal threshold at 10 °C which assumes zero biological activity such that sedimentation is the dominant removal mechanism; and, a 70 % BOD threshold at 25 °C where biokinetics are assumed not to be rate limiting. The removal performance between these temperature limits assumes a positive linear function which is proportional to temperature. It is this temperature range which primarily corresponds to the published studies which are between 17 °C and 27 °C. Removal performance varies widely from 82 % BOD removal for a pond in Kenya, operating at 17^{o}C (Pearson et al., 1996), to a BOD removal efficiency of 50 % from an AP operating at 28 ^{o}C in Brazil (De Oliveira, 1990) and at 24^{o}C in India (Dhariwal and Soni, 2008) (Table 2-2). Reported removal efficiencies show a poor correlation with this temperature relationship (Figure 2-3).

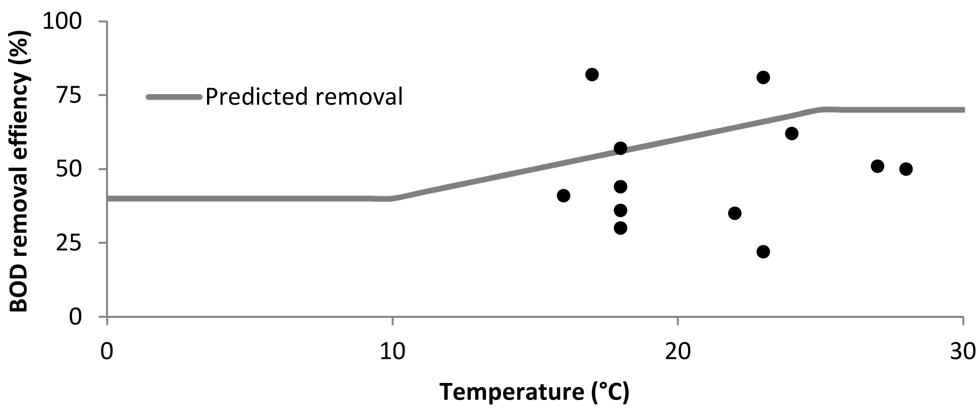

Figure 2-3: Reported BOD removal efficiency from literature with respect to temperature, including predicted removal rates from Mara and Pearson (1998)

Clearly this discontinuity suggests that, whilst temperature is important, design characteristics are potentially more important in achieving continuous performance. To illustrate, Papadopoulos et al. (2003) reported consistent BOD removal during long term operation of an AP in Greece, despite marked seasonal transitions in temperature, with a mean monthly range of 2.7 to 25.5 ^{o}C. The authors attributed the consistency in performance to sedimentation of organically bound particulate matter acting as the dominant removal mechanism. Similarly, Saqqar and Pescod (1995b) found no clear increase in BOD removal through an increase from 12 to 28 ^{o}C for an anaerobic pond in Jordan.

2.2.4 The impact of sludge layer accumulation on current anaerobic pond performance and desludge frequency

As a passive process, which is generally designed for plug flow conditions to dominate, process control is generally limited to managing the depth of the accumulated sludge layer. The sludge layer is significant to operation since this represents the most anaerobically active region in the pond. Based on this assumption, it could be postulated that a thick sludge layer presents a more anaerobically active region. However, many environmental factors influence the rate of growth or accumulation of the sludge layer, resulting in a heterogeneous particulate stratum comprised of inert solids, inactive cells and non-biodegraded organics in addition to the active biomass (Papadopoulos et al., 2003). Consequently, the depth of the sludge layer is difficult to predict but has several operational consequences. For example, based on an empirical design philosophy, the theoretical HRT in APs are based on 'empty bed contact times'. Significantly, as the sludge layer increases, the available pond volume will decrease, shortening the actual HRT whilst also increasing the organic loading rate above the design OLR (Peña et al., 2000; Nelson et al., 2004). As a consequence, lower apparent sludge accumulation rates have been observed. This can be explained as the result of higher internal velocities generating short-circuiting which permitted the wash out of solids into the effluent with the subsequent impact of a poorer quality effluent (Schneiter et al., 1993). Several authors have therefore proposed a 'critical' desludging volume of around 33 % (Mara and Pearson, 1998; Picot et al., 2005b). Vega et al.

(2003) supported this assessment, reporting that 50% sludge volume in a pond will lower treatment efficiency, whereas at 30 %, the presence of biomass for degradation can improve removal efficiency. Nelson et al. (2004) proposed that at the inception of design, APs could be sized to ensure degradation balances accumulation of solids, to minimise or potentially omit desludging. However, this was only deemed possible at operating temperatures greater than 19 ^{o}C (Schneiter et al., 1993).

Whilst the growth of the sludge layer to a 'critical' depth can present operational problems, equally it is the desludging stage which governs the operational process economics of APs (Carre et al., 1990). As such, the desludging interval in pond systems is typically fixed to minimise cost (Agunwamba, 1993). To aid in the design stage and also to enable a prediction of desludging frequency as an operational tool, a number of empirical models have been presented to predict sludge accumulation. Mara and Pearson (1998) proposed the following equation for desludging frequency:

$$n = \frac{V_a}{3Ps} \qquad \textbf{Equation 2-3}$$

where n is the desludge interval, in years; V_a is the volume of the anaerobic pond, in m^3; P is the population served; and, s is the sludge accumulation rate, in m^3 PE^{-1} y^{-1}. Whilst most studies quote a sludge accumulation rate, Saqqar and Pescod (1995a) noted this value will be site-specific. Thus whilst useful from an operational perspective, the authors proposed a more robust philosophy based upon on mass fluxes of suspended solids and BOD:

$$V_{AS} = K_{AS}\left[\frac{1.7\ F_{XVSS,0} + 4.5\ F_{XFSS,0} + \ F_{CBOD,0}}{\rho_w}\right] \qquad \textbf{Equation 2-4}$$

where V_{AS} is the volume of accumulated sludge, in m^3 d^{-1}; K_{AS} is called the accumulated sludge coefficient; $F_{XVSS,0}$ is flow rate of volatile suspended solids at the pond inlet, in kg d^{-1}; $F_{XFSS,0}$ is flow rate of fixed suspended solids at the pond inlet, in kg d^{-1}; $F_{CBOD,0}$ is flow rate of total BOD at the pond inlet, in kg d^{-1}; ρ_w is the density of water in kg m^{-3}. Whilst the prediction was tentatively more accurate, through the specific inclusion of influent characteristics, K_{AS} remains an empirically derived constant based on pond experience. For example, the same authors cited a K_{AS} of 0.6 for an AP in Jordan, whereas Paing et al. (2000) and Papadopoulos et al. (2003) reported K_{AS} of 1.4 for APs in France and Greece respectively. This deviation demonstrates significant variation based on environmental and process variation. Interestingly, the latter values also exceed unity which had been previously suggested as representative of the highest potential sludge accumulation rate (Saqqar and Pescod, 1995a). Based on site observation, Papadopoulos et al. (2003) subsequently incorporated temperature effect to more adequately describe K_{AS} for seasonal variation:

$$K_{AS} = 0.00898\, T^2 - \ 0.9442T + 12.967 \qquad \textbf{Equation 2-5}$$

where T is the ambient air temperature, in ^{o}C. The current perception is that sludge accumulates in cold periods when the biomass is inactive, and then decreases in warmer weather when degradation is higher (Picot et al., 2003; Papadopoulos et al., 2003). A 'critical' transition temperature for biomass activity of between 14 ^{o}C and 17 ^{o}C has been suggested. Papadopoulos et al. (2003) also postulated that the degradation of solids upon the return to temperature, will only occur in the active layer, whilst the inert layer underneath will exhibit a steady increase throughout the year, with fixed solids adding volume and compaction slightly reducing volume. To reinforce conservative design, Mara and Pearson (1998) suggested using an *s* of 0.1 m^3 PE^{-1} $year^{-1}$ which appears adequate based upon comparison of *s* values retrospectively computed from published studies (Table 2-3). Clearly accumulation modelling provides a convenient and simplified platform for understanding sludge accumulation. However, several non-linear effects will significantly impact process operation. For example, several authors postulate that sludge accumulation rate will slow as the pond establishes and matures (Picot et al., 2003; Picot et al., 2005b; Green et al., 1995). An analogous trend has been observed in septic tanks where sludge volume per capita increased for the first three years of operation, but then decreased in the fourth year (Philip et al., 1993). Interestingly, the accumulation models also neglect the topographic description of the accumulated sludge layer.

Table 2-3 Sludge accumulation rates, s, reported in the literature

Reference	*Pond type*	*Loading rate* ($gBOD\ m^{-3} d^{-1}$)	*HRT* (*d*)	*Temperature* (^{o}C)	*s* ($m^{3}\ PE^{-1}\ year^{-1}$)
Abis and Mara (2003)	Primary facultative[1]	79	2.5	9.5	0.13
Carre et al. (1990)	Primary[2]	N/A	N/A	N/A	0.12
Konate et al. (2010)	Anaerobic	165	3.0	26.5	0.04
Nelson et al. (2004)	Primary anaerobic	N/A	2.5	12.2	0.02
Nelson et al. (2004)	Primary facultative	N/A	24.0	21.1	0.04
Nelson et al. (2004)	Primary facultative	N/A	10.6	16.4	0.02
Philip et al. (1993)	Septic tanks[3]	N/A	N/A	N/A	0.07
Picot et al. (2005b)	Primary facultative[4]	113	N/A	N/A	0.08

[1]Mean of 3 ponds reported, [2]Mean of 12 ponds reported, [3]Mean of 33 tanks reported, [4]Mean of 19 ponds reported

For example, numerous authors have reported that the sludge layer is highest at the inlet (Abis and Mara, 2005; Carre et al., 1990; Nelson et al., 2004; Nelson and Jiménez, 2000; Paing et al., 2000; Picot et al., 2005b; Schneiter et al., 1993). In contrast, Saqqar and Pescod (1995a) determined that sludge accumulation was highest in the centre of the pond. The authors attributed the effect to the inlet geometry imposing high velocity which induced jetting. As sludge accumulation generally is more critical at the inlet, it is this area which will decide the desludging frequency of the pond (Abis and Mara, 2005). Importantly, since this volume is only a fraction of the total volume, desludging will occur more frequently than predicted with a total mass balance. Desludging frequency for APs has been reported between two and four years (Alexiou and Mara, 2003; Papadopoulos et al., 2003; Konate et al., 2010), and is usually performed when sludge volume reaches 33 or 50 % of total pond volume (Agunwamba, 1993; Mara and Pearson, 1998; Konate et al., 2010). Due to the extended sludge age, sludge tends to be well stabilised (Konate et al., 2010), with a total dry solids content of 11 % reported in France (Picot et al., 2005b) and 6 % in Burkina Faso (Konate et al., 2010). Whilst metals concentrations have not found to be a hazard in ponds treating domestic wastewater, accumulation of helminth eggs means many AP sludges require a level of treatment before application to land (Mara and Mills, 1994; Konate et al., 2010).

2.3 Comparison of biogas production from anaerobic ponds treating various source waters

Retention of biogas is only possible with a gas collection system which is now either installed retrospectively on existing ponds or integrated into new build designs. The main components of biogas from APs are carbon dioxide, CO_2, and methane, CH_4, both of which are greenhouse gases (GHGs) thus coverage and utilisation of the gas is necessary for limiting process carbon footprint (Noyola et al., 2006); utilisation has the potential to shift the carbon balance to carbon positive through production of green electricity to grid (McAdam et al., 2012). The first covered APs were introduced for agricultural sludges, such as swine, poultry and dairy manure (Safley Jr. and Westerman, 1988). Due to their extended solids retention times, APs have shown comparable performance to mesophilic digesters, as the longer solids retention times compensate for the lower temperatures (Heubeck and Craggs, 2010). Anaerobic process energy balances have demonstrated that full-flow wastewater treatment cannot be heated due to the high fluid flow rates, the high specific heat capacity of water and the comparatively low organics concentration of the feedwater (Martin Garcia et al., 2013). Consequently, due to the lower growth rates of psychrophilic methanogens (Lettinga et al., 2001) ponds require at least double the retention time of mesophilic digesters (Craggs et al., 2008). The first reported covered ponds treating domestic wastewater were evolved during the development of mixing pits in advanced facultative ponds in California (Oswald et al., 1994; Green et al., 1995) and the covering of the APs at the Melbourne Water Western Treatment Plant in Australia (Hodgson and Paspaliaris, 1996).

Methane production varies spatially within a pond, with greatest production at the inlet (Safley Jr. and Westerman, 1988; 1989). Paing et al. (2000) observed that although the methanogenic potential of the sludge in an AP was greater towards the outlet, this did not correlate to greater biogas production due to a smaller volume of accumulated sludge in this area. Many authors have noted a strong correlation between biogas production and temperature (Craggs et al., 2008; Picot et al., 2003; Safley Jr. and Westerman, 1989; Safley Jr. and Westerman, 1992; Toprak, 1995). It is postulated that this is associated with sludge accumulation at colder temperatures, followed by sludge degradation and higher biogas production in the warmer months (Picot et al., 2003; Safley Jr. and Westerman, 1989). Interestingly, although the volume of biogas produced increases with temperature, biogas production has been observed at temperatures as low as 3 ^{o}C (McGrath and Mason, 2004). It is hypothesised that insulation provided for by the water column and surrounding soil, enables temperature buffering and can therefore facilitate year-round biogas production (Craggs et al., 2008; Park and Craggs, 2007; Picot et al., 2003; Safley Jr. and Westerman, 1989). It has been observed that an AP's capacity for methanogenesis at low temperature increases with pond age, as the biomass acclimatises (Heubeck and Craggs, 2010). Temperature affects biogas composition as well as volume. Biogas methane content increases with decreasing temperature, which has been attributed to increased preferential CO_2 absorption into the liquid at low temperatures (Craggs et al., 2008; Safley Jr. and Westerman, 1992). However, Noyola et al. (2006) observed that an equivalent increase in CH_4 solubility would occur at lower temperatures (Figure 2-4). To demonstrate, Cookney et al. (2012) noted that the effluent from a high rate UASB treating unheated wastewater (average 16 °C) was supersaturated with methane, resulting in a loss of approximately 45 % of the produced methane as a dissolved emission.

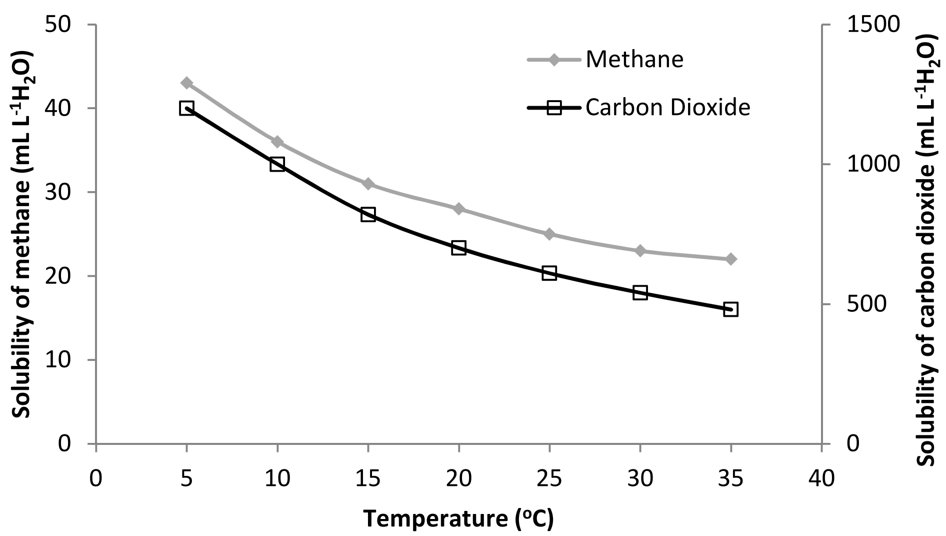

Figure 2-4 Change in solubility of methane and carbon dioxide with temperature

Whilst this is a consideration for both carbon and energy balances, biogas collection from APs has proved economically attractive, with the largest reported AP, in Melbourne, Australia, estimated to generate an annual revenue of $1.8mAUS from biogas produced from domestic wastewater (DeGarie et al., 2000).

2.4 Hydraulic design of anaerobic ponds

For open AP structures, environmental factors can impact on pond hydrodynamic performance. For example, thermal stratification from solar warming and differences in ambient temperature can enhance short-circuiting (Agunwamba, 2006; Kehl et al., 2009; Moreno, 1990), rainfall and evaporation add or decrease to the pond volume altering flow and velocity profiles (Abbas et al., 2006) and wind speed can influence flow as well as promoting oxygen mass transfer at the pond surface through forced convection which has the potential to inhibit anaerobic processes (Peña et al., 2000; Vorkas and Lloyd, 2000). However, now pond covers are a key principle in AP design for biogas recovery, environmental impacts are reduced. Several authors have subsequently also cited advantages to covering, since the covering structures are likely to regulate the sludge temperature, with the dark materials also enabling solar heat absorption (Safley Jr. and Westerman, 1992; Heubeck and Craggs, 2010).

The principal limitation of utilising ponds is their land requirement. Consequently, the area required for the pond can be minimised through shortening HRT. However, as APs possess significant internal fluid volumes and are designed for plug flow, hydraulic failure in the basic design is common and if avoided, can reduce the overall volume requirement. To demonstrate, in practice, the recorded actual (HRT_a) has been reported to vary by between 23 % and 116 % of the theoretical HRT (HRT_t) (Muttamara and Puetpaiboon, 1997; Vorkas and Lloyd, 2000). This difference between HRT_a and HRT_t is driven by short-circuiting, or inversely, dead zones, introduced into the ponds hydraulic regime. In the most basic design, the AP is an unbaffled tank. For this design, the aspect ratio, or length-width ratio (L:W) has been investigated as the primary design parameter. Generally, higher L:W ratios tend toward plug flow conditions, which has been reported to improve hydraulic performance (Abbas et al., 2006;

Abbassi et al., 2009; Persson, 2000; Shilton and Mara, 2005). Interestingly, this falls within the typical design range proposed for primary sedimentation tanks of 3:1 to 5:1; the principle for these dimensions is analogous to AP design principles, to maximise sedimentation rate by minimising short-circuiting. To reduce land requirement, Agunwamba (2001) investigated the effect of tapering the pond. Whilst land requirement decreased, the authors cited a concomitant decrease in removal efficiency, concluding that the performance losses due to the tapered design outweighed potential advantages in land reduction. To prevent channelling, orientation of the inlet and outlet have also been investigated (Abbassi et al., 2009; Moreno, 1990; Peña et al., 2000; Persson, 2000). Intuitively, opposite corners for entrance and exit have been demonstrated as most hydraulically efficient (Peña et al., 2000; Moreno, 1990), the effect of which is more pronounced for small L:W ratios (Agunwamba, 2006). Further guidance proposed by Mara and Pearson (1998) suggest that the outlet should be located under the water surface and/or fitted with a scum guard to prevent floating scum from escaping in the effluent (Mara and Pearson, 1998).

The inclusion of baffles into AP design enables the opportunity to develop more clearly defined hydraulic regimes. To illustrate, horizontal baffles can improve hydraulic performance by enhancing the path length of the AP, enforcing plug flow whilst minimising short circuiting (Muttamara and Puetpaiboon, 1997; Moreno, 1990; Shilton, 2000; Vega et al., 2003). The horizontal baffles force the flow around the sides of the baffles, thereby ensuring the flow cannot 'short circuit', and flow directly from the inlet to the outlet, but has to manoeuvre through a great proportion of the pond volume in order to reach the outlet. Furthermore, the small aperture created by the baffle openings mean the flow has to move through a smaller area when passing the baffle, increasing the velocity at these turning points of the baffles. This increased velocity both increases turbulence of the flow, thereby agitating more of the biomass, as well as creating a backpressure whereby some of the flow is recirculated backwards, creating turbulence against the overall flow direction as well as moving to areas of the pond that would not be utilised by a preferential flow pattern without turbulence. It is posited that this turbulence, created more agitation and thereby greater mixing, is advantageous to enhance contact between biomass and substrate. In a CFD study, Abbas et al. (2006) determined that four baffles presented an optimum configuration with respect to minimising short-circuiting (Abbas et al., 2006), whereas in a previous study, Vega et al. (2003) suggest two baffles. Other research groups have also presented evidence for small 'stub' baffles (Shilton and Harrison, 2003; Persson, 2000; Moreno, 1990), with similar numbers of baffles used but smaller structures, which presented similar results at a fraction of the construction costs (Moreno, 1990). Peña et al. (2003) studied vertical baffling (forcing the flow up and under baffles) as well as the more traditional horizontal baffling (forcing the flow around the edges of baffles). The studies showed that whilst better solid removal was observed with the horizontal baffles, the greater mixing effect of the vertical baffles produced higher COD removal. The most stable configuration studied was a mixing pit, created by introducing the flow into a deeper section than the rest of the pond, a design pioneered in California for advanced facultative ponds (Oswald, 1991; Green et al., 1995). Interestingly, Muttamara and Puetpaiboon (1997) reported that as an aside, through increasing the submerged surface area in the pond with baffles, biofilm growth can be stabilised, potentially enhancing biological activity.

Incorporating vertical baffles into APs broadly yields a reactor design analogous to the anaerobic baffled reactor (ABR) pioneered at Stanford University in the early 1980's (Barber and Stuckey, 1999). The ABR utilises vertical baffles to induce over-under flow through the reactor, to enhance mixing. In a vertically baffled system, the baffles stretch the width of the reactor, but leave openings alternately at the top and bottom of the reactor. Many of the advantages in using these baffles have the same fundamental principles as horizontal baffles: the baffles force the flow to take an indirect path from inlet to outlet, thereby reducing short circuiting and using more reactor volume; the small aperture of the baffles cause increased velocity past the baffle and also backpressure and recirculation, thereby increasing turbulence and mixing effects. However, a key distinction with vertical baffling is the liquid flow, when it is forced under the baffle, is driven through the settled solids layer on the base of the reactor, creating a direct contact between settled biomass and liquid substrate. This contact also increase agitation, as solids will be carried into the liquid and suspended by the flow passing through. At this stage, and upflow section is created as the flow is forced from under the previous baffle to over the following one, allowing suspended biomass to settle again, and preventing biomass being washed out over the next baffle. In order to ensure this happens, the flow rate in the reactor must be maintained low enough that the upflow velocity in these sections is not great than the settling velocity of the biomass. As long as this is maintained, a vertically baffled system offers the advantages of horizontal baffles of enhanced biomass-substrate contact, however, the forcing of the flow through the viscous sludge increases flow resistance, and combined with the repeated upflow sections in the reactor, means higher headloss is experienced in vertically baffled systems. These head losses may lead to pumping requirements that might affect the low energy requirements expected from pond systems. Therefore, flow rates or baffle numbers must be kept low in order to ensure head losses do not compound. The key advantages of low flow vertically baffled systems, such as baffled ponds and ABRs, over other high-rate anaerobic systems such as UASBs and expanded granulated sludge blankets (EGSBs) are a simple design, low capital and maintenance costs, long biomass retention times and high resilience to both hydraulic and organic shock loads (Barber and Stuckey, 1999; Dama et al., 2002). The distinction between the ABR and vertically baffled APs are a higher number of baffles which intensifies the process, to create a flow regime similar to several UASBs in series, albeit at the expense of a higher headloss. Consequently, less hydraulic dead-space has been reported at <8 to 18 % versus 80 % for a completely stirred tank reactor (Barber and Stuckey, 1999). When operated on a number of wastewater matrices, tCOD removal efficiency of 60 to 98 % has been reported (Langenhoff et al., 2000; Langenhoff and Stuckey, 2000), though the higher removal rates were recorded with heated (mesophilic temperature) feedwaters. However, lower temperature studies have demonstrated potential with 70 % COD removal at 20 °C and 60 % reported at 10 °C (Langenhoff and Stuckey, 2000). Through compartmentalising design, ABRs partially separate the phases of anaerobic digestion along the length of the reactor, with hydrolysis and acidogenesis occurring at the front end, closer to the inlet, and methanogenesis more predominant in later compartments (Barber and Stuckey, 1999; Dama et al., 2002; Langenhoff et al., 2000), enabling more favourable biological conditions (Barber and Stuckey, 1998). Whilst favourable results have been obtained, most studies are based on synthetic feedwaters comprised of soluble substrate (Barber and Stuckey, 1999). However, Dama et al. (2002) reported >70 % COD removal from a pilot-scale reactor (3.2 m^3) treating domestic/industrial wastewater, which suggests scale-up is feasible. Despite this, the ABR has struggled to establish as a full scale technology, largely due to difficulties with handling crude wastewater with high solids content, which can lead to solids washout and clogging under the baffles (Tilley et al, 2014).

2.5 Discussion

Recommendations for OLRs in APs have been designed conservatively, and analysis shows that in practice APs are loaded under these values. This appears counter-intuitive in light of high removal performances reported from higher loaded ponds. It is suggested that under-loading occurs to prevent odour nuisance, a commonly sighted problem with APs (Mara and Pearson, 1998). This aspect was a significant factor in development of AP designs, although as covers become standard, this issue will be largely negated (Park and Craggs, 2007). With positive experiences of highly loaded ponds (Pearson et al., 1996) and high-rate ponds (Peña, 2003; Peña, 2010), the relationship between loading rate and pond performance requires further investigation. Ponds have been successfully operated at OLRs of 1.82 kgBOD m^{-3} d^{-1} (Alabaster et al., 1991) at temperatures of 24 ^{o}C, although the limits have not been tested for lower temperatures despite the potential to approach these loadings. Removal efficiencies in APs are currently projected on the basis of design calculations that link removal efficiency to temperature (Mara and Pearson, 1998), although it is posited the role of temperature-dependent biological activity on removal efficiency has previously been over-estimated. From the data analysed in this review, the lack of correlation between temperature and removal efficiencies, coupled with the strong correlation between low HRT, high loading, and high removal efficiencies suggest that sedimentation is almost entirely responsible for the removal efficiency an AP.

Pond sizing should remain dependent on OLR, but whereas this is currently calculated from average air temperature in the coldest month, this approach should be reconsidered. Recommended loading rates should be defined by operational parameters such as reduction of removal efficiency or excessive sludge accumulation rather than by odour, so that APs are sized to meet operator requirements for removal efficiency or effluent quality, and desludging interval. When considering desludging interval, the spatial distribution of sludge in an AP should be considered. It has been noted that most sludge accumulates near the inlet (Abis and Mara, 2005; Carre et al., 1990; Nelson et al., 2004; Nelson and Jiménez, 2000; Paing et al., 2000; Picot et al., 2005b; Schneiter et al., 1993), and therefore the impact of sludge accumulation in this region on pond performance should be investigated. Studies have shown that high sludge volumes can have a significant impact of pond hydrodynamics (Peña et al., 2000), and that sludge volume can vary seasonally (Papadopoulos et al., 2003), although this is not currently considered at the design stage. During colder periods, biogas production will be low, and therefore the effects of mixing due to gas bubbling reduced, whilst sludge volume at its annual peak, reducing actual HRT within the pond. At the highest temperature these effects will be reversed, with low sludge accumulation leading to improved hydrodynamics, but with high biogas production creating greater mixing.

Traditionally, APs have been designed for plug-flow conditions, to maximise sedimentation in order to remove solids for subsequent ponds (Alexiou and Mara, 2003). Whilst this trend has demonstrated improvements over earlier pond designs (Abbas et al., 2006; Muttamara and Puetpaiboon, 1997; Persson and Wittgren, 2003), the ability of APs to maximise biological activity through biomass contact will always be restricted whilst the primary focus is on settlement. The literature demonstrates APs have been capable of high removal of solids at short HRTs and high loading rates, which suggest the optimal operational parameters of the AP are yet to be found, and that greater biomass/substrate contact may improve organic degradation without significant additional biomass washout (Peña et al., 2003). Work on mixing pit (Green et al., 1995) and vertical baffle designs support this hypothesis (Peña and Mara, 2003), as do findings from ABR studies (Barber and Stuckey, 1999). The use of covers must be recommended to prevent GHG emissions (Noyola et al., 2006), control odour problems (Shelef and Azov, 2000), and where feasible generate electricity (Hodgson and Paspaliaris, 1996).

Whilst design improvements must continue to be sought, the traditional advantages of APs should also be remembered. Higher loading rates, and the use of baffles, may produce improved mixing profiles and biogas production, but they will also increase sludge accumulation within the pond and may concentrate the sludge towards the inlet. Desludging frequency is the principle operational cost of APs and most demanding maintenance aspect (Carre et al., 1990; Picot et al., 2005b), and the extended sludge retention times, leading to reduced sludge handling activities, are an attractive aspect for considering ponds as a wastewater treatment option, particularly for decentralised works (McAdam et al., 2012). Therefore, improved guidelines for the design of APs must allow for flexibility in specific designs, depending on the situational requirements of reduced physical footprint, treatment efficiency and effluent quality, renewable energy potential, and sludge management benefits.

2.6 Conclusions

The evolution of AP design since its inception in the 1950s has been studied. Anaerobic ponds have been implemented worldwide, however, their design remains firmly rooted as a roughing stage in larger WSP systems, where retrospective adaptation for energy generation has generated a non-idealised design for ponds.

Whilst APs are now designed and built for energy recovery, the most commonly adopted design guidelines are those proposed by Mara and Pearson (1987). Whilst these presented a step change in design at their implementation, the design envelope has now changed:

- Current design specifies maximum OLR to avoid diffuse odour. With the introduction of covering and more advanced understanding of AP hydraulics, higher OLR can be achieved, providing smaller ponds of equivalent or higher performance than originally designed

- Current design guidance is based on OLR and HRT which are both transient in APs due to the developmental growth of the sludge layer
- Furthermore sludge accumulation models proposed for design or operational support do not assume topographic detail
- Research has now demonstrated the advantage of baffled structures, and other hydraulic structures (e.g. mixing pits, inlet/ outlet orientation) to advance the efficiency of the hydraulic regime. However, their implementation to date has not been wide ranging
- New guidelines should be developed, in order to reflect recent research into increased loading rates and the use baffling, whilst also considering traditional benefits of reduced sludge handling requirements

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3 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 PE^{-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 AP 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^{-1} 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

3.1 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 digestion (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, 2013). Consequently, sludge produced at small works is tankered to centralised WWTW 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 tankered offsite to centralised AD facilities, but also increases the risk of local greenhouse gas (GHG) 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.

APs are typically dimensionalised similarly to rectangular primary sedimentation tanks (PSTs) in a European WWTW (3:1 Length:Width aspect ratio) to enable effective solids capture (Guyer, 2013). However, APs are also specifically oversized to allow extended sludge residence times (therefore combining both primary sedimentation tank and sludge holding plus treatment 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., 2013) 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) quantifying 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 WWTWs to benchmark comparative environmental performance.

3.2 Materials and Methods

3.2.1 Experimental reactor design

A pilot-scale horizontally baffled AP was constructed of 12 mm 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). 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' (Figure 3-1). 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 contained 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, internal liquid sampling ports were installed at 0.25, 0.75 and 1.25 m along the reactor length to aid diagnosis of the fate of sanitary parameters.

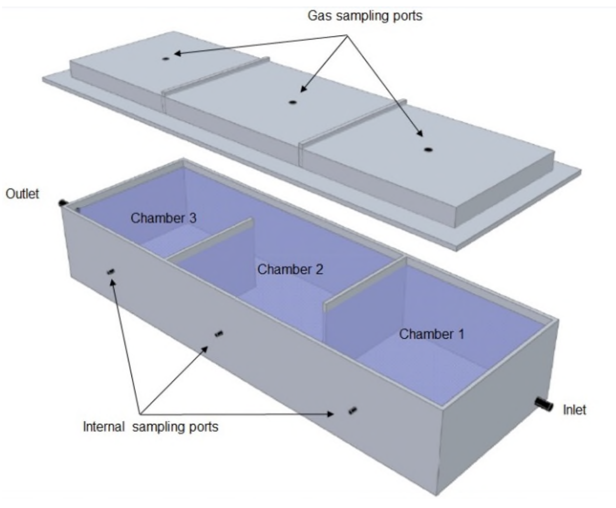

Figure 3-1 Layout of the pilot scale horizontally baffled anaerobic pond (HBAP), detailing the locations of the inlet, outlet and internal sampling ports used for analysis.

The reactor was initially seeded with 7 % by volume anaerobic sludge (volatile solids, VS = 36 g L^{-1}) collected from a mesophilic AD. The AP was located in Cranfield's piloting facility at Cranfield University sewage treatment works to enable operation on real municipal wastewater. The AP was fed crude wastewater at a liquid flow rate of 75 L d^{-1}, yielding a theoretical hydraulic retention time (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 (Peña, 2002; El-Deeb Ghazy et al., 2008; De Oliveira, 1990). The piloting facility was unheated. Consequently the AP was subjected to an ambient air temperature range of -4.1 to 22.7 ^{o}C over the duration of the study. 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 averages 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.

3.2.2 Determination of sludge degradation from three full-scale STWs

Sludge samples were taken from three decentralised WWTW in the UK, which contained a 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 = 210 m^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 ±2.0 ^{o}C) for 8 weeks. Sludge samples were setup in triplicate.

3.2.3 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 1.2 µm glass fibre filter (Whatman, Maidstone, UK) with the particulate COD fraction (pCOD) calculated by subtracting sCOD from tCOD. In order to measure the decline in energy recovery potential from the sludge stored in the holding tanks, the calorific value (CV) was determined. Calorific value is defined as the amount of produced by the complete combustion of a material, and change in CV of the sludge with time was used to measure the decline in stored energy within the sludge. The CV was measured using bomb calorimetry according to CEN/TS 15400 (British Standards Institution, 2006) by Marchwood Scientific Services, Southampton, UK. A range of six volatile fatty acids (VFAs), acetic, propionic, butyric, n-butyric, i-valeric and n-valeric, were determined by high performance liquid chromatography (HPLC) 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 of operation using a perspex tube graduated at 1 mm intervals. To enhance spatial resolution, a grid of 0.1 m x 0.1 m was used. Ambient and liquid temperatures were recorded at the time of sampling using a digital probe thermometer, with a sensitivity of ±0.05 ^{o}C.

3.3 Results

3.3.1 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, 8 and 40 kg m^{-3} were measured in sludge samples from the WWTW comprising the TF, RBC and OD respectively. An initial increase in sCOD was noted at the start of the trial which was indicative of hydrolysis (Figure 3-2). However, following 6, 4 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 2.1×10^{-6}, 2.0×10^{-6} and 4.8×10^{-5} $kgCH_4$ d^{-1} were recorded for the TF, RBC and OD respectively. As a consequence, following eight weeks storage, CV reduced from 13,781, 13,361 and 13,767 kJ kg^{-1} for the TF, RBC and OD WWTW respectively to 12,432, 12,056 and 11,990 kJ kg^{-1}, equivalent to a reduction in mean CV of between 9.8 % and 12.9 %.

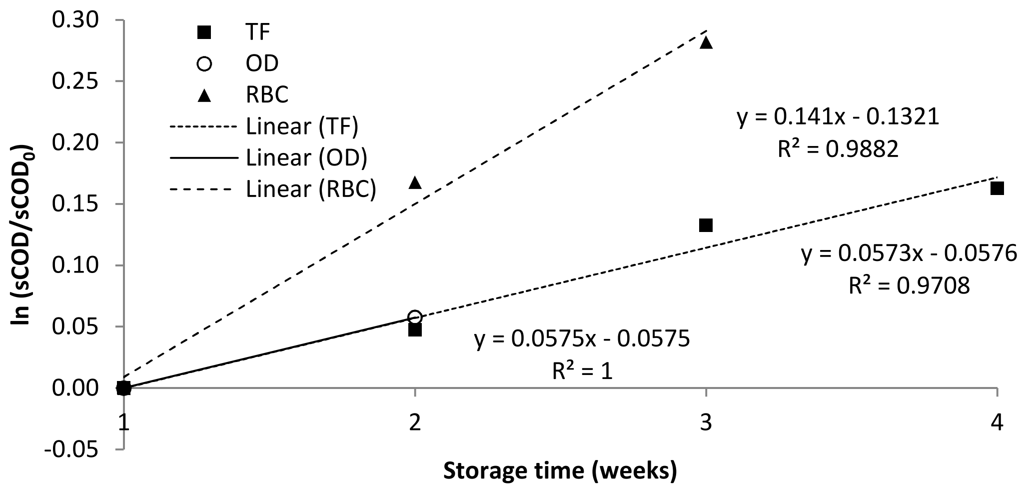

Figure 3-2 Soluble COD production from three different sources of on-site sludge during the initial stage of sludge storage, a trickling filter (TF), oxidation ditch (OD) and rotating biological contactor (RBC)

3.3.2 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. Fractionated COD data was also collated into monthly averages to discern the effect of temperature on removal (Figure 3-3).

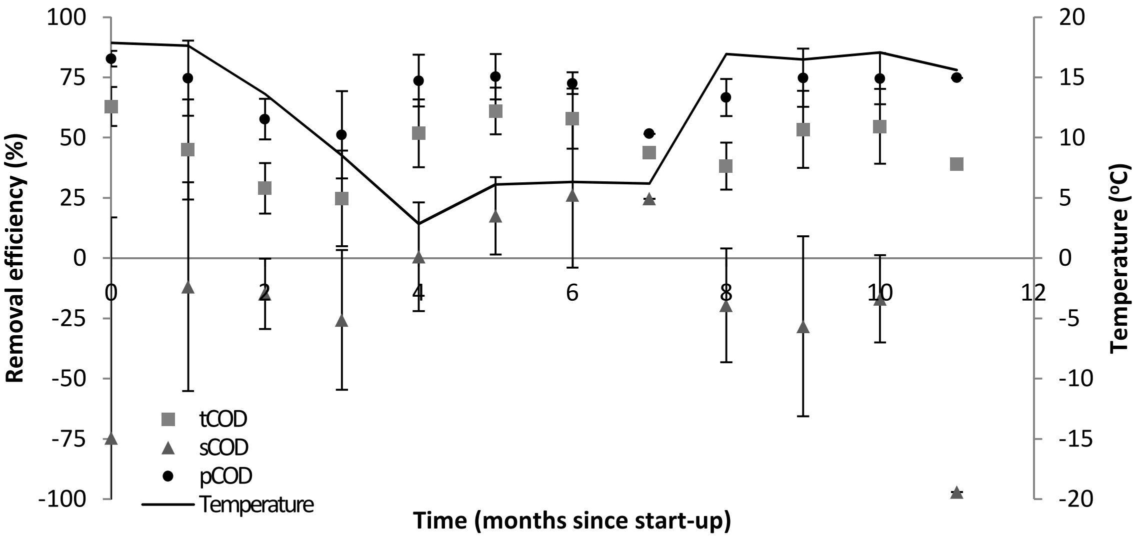

Figure 3-3 Mean monthly effluent temperature and removal efficiency of COD fractions in a pilot-scale AP over a 12 month period

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 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 ^{o}C; Jun.-Aug., 75 ±10 %, $T_{effluent}$ = 16.7 ^{o}C). However, the impact of temperature on sCOD removal was more evident. To illustrate, during the summer period, negative sCOD removal of -26 ±33 % was recorded (Jun.-Aug., $T_{effluent}$ = 16.7 ^{o}C), whereas during winter, positive sCOD removal of 11 ±25 % was determined (Dec.-Feb., $T_{effluent}$ = 4.6 ^{o}C). The increase in sCOD with temperature is indicative of VFA formation (McAdam et al., 2012), which was supported by a weakly positive correlation between effluent VFA concentration and effluent temperature (Figure 3-4). More specifically, at effluent temperatures above 12 °C, VFA concentration markedly increased as a proportion of sCOD, whereas at effluent temperatures less than 15 ^{o}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 VFA concentration.

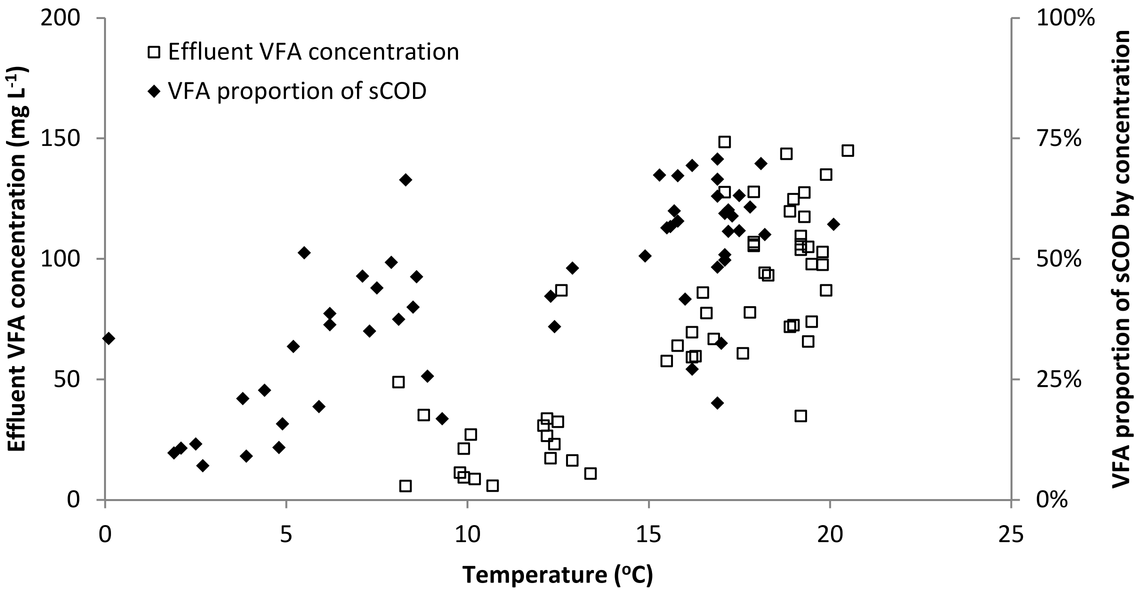

Figure 3-4 Effluent volatile fatty acid (VFA) concentrations and VFA proportion of effluent sCOD from the pilot scale AP over 12 months study (n = 56)

3.3.3 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 3-5). The influent TSS profile generated from the annual data indicated an unstable TSS concentration profile within the influent (TSS range 91 to 1573 mg L^{-1}), as demonstrated by the positive skew above the 90th percentile. Median particle size in the influent ranged from 35 to 235 µm. 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, 77 and 80 mg L^{-1} were recorded at the 50th, 75th and 90th percentiles, confirming the characteristic narrow distribution. A d_{50} median particle size of 20 µm was measured in the effluent.

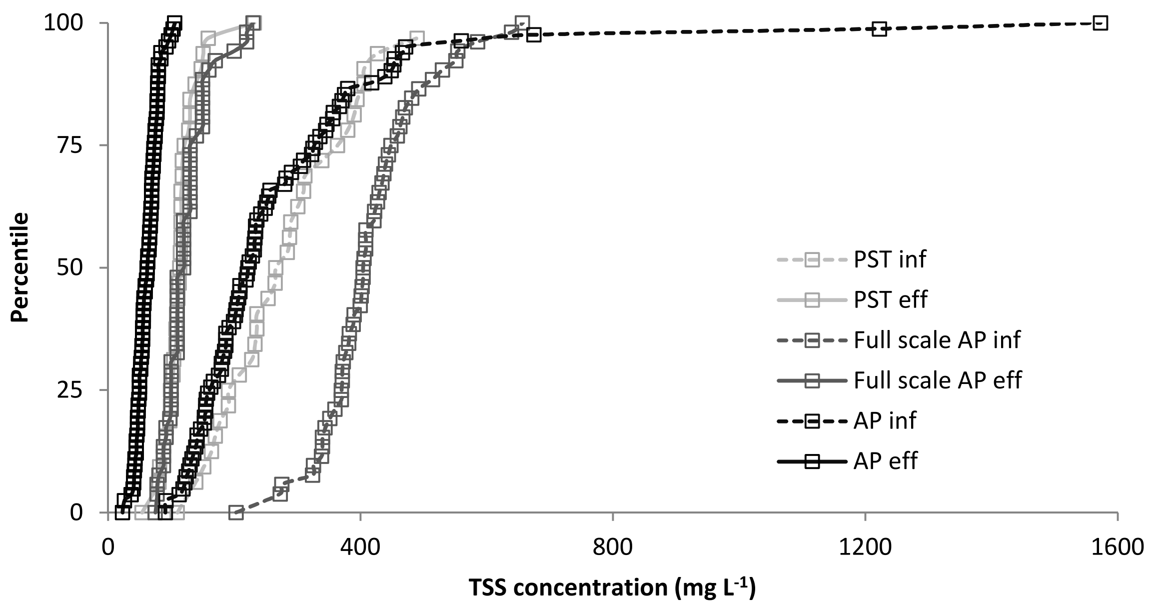

Figure 3-5 Resilience curves for TSS influent and effluent concentrations for the pilot-scale AP from this study (n=82), and a full scale AP (n =52) and a primary sedimentation tank PST (n =40)

The effluent profile was compared to the effluent TSS profile generated from a full-scale UK 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 3-6). 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 PE^{-1} y^{-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.

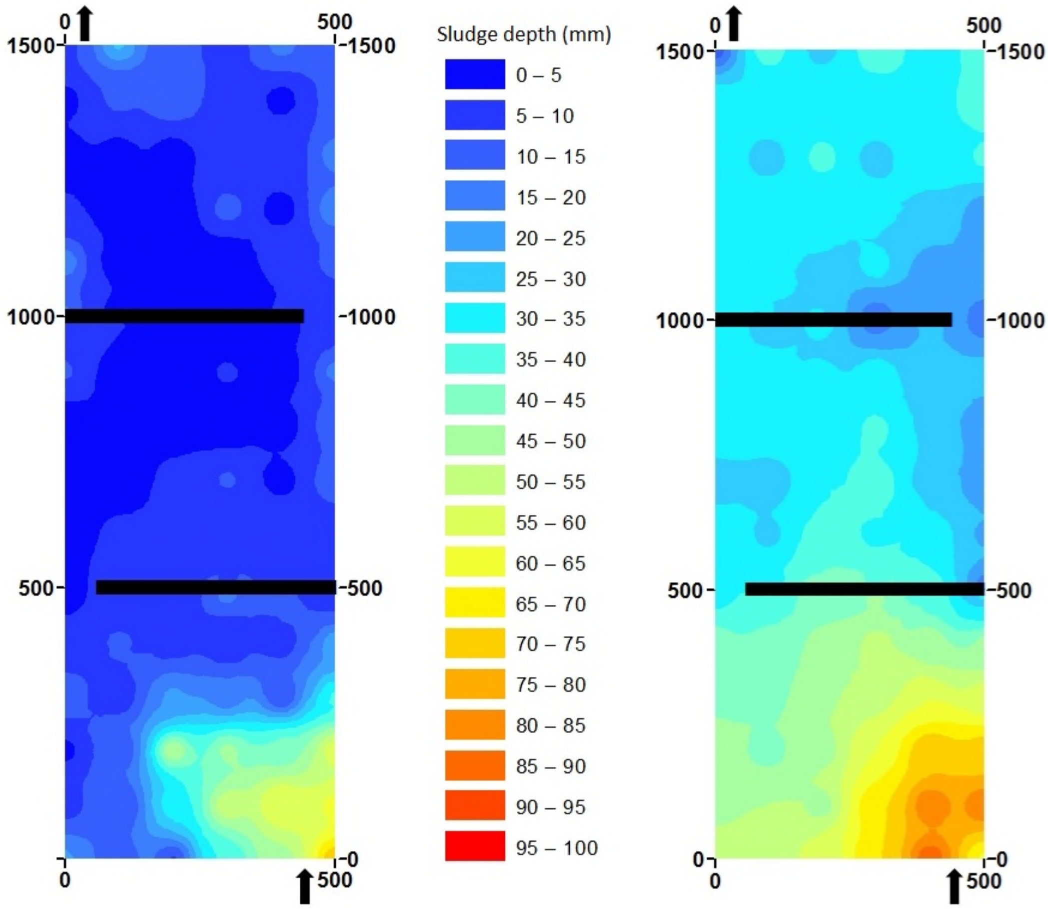

Figure 3-6 Sludge accumulation map at (left) 219 days operation and (right) 534 days operation, produced from 96 sludge depth measurements on a 100 mm x 100 mm grid

3.3.4 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 3-7). A mean annual production rate of 3.69 LCH_4 m^{-3} wastewater treated (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, mean biogas production was significantly higher in the second winter at 2.53 LCH_4 m^{-3} WWT, compared to the initial winter period (Q2, 0.22 LCH_4 m^{-3}WWT), indicating acclimation to have occurred over the study.

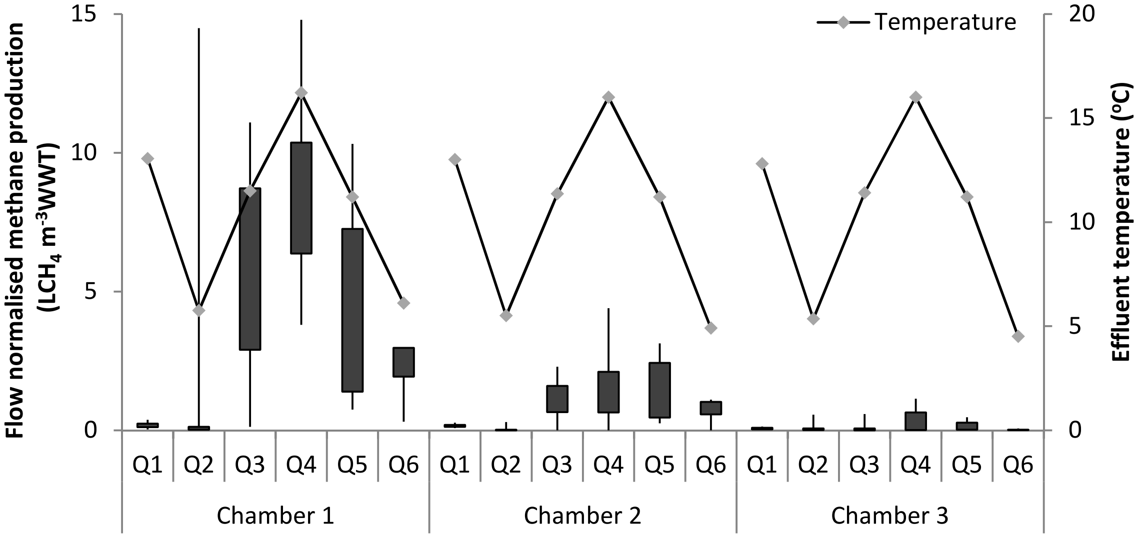

Figure 3-7 Average methane production from biogas, separated by reactor chamber, over the total study period (quarters 1 to 6, n =54) with mean effluent temperature

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 3-8). 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.

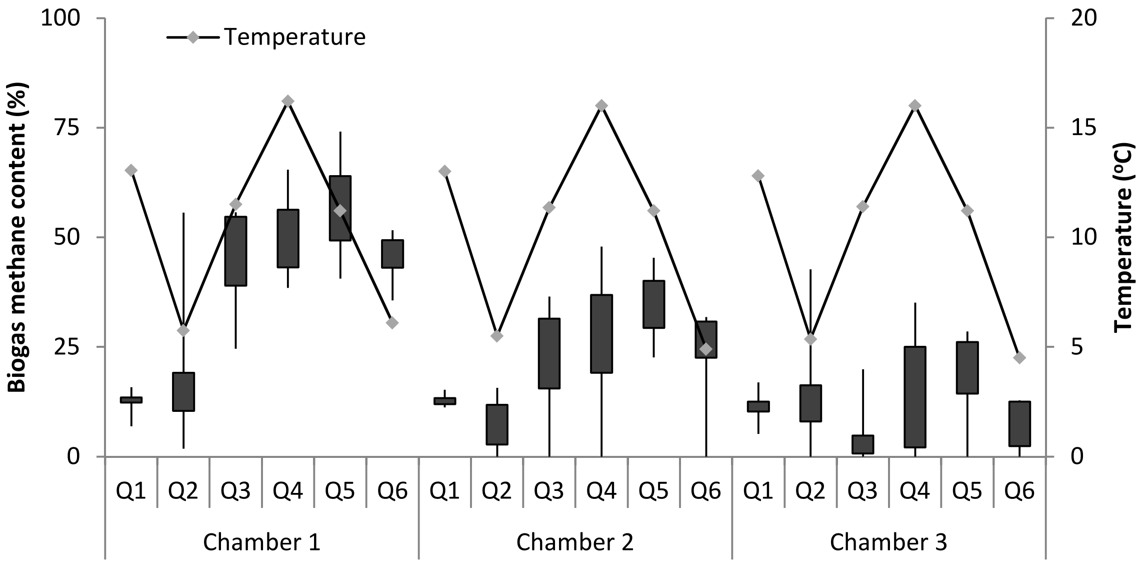

Figure 3-8 Average biogas methane content, separated by reactor chamber, over the total study period (quarters 1 to 6, n =54) with mean effluent temperature

Total methane gas production ranged between 0.02 LCH_4 m^{-3}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 3-9) 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 ^{o}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 ^{o}C to below 0.5 ^{o}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 APs, which resulted in an effluent temperature profile more closely described by ambient air temperature than the influent wastewater ($T_{ambientair}$ -4.1 to 22.7 ^{o}C). For the full data set above 8.8 ^{o}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.

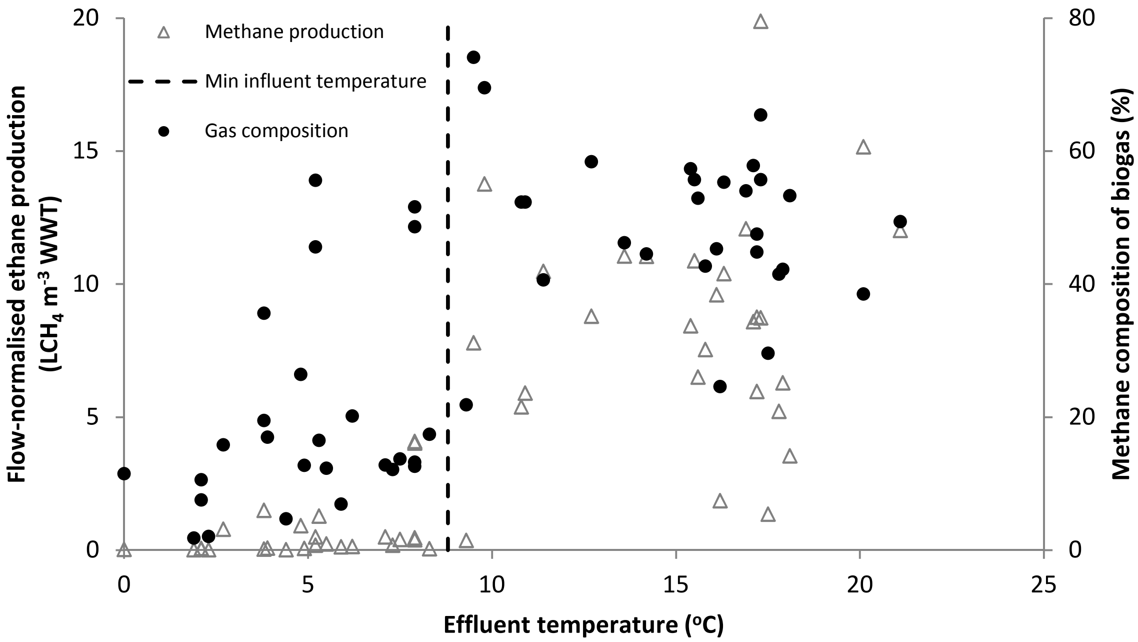

Figure 3-9 Flow-normalised methane production and composition of biogas against temperature (n = 54)

3.4 Discussion

Data collected from this research demonstrate 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, the methane production rates measured from three sludge holding tanks sludge samples illustrate that between 1.15 and 26.8 $kgCH_4$ $tonne^{-1}$RDS would be released over a typical 90 day retention time using conventional open sludge stoarge, or 0.05 to 1.2 gCH_4 m^{-2} d^{-1}. Whilst lower than those recorded in the literature, 36 $kgCH_4$ $tonne^{-1}$RDS and 7 gCH_4 m^{-2} d^{-1} (Hobson, 2001; Czepiel et al., 1993), the data provides a conservative estimate of UK sludge holding tank methane emissions. Importantly, these data suggest that by replacing sludge holding tanks with APs, this release which is equivalent to approximately 57 $mgCH_4$ PE^{-1} d^{-1} could be omitted through methane retention within the covered pond.

Following continued AP operation, it follows that effluent quality will decline due to washout if desludging has not been undertaken (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 PST despite having operated the AP without desludging. Spatial distribution of the resident sludge volume at 219 d illustrated that 67 % of retained sludge was in the first chamber (Figure 3-6) and is consistent with reports on full scale APs (Paing et al., 2000; Picot et al., 2005b). 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 and Harrison, 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, evidenced by the consistent effluent profile (TSS 23-106 mg L^{-1}, d_{50} 4 -19 µm), and so presents a suitable replacement for existing PSTs. The AP in this study was dimensioned to reflect full scale standard design practice (3:1 L:W) and enable scale-up comparisons, an approach that has been adopted previously (Dama et al., 2002; Muttamara and Puetpaiboon, 1997). 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 PE^{-1} y^{-1} was recorded based on data at the completion of the trial, which is in the range of earlier APs operated at higher temperatures (Picot et al., 2005b; Nelson et al., 2004). 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, with microbial communities adapting to both the psychrophilic temperatures and the available substrate (Weiland and Rozzi, 1991). The VFA formation observed in this study has also previously been considered an indication of acclimation, where VFA have been observed in effluent for up to a year following start-up (Picot et al., 2003). At low temperature, homoacetogens have faster growth rates than methanogens (Kotsyurbenko et al., 2001), which can explain the period of VFA build-up in an AP prior to methanogenic establishment. However, VFA formation was noted at the end of the study period (>500 d), 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 (VFA) formed in the first chamber and the sludge layer resident in the subsequent two chambers. Further optimisation of AP design could be considered to enhance VFA utilisation and improve methane yield. For example, driving contact between soluble substrate and the active sludge layer in the latter pond section using engineering interventions such as vertical baffling to enhance methane production.

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 resemble the influent wastewater, which in this study was consistently above 8.8 °C (Park and Craggs, 2007; Safley Jr. and Westerman, 1989). Consequently, the mean yield recorded above 8.8 °C of 8.48 LCH_4 m^{-3}WWT 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. To illustrate, after ten years of operation, an AP in Melbourne, Australia, delivered a yield of 0.16 m^3CH_4 m^{-3}WWT, around eight times higher than this study. Whilst an equivalent yield cannot be expected due to the temperature differential (Melbourne sewage average temperature, 19.6 ^{o}C, northern hemisphere, 12 ^{o}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. Since biogas methane content remained >35% follow start-up (even during winter), there is potential for small scale electrical production through combined heat and power (CHP).

3.5 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.

- Estimated methane emission rates from sludge holding tanks in temperature conditions present compelling evidence for the need to capture fugitive emissions. However, utilisation of fugitive methane from sludge holding tanks is unlikely to be economically viable. Replacing sludge holding tanks with APs increased methane yield by around 50 times, which suggests small scale electrical production is possible.

- To achieve extended sludge storage up to 10 times as proposed, an extended land area is demanded to support an extended HRT. Whilst potentially constraining for large-scale WWTW in urbanised areas, their application at small-scale, rural works is considered viable. Since up to 80 % of the solids separation occurred in the front third of the AP, scale could be considerably reduced.
- Based on the yield in this study, 0.25 kW_e of electrical generation capacity is required per 100 PE, indicating payback of around three years. However, the increase in methane yield between winters suggests a higher yield is possible with longer operation and design improvements.
- 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.

3.6 References

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4 Development of a staged anaerobic pond design through pilot trials and computational fluid dynamics

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Abstract

Since their inception as a roughing stage for larger pond treatment systems, the principle focus of anaerobic ponds APs has shifted from solids removal to optimising biogas production for renewable energy, and smaller physical footprints to reduce land requirements. In this study, a horizontally baffled (HBAP) and vertically baffled (VBAP) anaerobic were first compared before a staged pond was trialled. Distinct differences in removal performance of COD fractions were observed between the single stage baffled reactors, with particulate COD removal of 78 % in the HBAP cf. 32 % in the VBAP and soluble COD removal of -26 % in the HBAP cf. 19 % in the VBAP, (temperature 17.2-20.1 ^{o}C, mean 18.1 ^{o}C). A staged reactor (SAP) was constructed by placing the HBAP upstream of the VBAP, with an additional HBAP used as a control pond (CAP). No significant differences in removal performance was observed between the CAP and SAP (temperature 5.0-14.3 ^{o}C, mean 9.7 ^{o}C) however, methane biogas production at the end of the study were 6.09 and 9.04 $LCH_4\ m^{-3}$WWT for the CAP and SAP, respectively, despite the decrease in temperature. Specific methanogenic activity was found to be higher closer to the outlet for both CAP and SAP, suggesting active biomass despite low sludge volumes and reduced substrate availability. Hydrogenotrophic methanogenesis was found to dominate over aceticlastic, which has been found previously at low temperature and may explain the lack of acetate utilisation. Extended operation of the reactors, and trials with increased loading rates, may lead to greater distinctions between the single and two staged designs.

Keywords: Waste stabilisation lagoons, psychrophilic wastewater treatment, biogas

4.1 Introduction

Energy demand, greenhouse gas (GHG) emissions, and operational costs are continuing to rise year on year in the wastewater treatment sector, with traditional treatment options, such as primary sedimentation tanks and the activated sludge process (ASP), unable to provide sustainable solutions to increasing volumes and tightening quality standards (Chapter 1). These processes produce inherent fugitive GHG emissions, whilst also generating large quantities of sludge for disposal, and in the case of ASP high electrical demands for aeration (McAdam et al., 2011). Anaerobic ponds (APs) present an exciting opportunity to curb these trends by delivering three key benefits for more sustainable domestic wastewater treatment: a reduction in organic carbon load onto secondary aerobic treatment processes reduces electrical energy demand (McAdam et al., 2012); low energy demand and low sludge management requirements (Alexiou and Mara, 2003) provide a small energy and carbon footprint; and the retained carbon can be converted to biogas for subsequent utilisation in renewable energy generation (Shilton et al., 2008).

Anaerobic ponds were originally developed as a pre-treatment stage in larger pond systems (Pescod, 1996) to decrease particulate loading on downstream facultative and maturation ponds. In such systems, design loading rates were developed through empirical observation and were deliberately conservative in order to minimise odour nuisance from the uncovered ponds, thereby inhibiting the potential for biogas production (Park and Craggs, 2007). The covering of APs is now recommended for environmental protection (Noyola et al., 2006) and energy capture (Park and Craggs, 2007). As the role of APs changes from primary sedimentation to more complete organic breakdown and flexibility within treatment flowsheets, a new design approach is required that is focused on optimising the biological processes within the ponds whilst reducing physical footprint, alongside the traditional requirement of solids removal. The separation of solids retention time (SRT) from hydraulic retention time (HRT) is vital, to ensure sufficient retention and degradation time for particulate carbon, whilst contact between the retained biomass and the liquid layer must also be facilitated to target soluble carbon fractions that are an essential step in methanogenesis (Lew et al., 2009).

Traditionally APs have been designed as single-stage unbaffled reactors, rectangular in shape with a recommended 3:1 length:width ratio, and designed for a recommended HRT between one and three days, depending on the operating temperature (Mara and Pearson, 1998). However, recent studies on APs and anaerobic baffled reactors (ABRs) have investigated the incorporation of baffles to improve hydrodynamic performance and increase mixing (Peña et al., 2003; Langenhoff and Stuckey, 2000). Horizontal baffles, which produce a lane system creating 'side to side' flow, move the flow regime closer toward plug flow conditions (Muttamara and Puetpaiboon, 1997), thereby maximising sedimentation. In contrast, vertical baffles create 'up-and-under' flow, which provides greater biomass contact and has been demonstrated to separate the stages of anaerobic digestion along the reactor length, with acidogenesis observed in the compartments closest to the inlet and methanogenesis further down the reactor (Barber and Stuckey, 1999). The development of specific microbial communities within each chamber was observed, and this separation, due to preferential conditions for differing but complementary communities along the reactor length, has been found to increase acidogenic and methanogenic activity by up to a factor of four (Barber and Stuckey, 1999).

The incorporation of baffles into APs will affect the flow profile through the pond, and quantifying changes in hydrodynamics facilitates greater understanding of pond treatment mechanisms (Peña et al., 2003; Persson and Wittgren, 2003; Abbas et al., 2006; Abbassi et al., 2009). Hydrodynamic performance of ponds can be assessed from residence time distribution (RTD) data through a variety of established analytical tools. The most common is the comparison of the theoretical HRT (HRT_t, defined as pond volume divided by the flow rate), with the actual HRT (HRT_a), calculated from collected RTD data, used to assess hydrodynamic efficiency (Abbas et al., 2006). When $HRT_a < HRT_t$ the pond volume is not fully utilised, leading to hydraulic dead space (Moreno, 1990) and preferential flow patterns, expressed by the short circuiting quotient, S (Persson, 2000). Variance of RTD, σ^2 (Muttamara and Puetpaiboon, 1997), dispersion number, δ (Abbassi et al., 2009), and the tanks in series model (Persson, 2000) analyse the flow regime between completely mixed and plug flow, with pond systems traditionally designed for plug flow (Persson and Wittgren, 2003; Abbassi et al., 2009). Separately these tools can be used to scrutinise certain aspects of pond hydrodynamics. However, they are rarely analysed together to form a holistic diagnosis (Persson, 2000).

Whilst the hydrodynamic performance of ponds has traditionally been analysed through experimental tracer studies, computational fluid dynamics (CFD) modelling has become an increasingly more powerful and accessible tool for pond designers since its first application for this purpose by Wood et al. (1995). Studies using CFD to investigate pond design have been numerous, and have included pond geometry, inlet and outlet location, and various horizontal baffling configurations (Wood et al., 1995; Persson, 2000; Salter et al., 2000; Vega et al., 2003; Shilton and Mara, 2005). However, most lack validation through comparison with experimental data (Shilton et al., 2008; Alvarado et al., 2012). Additionally, the majority of studies reported have been conducted on facultative or maturation ponds, with a focus on achieving plug flow conditions (Shilton and Harrison, 2003), whereas the importance of mixing for biomass contact with the liquid layer is being increasingly recognised in APs (Peña et al., 2003). Furthermore, whilst the evolution of CFD models from two to three dimensions has led to increased modelling potential, to date vertical baffles have not been studied. The use of CFD modelling, if suitably validated with experimental tracer studies, can provide insight into intra-pond flow characteristics that are not possible from merely analysing tracer study data (Shilton, 2000).

This paper reports on the development of a staged anaerobic pond (SAP), developed through initial study of horizontally (HBAP) and vertically (VBAP) baffled anaerobic ponds, through pilot scale trials and CFD modelling. The aim of the study was to assess the effect of differing baffle orientations in single stage reactors, and subsequently between a two stage and single stage AP. The aim was achieved through three objectives:

1. Determine how baffles affect the hydrodynamic characteristics and flow regimes within APs through CFD modelling and experimental validation
2. Compare the hydrodynamic profiles acquired in objective 1 to removal efficiencies and biogas production of AP designs operated on real wastewater, to determine how baffle design affects overall AP performance
3. Through investigation of internal sampling within the pilot APs, identify the impact of individual baffles in separating solids retention biological activity for fractionated carbon degradation and methanogenic activity along reactor length, and how this contributes to the overall performance of the AP design identified in objective 2

4.2 Materials and Methods

4.2.1 Experimental set-up

Two pilot-scale reactors were constructed of 12 mm thick uPVC sheeting and sealed with PVC hot welding. The internal dimensions were 1.5 m x 0.5 m x 0.25 m for the VBAP and 1.5 m x 0.5 m x 0.31 m for the HBAP, giving hydraulic volumes of 188 L and 230 L, respectively. A 3:1 length:width ratio was used in accordance with recommended AP design (Mara and Pearson, 1998). The VBAP contained four baffles located at L/5, 2L/5, 3L/5 and 4L/5, which extended the entire width of the reactor and 80 % of its height. The baffles alternated between sitting on the base of the reactor, thus forcing flow over the baffle, and standing against the lid of the reactor, forcing flow under the baffle (Figure 4-1). The HBAP contained two baffles, located at L/3 and 2L/3 along the reactor length, which extended the entire height of the reactor and 85 % of the reactor width (Peña et al., 2003). The reactors were sealed with gas-tight lids.

The SAP was created by connecting the two in series, with the HBAP located upstream of the VBAP. A control pond (CAP) was constructed with the same specifications as the original HBAP. Side ports were fitted to the CAP and SAP for sampling from each chamber created by the baffles. The side ports were labelled C1, C2 and C3 for the CAP chambers; H1, H2 and H3 for the chambers in the first (HBAP) stage and V1, V2, V3, V4 and V5 for the chambers in the second (VBAP) stage of the SAP. All reactors were

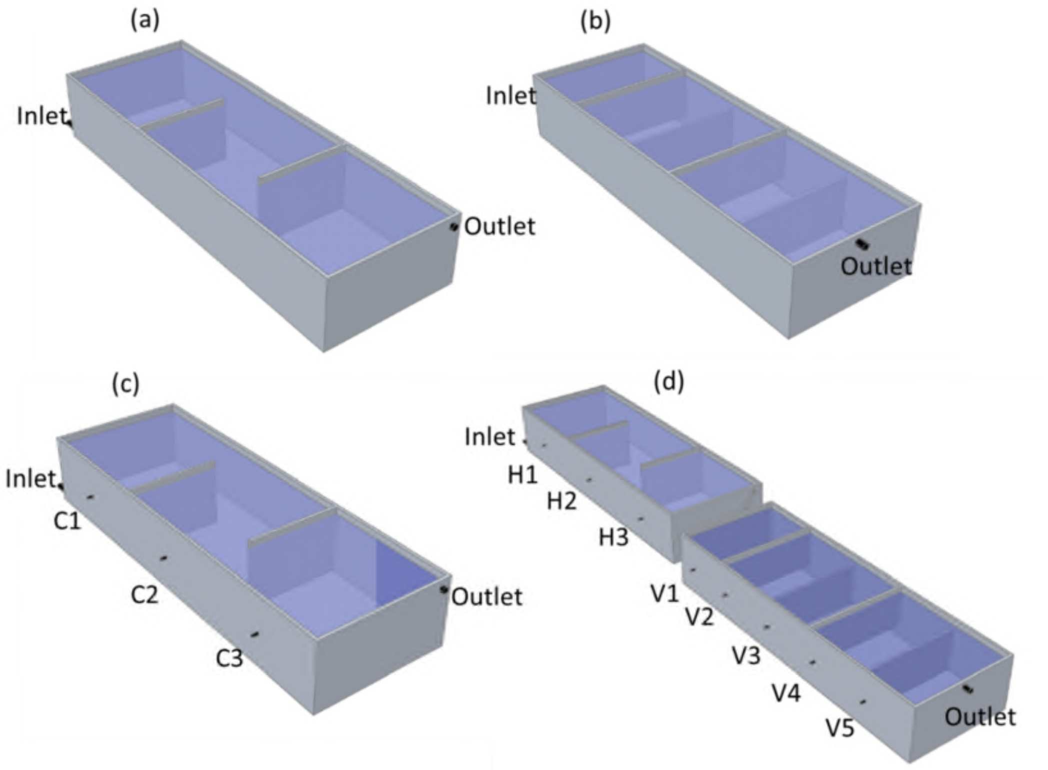

Figure 4-1 Layouts of the reactors used in the study. The (a) horizontally baffled anaerobic pond (HBAP), (b) vertically baffled anaerobic pond (VBAP), (c) control pond (CAP) and (d) staged pond (SAP).

initially seeded at 7 % of their volume with mesophilic anaerobic sludge from a digester (volatile solids, VS = 36 g L^{-1}), filled with crude wastewater from the Cranfield University sewage treatment works and left in batch for one day. They were then fed continuously with crude wastewater at a liquid flow rate of 75 L d^{-1}. The SAP was operated at a flow rate 150 L d^{-1} to produce the same HRT as the control. The HBAP and VBAP were operated for 43 days, during which time the ambient temperature ranged from 17.2 ^{o}C to 20.1 ^{o}C, with a mean of 18.1 ^{o}C. The SAP and CAP were operated for 111 days, with a temperature range of 5.0 ^{o}C to 14.3 ^{o}C, mean 9.7 ^{o}C. For the CFD validation only, an unbaffled pond (UAP) was created by removing the baffles from the HBAP. Tracer studies for CFD validation experiments were conducted in all reactors operating with water only and without seed.

4.2.2 Analytical methods

Influent and effluent were analysed three times a week in duplicate, whilst internal sampling in the SAP trial was conducted once a month. Total suspended solids (TSS), volatile suspended solids (VSS), total COD (tCOD) and soluble COD (sCOD), biochemical oxygen demand (BOD_5) were measured according to standard methods (APHA, 1998). Samples for sCOD were filtered through a 1.2 µm glass fibre filter (Whatman, Maidstone, UK). Particulate COD fraction (pCOD) was calculated by subtracting sCOD from tCOD. Ambient and liquid temperatures were recorded at the time of sampling. Six volatile fatty acids (VFA) were measured using high performance liquid chromatography (HPLC) in a fermentation separation column (Bio-Rad, California, USA). Biogas was captured from the lids of the reactors in gas-tight sampling bags and analysed twice a week for total volume and gas composition. Gas volume was measured through displacement (Mshandete et al., 2005) whilst composition was measured by gas chromatography with a thermal conductivity detector (CSi 200 Series, Cambridge Scientific Instruments Ltd, Cambridge, UK). Specific methanogenic activity (SMA) tests were carried out in triplicate for sludge samples taken from each chamber from the CAP and SAP at the end of the study, with separate tests for hydrogen and acetate substrates. All SMA assays were prepared and analysed according to Collins et al. (2003).

Tracer studies were performed with Lithium (Li^+). A LiCl (>99 % reagent grade) solution of 306 g L^{-1} was prepared, and a pulse signal of 4 mL de-ionised water was introduced to the influent, for a total pulse Li^+ mass of 200 mg. Grab samples were collected in the effluents at regular intervals over a period equivalent to 3 HRTs. Control samples were taken prior to dosing to analyse for background Li^+. Lithium concentrations were determined by atomic emission spectroscopy (Perkin Elmer model AAnalyst 800, using an air-acetylene flame method at 670.80 nm) with a minimum detection limit of 0.05 mg L^{-1}.

4.2.3 CFD modelling

Computational fluids dynamics (CFD) modelling was undertaken to model the flow patterns within the ponds (Appendix A). Three dimension single phase CFD simulations were performed using the commercial software FLUENT v14.0.0 (ANSYS). Geometries for the four reactor designs were drawn in AutoCAD 2007 (AutoDesk, Inc.) and meshed using ICEM CFD (ANSYS). Meshes contained total elements of 1,535,058 for the UAP, 2,234,971 for the HBAP, 2,060,338 for the VBAP and 3,012,830 for the SAP. The laminar flow model was used as the Reynolds number for all ponds was less than 6. The fluid in the ponds was assumed to be incompressible and exhibiting Newtonian fluid properties of water with a density of 998.2 kg m^{-3} and a dynamic viscosity of 1.003×10^{-3} kg m^{-1} s^{-1}. The inflow boundary condition was defined as a mass-flow-inlet with a constant mass flow of 1.3×10^{-3} kg s^{-1} for the single stage reactors and 2.6×10^{-3} kg s^{-1} for the staged reactor. The outflow boundary condition was defined as a pressure-outlet with a gauge pressure of 0 pascal. The tracer RTD analysis was performed by imposing a transient simulation of the tracer as a scalar on the velocity and turbulent fields obtained from the flow simulation using the method proposed by Alvarado et al. (2012).

4.3 Results

4.3.1 Hydrodynamic studies and validation of CFD modelling

Validation of CFD models was conducted through comparison of hydrodynamic indicators calculated from the RTD data collected from CFD simulations and empirical tracer studies. Whilst a good fit was obtained between the computational and empirical data, divergences in all reactors were found to be complimentary in identifying areas where the CFD models altered from empirical findings. The CFD simulations indicated higher degrees of short-circuiting than the empirical data, evidenced through lower short-circuiting quotients (S) recorded in all CFD simulations than the empirical counterparts (mean difference across all four reactors studied, μ_{diff} = 0.09, σ = 0.03, n = 4). This is supported by lower HRT_a values for all CFD simulations relative to collected empirical data, and higher calculated dead space percentage (μ_{diff}

= 0.46 d, σ = 0.10, n = 4 and μ_{diff} = -20 %, σ = 4, n = 4 for HRT_a and dead space percentage, respectively). Additionally, the CFD simulations indicate more plug flow characteristics than the empirical data. Lower variance was observed in all CFD simulations compared to empirical runs (μ_{diff} = 0.46 d, σ = 0.10, n = 4), whilst higher N values from the tanks in series model (μ_{diff} = -3.00 d, σ = 1.638, n = 4) also demonstrate the CFD simulations portrayed more plug flow conditions than the empirical data. The dispersion numbers showed the largest correlation between the CFD and empirical RTDs, with differences of 0.00, -0.07 and 0.01 calculated for the HBAP, VBAP, and SAP, respectively. The dispersion number is a function of the variance and measured HRT, and the differences in these values are offset in the calculation to provide the close correlation. The exception was the UAP, where a difference of -0.24 was calculated (δ = 0.26 for empirical RTD *cf.* δ = 0.50 for the CFD RTD), indicating more plug flow conditions in the empirical RTD.

The benefits of baffling in improving pond hydrodynamics were evident in both the CFD simulations and the experimental tracer studies. The most efficient hydrodynamic design of the four configurations studied was the SAP, with a dead space volume of 10 % and a short circuiting quotient, S, of 0.47 from the experimental RTD and dead space volume = 36 %, S = 0.36 for the CFD simulation (Table 4-1). Short circuiting quotients were similar in the baffled single stage ponds, with S_{exp} = 0.43, S_{CFD} = 0.32 for the VBAP *cf.* S_{exp} = 0.40, S_{CFD} = 0.29 for the HBAP, whilst a higher degree of short circuiting was evident in the unbaffled pond, demonstrated by the lowest short circuiting quotient, S_{exp} = 0.22, S_{CFD} = 0.18. Dead space volumes were also similar in the single stage baffled ponds, with 43 % recorded in both the HBAP and VBAP in the CFD simulations, and 20 and 27 % in the tracer studies for the HBAP and VBAP, respectively. Whilst overall hydrodynamic efficiency, measured through dead space and short circuiting, were similar in the HBAP and VBAP, differences were evident in the flow regimes. The VBAP created more plug flow conditions, with lower dispersion numbers and higher tanks in series (Table 4-1). The most plug flow conditions were found in the SAP, with dispersion numbers δ_{exp} = 0.10, 0.16, 0.15 and 0.26, and δ_{CFD} = 0.12, 0.15, 0.23, 0.50 for the SAP, VBAP, HBAP and UAP, respectively. The tanks in series models supported this finding, with the SAP and VBAP having similar N values, VBAP N_{exp} = 7.43, N_{CFD} = 11.93 *cf.* SAP N_{exp} =7.06, N_{CFD} = 11.43, whilst the lowest values were found in the unbaffled case, UAP N_{exp} = 4.49, N_{CFD} = 4.95.

The velocity profiles generated in the CFD simulations provide further insight into the flow mechanisms generating the hydrodynamic data. In the UAP, where a high degree of short circuiting was calculated, a clear preferential flow pattern can be observed passing from the inlet directly to the outlet (Figure 4-2). Recirculation, caused by the small area of the outlet compared to the flow rate, generate a back-mixing effect, although dead space is evident in the corners of the pond. In the baffled ponds, the preferential flow pattern is disrupted by the baffles, which generate their own back-mixing effect. Recirculation between baffles is evident, which reduces dead space by utilising more of the pond volume, whilst creating an overall plug flow effect through the sequential detention of the flow in each chamber. This effect is more pronounced at higher velocities, as recirculation is evident in all three chambers of the horizontally baffled section of the SAP, whereas in the single stage HBAP, recirculation occurs in the front chamber but a preferential flow pattern is evident in subsequent chambers at the lower velocities. Whilst more plug flow conditions were found with vertical baffles in the single stage ponds, this may also be a factor of the number of baffles, with the SAP generating greatest plug flow with the highest number of baffles (6 between the two stages), followed by the VBAP (4 baffles) then the HBAP (2 baffles).

Table 4-1 Hydrodynamic data calculated for four anaerobic pond designs with experimental tracer studies and computational fluid dynamics modelling

	UAP			VBAP			HBAP			SAP			Difference	
	Exp.	CFD	Diff	Exp.	CFD	Diff	Exp.	CFD	Diff	Exp.	CFD	Diff	μ_{diff}	σ
HRT_a (d)	1.75	1.38	0.37	1.69	1.33	0.36	1.85	1.33	0.52	2.07	1.48	0.59	0.46	0.10
HRT_a/HRT_t (%)	76	60	16	73	57	16	80	57	23	90	64	26	20	4
Short circuiting quotient, S	0.22	0.18	0.04	0.43	0.32	0.11	0.40	0.29	0.11	0.47	0.36	0.11	0.09	0.03
Dead space volume (%)	24	40	-16	27	43	-16	20	43	-23	10	36	-26	-20	4
Variance, σ^2 ($days^2$)	1.20	1.08	0.12	0.72	0.45	0.27	0.90	0.62	0.28	0.76	0.47	0.29	0.24	0.07
Dispersion number, δ	0.26	0.50	-0.24	0.15	0.15	0	0.16	0.23	-0.07	0.10	0.12	0.01	-0.07	0.10
Tanks in series, N	4.49	4.95	-0.46	7.43	11.93	-4.50	5.96	8.61	-2.65	7.06	11.43	-4.37	-3.00	1.64
Tracer recovered (%)	102	68	34	94	94	0	100	91	9	110	94	16	15	12
Maximum velocity v_{max} ($m\ s^{-1}$)	N/A	1.47×10^{-2}	N/A	N/A	1.37×10^{-2}	N/A	N/A	1.52×10^{-2}	N/A	N/A	1.04×10^{-1}	N/A	N/A	N/A
Minimum velocity v_{min} ($m\ s^{-1}$)	N/A	1.21×10^{-9}	N/A	N/A	7.10×10^{-11}	N/A	N/A	1.83×10^{-9}	N/A	N/A	7.22×10^{-8}	N/A	N/A	N/A

UAP – Unbaffled anaerobic pond; VBAP – vertically baffled anaerobic pond; HBAP – horizontally baffled anaerobic pond; SAP = staged anaerobic pond; Exp. – experimental tracer study data; CFD – computational fluid dynamics simulation data; Diff = difference between experimental and CFD values; μ_{diff} – mean difference between experimental and CFD for all cases; σ – standard deviation of μ_{diff}

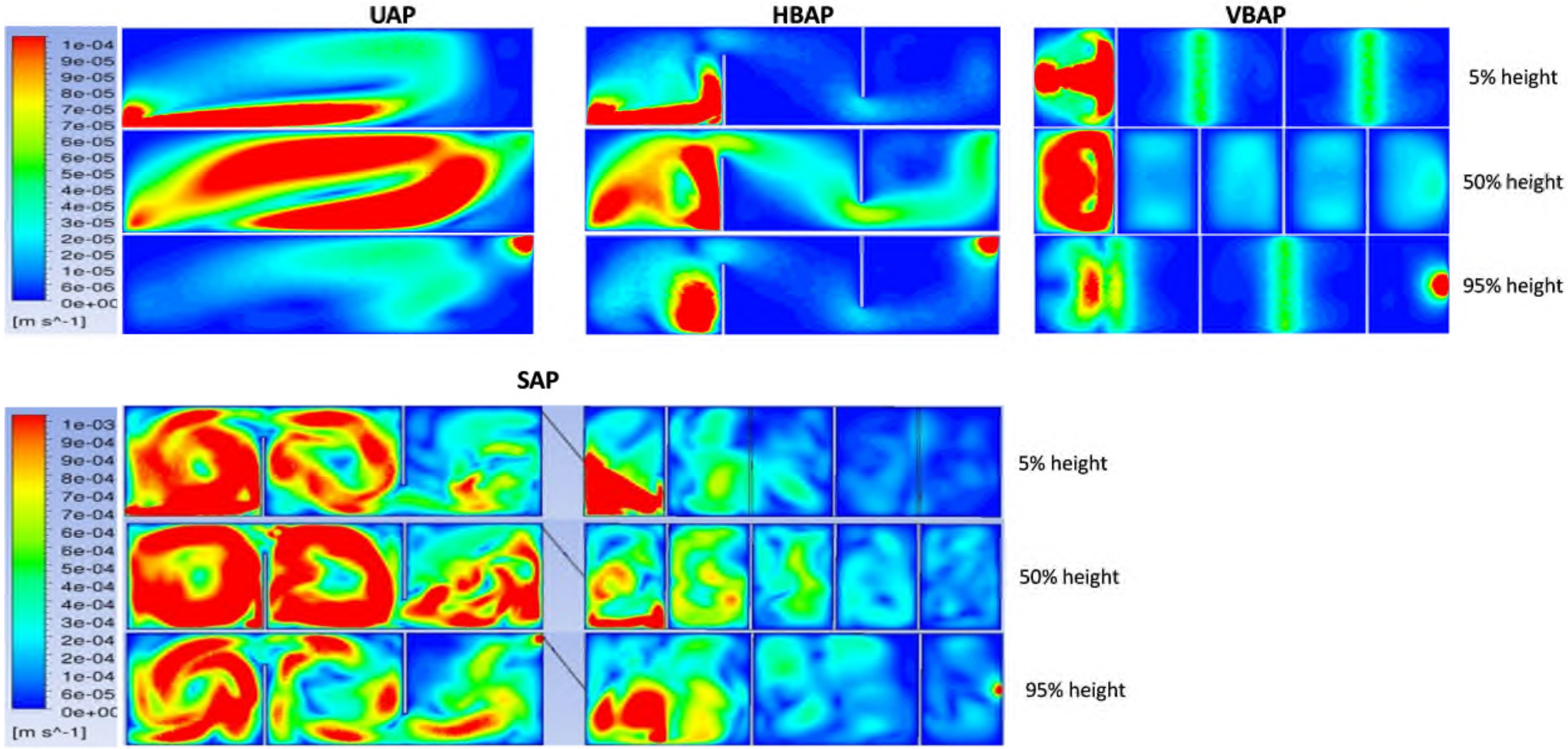

Figure 4-2 Velocity profiles generated from computational fluid dynamics for the unbaffled (UAP), horizontally baffled (HBAP), vertically baffled (VBAP) and the staged anaerobic ponds (SAP).

4.3.2 Comparison of the horizontally and vertically baffled ponds

In the wastewater trials, high and consistent particulate removal in the HBAP contrasted lower and more variable removal in the VBAP (Figure 4-3). To illustrate, mean TSS removal efficiency in the HBAP was 80 ±9 % (n = 14) *cf.* 35 ±15 % (n = 20) in the VBAP. This corresponded to mean pCOD removal of 73 ±21 % in the HBAP compared to 32 ±32 % in the VBAP. By contrast, a mean sCOD removal of -15 % was recorded in the HBAP whilst positive removal of 21 % was recorded in the VBAP.

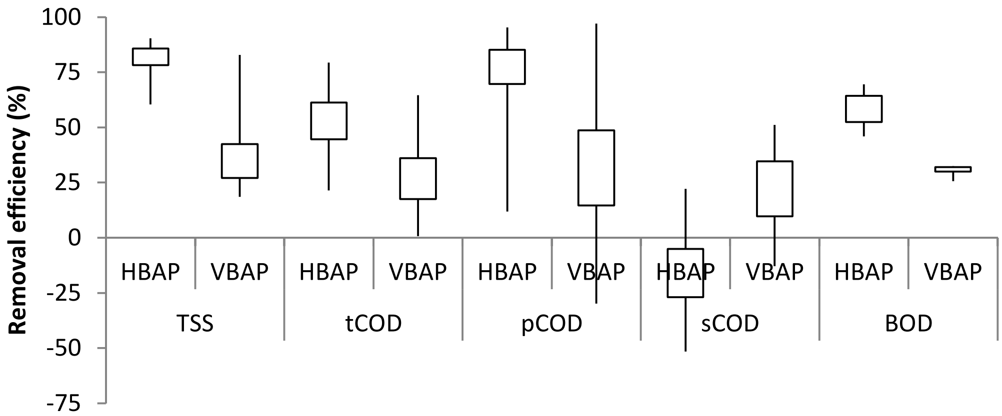

Figure 4-3 Removal efficiencies from the horizontally (HBAP) and vertically baffled (VBAP) anaerobic ponds.

4.3.3 Pilot trial of a staged anaerobic pond design

There were no statistical differences between the removal efficiencies of the CAP and SAP for any of the measured sanitary parameters to a 95 % confidence level (Figure 4-4). Analysis of variance tests were carried out on the data sets from the staged pond trial, with unpaired t-tests used for normally distributed data sets and Mann-Whitney tests for non-parametric data. Particulate removal was concentrated at the front of both reactors, with 65 % of total TSS removal observed in the first chamber of the CAP (equal to 33 % of total reactor length), and 85 % observed in the front chamber of the SAP (equal to 17 % of total reactor length). Total sludge accumulation in the front chamber of each reactor was 15.0 and 20.6 L for the CAP and SAP respectively (Table 4-2), comprising 63 and 39 % of the total sludge volume for each reactor, suggesting the settlement of solids is more dependent on the baffle placement than on reactor length. Sludge in the second (vertically baffled) stage of the SAP was evenly distributed, with 4.1 L observed in the first chamber, V1, and 4.0 L observed in the final chamber, V5, suggesting there was little sludge carry-over from the first stage, and the initial seed remained immobilised in the respective chambers.

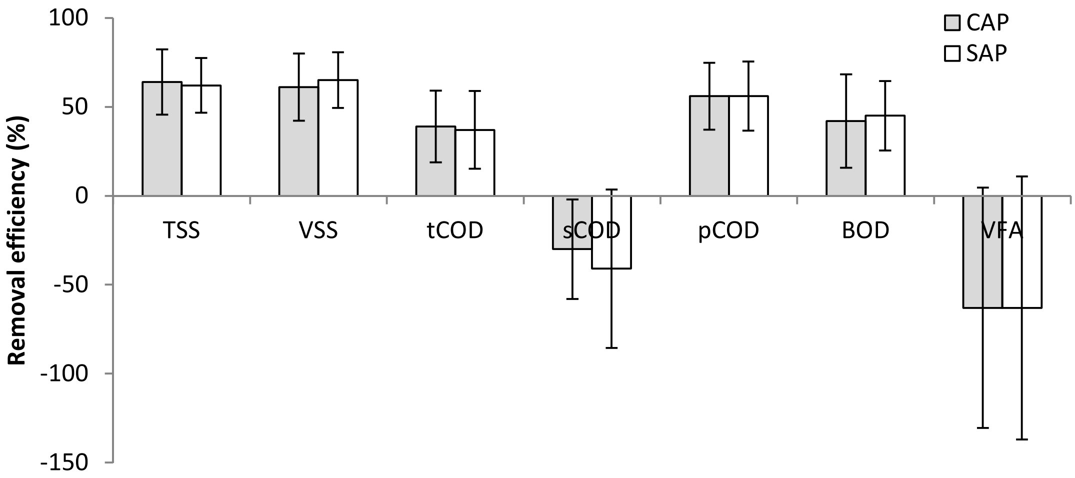

Figure 4-4 Removal efficiencies from the control (CAP) and staged (SAP) anaerobic pond trial.

Negative sCOD removal was experienced in both ponds (Figure 4-4), with mean -30 ±28 % and -41 ±45 % for the CAP and SAP, respectively. No relationship was observed between negative sCOD removal and time in either pond, suggesting rapid

Table 4-2 Performance data from the staged anaerobic pond trial

	Sampling point	Length along pond		Concentration									Acetic acid proportion of total VFA	Vol acc. sludge	Biogas methane production	SMA Acetate	SMA Hydrogen
				TSS	VSS	tCOD	sCOD	pCOD	BOD_5	Alk	Total VFA	Acetic acid					
		L/L_{total}	pH	mg L^{-1}									%	L	$LCH_4\ m^{-3}WWT$	$mgCH_4\ gVSS^{-1}\ d^{-1}$	$mgCH_4\ gVSS^{-1}\ d^{-1}$
	Inf	N/A	8.0	277	235	451	87	364	196	182	102	22	22	N/A	N/A	N/A	N/A
Control pond	C1	0.17	7.22	151	132	309	151	158	177	195	154	74	48	15.0	2.92	0.61	13.94
	C2	0.50	7.18	148	128	312	136	176	115	209	171	91	53	5.6	0.26	1.77	956.95
	C3	0.83	7.18	117	98	279	131	149	106	217	148	78	53	3.1	0.19	0.73	841.03
	CAP eff	1.00	7.60	80	76	239	109	130	93	200	134	64	48	23.7*	3.36*	1.04**	604.03**
Two-stage pond	H1	0.13	7.34	612	535	1035	85	950	252	384	126	59	47	20.6	4.19	8.79	359.66
	H2	0.25	7.28	119	104	261	97	164	72	401	139	63	45	7.9	0.15	N/A	N/A
	H3	0.38	7.37	97	88	259	78	181	66	408	116	51	44	5.9	0.04	11.52	54.32
	V1	0.55	7.59	99	86	221	70	151	63	387	79	27	34	4.1	0.28	16.17	2,829.37
	V2	0.65	7.40	71	62	214	101	113	66	428	106	57	54	3.5	N/A	0.24	532.50
	V3	0.75	7.56	70	57	220	89	132	66	395	95	33	35	3.9	0.33	7.06	936.31
	V4	0.85	7.47	86	75	242	107	135	61	442	121	65	54	3.3	N/A	0.02	4,144.58
	V5	0.95	7.53	77	71	210	86	124	57	421	97	46	47	4.0	0.04	0.14	1,369.48
	SAP eff	1.00	7.60	91	72	245	114	132	91	208	136	64	47	53.1*	5.03*	6.28**	1,460.49**

TSS – total suspended solids, VSS – volatile suspended solids, tCOD – total chemical oxygen demand, sCOD – soluble chemical oxygen demand (<1.2 µm), pCOD – particulate chemical oxygen demand (>1.2 µm), BOD_5 – 5 day biochemical oxygen demand, Alk – alkalinity, VFA – volatile fatty acid, Vol acc. sludge – total accumulated sludge volume for chamber, SMA – specific methanogenic activity

* total for entire pond, ** weighted mean average for entire pond

initiation of acidogenesis without a start-up trend. The creation of sCOD in both ponds can be linked to VFA creation, as negative VFA removal was also experienced in both reactors. Acetic acid comprised 45 % of total measured VFA in both reactors (Table 4-2), suggesting a significant amount of acetate was not only generated in the ponds, but was still available as substrate throughout both ponds. Mean net $SMA_{hydrogen}$ = 604.03 $mLCH_4$ $gVSS^{-1}$ d^{-1} *cf.* $SMA_{acetate}$ = 1.04 $mLCH_4$ $gVSS^{-1}$ d^{-1} in the CAP, and mean net $SMA_{hydrogen}$ = 1,460.89 $mLCH_4$ $gVSS^{-1}$ d^{-1} *cf.* $SMA_{acetate}$ = 6.28 $mLCH_4$ g VSS^{-1} d^{-1} in the SAP, suggest hydrogenotrophic methanogenesis is the dominant digestion pathway within the ponds. In the CAP, highest SMA was found in the centre of the pond, C2, for both acetate and hydrogen substrates, whilst in the SAP, with the highest activity found in chamber V1 for acetate and chamber V4 for hydrogen (Table 4-2). Interestingly, SMA did not correlate with methane biogas production rates, which were found in the front chamber of both reactors. In C1 of the CAP headspace, biogas methane production was 2.92 LCH_4 m^{-3}WWT, or 87 % of total methane biogas production, whilst production in H1 of the SAP was 4.19 LCH_4 m^{-3}WWT, or 83 % of total SAP biogas methane production. Maximum biogas production measured in the second stage of the SAP was 2.23 LCH_4 m^{-3}WWT with a maximum biogas methane composition of 20 % *cf.* 16.63 LCH_4 m^{-3}WWT and 71 % for the first stage.

In contrast to VFA formation, start-up of overall biogas methane production was similar in both the CAP and the SAP, with a lag of 45 days before production was observed, and then increasing production until day 80 (Figure 4-5). Mean production rates recorded for the final two weeks of the study were 6.09 and 9.04 LCH_4 m^{-3}WWT. Whilst maxima found in the CAP are comparable to the SAP, low values were also measured throughout the study, whilst the SAP produced more consistent measurements (Figure 4-5). To illustrate, the range of production rates observed over the final two weeks of the study was 22.16 LCH_4 m^{-3}WWT for the CAP *cf.* 13.28 LCH_4 m^{-3}WWT for the SAP.

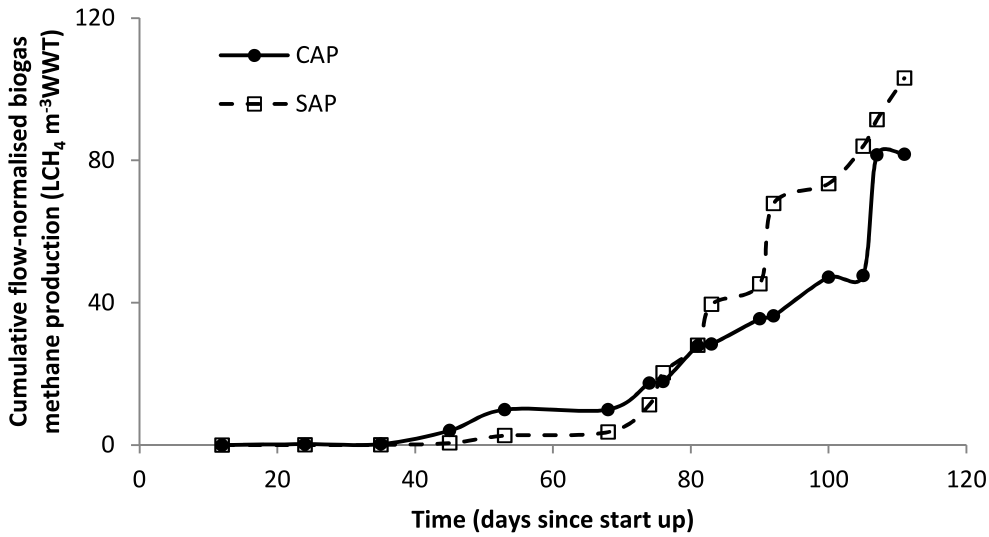

Figure 4-5 Cumulative flow-normalised biogas methane production in the control (CAP) and-staged (SAP) anaerobic ponds.

4.4 Discussion

4.4.1 Influence of baffle orientation on pond hydrodynamics and removal efficiency

The baffled AP designs were found to reduce short circuiting through the ponds compared to the unbaffled case, by dissipating the inlet jetting effect (Persson, 2000; Shilton and Harrison, 2003; Agunwamba, 2006) and creating recirculation between baffles (Shilton, 2000). The lower dispersion numbers observed in the baffled systems, indicating plug flow when increased mixing is expected, could appear counter-intuitive. However, the recirculation effect in the baffled reactors, caused by backpressure at each baffle and seen in the velocity profiles, generates mixing within each chamber whilst creating an overall plug flow effect of a series of stirred tanks (Grobicki and Stuckey, 1992). Recirculation was most pronounced in chamber 1 of both the HBAP and VBAP, with preferential flow patterns evident thereafter, suggesting baffle number may not have been a significant factor. Shilton and Harrison (2003) found that whilst a minimum of two baffles should be recommended, only small improvements are found with four baffles with further diminishing returns with increasing number of baffles. Whilst the merits of baffling against the unbaffled were evident, inconclusive results from comparison of the single stage baffled reactors suggest that for clean water trials, baffle orientation may not have a significant effect on pond hydrodynamics.

However, clear differences were observed between the baffle configurations in the wastewater trials. The removal efficiencies observed in the HBAP, compared to the VBAP, demonstrate that in single stage systems baffle orientation can have a distinct effect on removal performance. High and stable removal of particulates in the HBAP contrasted unstable particulate removal in the VBAP, and suggests the HBAP is more suited for primary treatment to capture particulates. Conversely, superior soluble carbon removal found in the VBAP, 19 % sCOD removal *cf.* -15 % in the HBAP, suggests that the vertical baffling can target soluble carbon fractions more effectively than horizontal baffles. These differences in removal performance lead to the development of a staged design, with the HBAP placed upstream of the VBAP, to maximise solids breakdown in the first stage and provide a soluble carbon substrate for degradation through enhanced mixing in the second stage (Lettinga et al., 2001; Van Haandel et al., 2006).

In all AP designs studied, the CFD simulations provided close correlation to experimental trials. Small changes observed, with greater short circuiting and more plug-flow characteristics found in the CFD, suggest a higher degree of mixing in the experimental tanks than was modelled computationally. This may be accounted for by the lack of thermal convection in the CFD model, which could cause greater mixing, and whilst the differences were small in the pilot-scale models used this may need to be considered at full scale (Agunwamba, 2006; Kehl et al., 2009; Pedahzur et al, 1993). Furthermore, the studied was conducted on clean water only, and therefore many parameters have not been considered. Future advancements into multi-phase CFD models could incorporate solids transport and accumulation (Alvarado et al., 2012), biogas bubbling, and sludge and wastewater rheology, enabling modelling to more accurately reflect wastewater trials. However, at present, liquid-only models such as those conducted in this study are still valuable to provide insight into comparative flow characteristics between potential AP designs (Shilton, 2000). Whilst CFD modelling can be a useful tool in trialling a large number of potential designs without the time and expense of pilot trials, the similarity in the hydrodynamic modelling contrasts the clear differences observed in removal mechanisms in the wastewater trial and demonstrates the need for pilot trials to be conducted after use of CFD as an initial selective tool (Abbas et al., 2006; Abbassi et al., 2009).

4.4.2 Development of a staged anaerobic pond design

Particulate removal in the SAP was close to the midpoint of the single stage reactors (pCOD removal 56 % for the SAP *cf.* 78 % for the HBAP and 32 % for the VBAP), whilst negative soluble sCOD was still observed. Accordingly, positioning the HBAP upstream of the VBAP did provide particulate retention and therefore reduce biomass washout from the VBAP, however, the function of the VBAP fundamentally changed from operating as a primary stage to a secondary stage of treatment. Whilst soluble carbon degradation in the single stage VBAP is likely to have been driven by the biomass/substrate contact provided by both baffle orientation and the volume of biomass retained, particularly by the first baffle, the absence of such volumes of biomass in the second stage of the SAP may have reduced its effectiveness. With time, biomass build up in the second stage may improve soluble carbon degradation. Alternatively, whilst the SAP was seeded with 17 % v/v sludge, ABRs have previously been seeded with sludge volumes up to 80 % (Barber and Stuckey, 1998; Langenhoff et al., 2000), and this may be required for more effective operation of the VBAP as the second stage. However, lager sludge volumes increase the risk of biomass washout, especially with the higher velocities applied to the two stage design (Dama et al., 2002).

In both the CAP and SAP, the first chamber, created between the inlet and first baffle, was found to be critical in the overall performance of the ponds. The vast majority of particulate removal, VFA generation, sludge accumulation and biogas production were found in this chamber in both ponds, irrespective that this chamber comprised a smaller proportion of the overall volume in the SAP than the CAP. Increased activity close to the inlet has been observed in unbaffled full scale ponds (Schneiter et al., 1993; Paing et al., 2000; Picot et al., 2005a), and can be attributed to the low flow rates applied to ponds leading to ineffective use of the entire pond volume. Higher loadings onto APs can lead to improved hydrodynamic performance, with increased mixing leading to greater biomass/substrate contact that is essential for soluble carbon breakdown (Peña et al., 2003), whilst also reducing the physical footprint (Li, 1992; Agunwamba, 2001). Higher loadings rates applied to the designs in this study may lead to greater differences between the single and two-stage systems, through driving increased adaptation of the differing microbial communities developing along the ponds' length by providing increased organic strength and accentuating the current differences in flow patterns due to baffle design.

The SMA assays found hydrogenotrophic methanogenesis to be dominant over aceticlastic pathways, meaning the methanogenic community preferentially metabolised hydrogen and carbon dioxide to methane rather than utilising acetate as the preferred substrate (Appendix A). This has been reported from other low temperature anaerobic studies (Collins et al., 2005; Connaughton et al., 2006), congruent with the lack of acetate targeting. The high SMA activity observed in the downstream chambers of the ponds demonstrates that the sludge in these areas is still active despite the small volumes (Paing et al., 2000). Previous investigations at full scale have shown that APs can take up to 2 years to mature, especially with respect to VFA degradation (Paing et al., 2000; Picot et al., 2003). Further, aceticlastic methanogens, which have lower kinetic rates than hydrogenotrophic orders and are more sensitive to lower temperatures (Connaughton et al., 2006), may establish in the ponds with extended operation time, especially in the SAP second stage where both the flow pattern and acetate rich substrate would provide preferential conditions for growth. The construction and location of the pilot models led to liquid temperatures closer to ambient temperatures than to the influent, whilst at full-scale buffering caused by surrounding earthworks would lead to a higher and more stable temperature range within the pond (Safley Jr. and Westerman, 1989; Park and Craggs, 2007). These low temperatures would also exacerbate the solution of methane within the liquid, and up to half the methane generated is likely to be lost in the effluent

(Noyola et al., 2006; Cookney et al., 2012), although this was not measured in the study. The ongoing establishment of microbial communities over extended pond operation and higher operational liquid temperatures should lead to increased organic degradation and subsequent biogas production if the designs were scaled up, whilst the recovery of dissolved methane would both increase methane recovery and reduce fugitive emissions from the effluent (Cookney et al., 2012).

4.5 Conclusions

The influence of baffle configurations on the performance of APs was studied across a broad spectrum of performance indicators, with a two stage design developed to optimise the findings from single stage horizontal and vertical baffle trials.

- The influence of baffling on pond hydrodynamics was demonstrated through experimental tracer and CFD modelling. Whilst plug flow tendencies were observed in the hydrodynamic data from baffled ponds, investigation of the CFD generated velocity profiles highlighted the recirculation within ponds, demonstrating the effectiveness of baffles in enhancing mixing whilst creating an overall plug flow effect
- Differences in removal mechanisms were found between horizontally and vertically baffled single stage APs, with horizontal baffles found to promote sedimentation and solids removal at the expense of soluble carbon washout, whilst the reverse was true of the vertically baffled AP
- A two stage AP design was developed, to promote sedimentation and solids breakdown in the first stage followed by targeting of the generated soluble fraction in the second stage. Whilst results at the low loading rates applied were not definitive, evidence suggests extended pond operation and higher loading rates may improve performance of the two stage AP
- Advantages of two stage system were found in improved hydrodynamic performance by optimising effective pond volume, higher and more stable biogas production compared to a single stage AP suggest more effective anaerobic breakdown, and evidence of the spatial distribution of the anaerobic

digestion process which may lead to more efficient anaerobic digestion with time as different microbial communities establish in the different preferential conditions created

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5 Performance of a two stage anaerobic pond at four hydraulic retention times

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Abstract

A two stage anaerobic pond (SAP) design was trialled against a single stage control (CAP) over four hydraulic retention times (HRTs). Experimental tracer studies were compared to CFD simulations, with the SAP showing greater hydraulic efficiency, and differences more pronounced at shorter HRTs. Greater flow recirculation between baffles was observed in CFD velocity profiles, demonstrating baffles can dissipate preferential flow patterns and utilise more effective pond volume, expecially at high flow rates. No statistical differences were observed in particulate removal between the ponds over all four HRTs, suggesting solids loading is not a critical factor in AP design, either for the use of baffling or design HRT. Biological activity was found to be more dependent on temperature than loading, although significantly higher biogas production rates were observed in the SAP than the CAP at 1.5 d and 1.0 d HRT, and microbial community profiling suggest the two stage design may be facilitating spatial separation of the anaerobic digestion process along reactor length. Hydrogenotrophic methanogensis was found to dominate over aceticlastic, with acetate oxidisation a likely degradation pathway. The study demonstrates both the potential of APs to be operated at shorter HRTs in psychrophilic conditions, as well as the opportunity for two stage designs to be investigated and developed to enhance the separate stages of the anaerobic digestion process through creating preferential conditions in different physical locations.

Keywords: psychrophilic; methane production; municipal wastewater

5.1 Introduction

The traditional approach to designing anaerobic ponds (APs) is currently being challenged, as the opportunities for shorter hydraulic retention times (HRTs) (Peña et al., 2003), the use of baffling (Peña et al., 2003; Vega et al., 2003; Shilton and Harrison, 2003), and the covering of APs for biogas collection (DeGarie et al., 2000; Parissopoulos et al., 2003; Noyola et al., 2006; Shilton et al., 2008) are being realised. Temperature-dependent design organic loading rates were developed through empirical observation, ranging from 100 gBOD m^{-3} d^{-1} for ambient temperatures < 10 ^{o}C, to 350 gBOD m^{-3} d^{-1} for temperatures >25 ^{o}C. The design loading rates were deliberately conservative, with the lower limit specified to ensure anaerobic conditions and the upper limit to minimise odour nuisance and the need for desludging (Mara and Pearson, 1998). In practice, even these conservative guidelines are rarely met, with odour nuisance cited as the most common reason for AP underloading (Pearson et al., 1996; Picot et al., 2005a; Archer and Mara, 2003; Alexiou and Mara, 2003). Covering of APs not only eliminates odour but reduces greenhouse gas emissions (Noyola et al., 2006), and the captured biogas can be used for energy generation thus providing an opportunity to reconsider appropriate loading rates based on the positive attributes of the technology rather than negating the negative ones (Hodgson and Paspaliaris, 1996; Park and Craggs, 2007). For instance, as the potential of APs for energy positive primary treatment has been recognised (McAdam et al., 2011), design focus is changing from primary sedimentation to more complete organic breakdown, with particular emphasis on identifying appropriate design geometry to maximise performance and reduce process scale (Vega et al., 2003; Agunwamba, 2006).

Currently, the costs associated with the associated extensive land requirements are the largest single barrier to uptake of APs (Xian-Wen, 1995; Agunwamba, 2001), with hydraulic retention times (HRTs) ranging from 1 and 4 days but most commonly between 2 and 3 days (Mara and Pearson, 1998). Reduction of land requirement, through shorter HRTs, improves the economic viability of APs whilst also offering process improvements. Higher organic loading rates provide more substrate for microbial growth, whilst the increased flow rates lead to greater mixing, reducing hydraulic dead space in the pond and facilitating biomass/substrate contact (Peña et al., 2003). However, shorter HRTs increase the potential for biomass washout, which must be avoided in order to allow sufficient solids retention time (SRT) within the process for degradation. For instance, 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. Therefore, separation of SRT from HRT is vital, to ensure sufficient retention and degradation time for particulate carbon, whilst contact between the retained biomass and the liquid layer must also be facilitated to target soluble carbon fractions that are an essential step in methanogenesis (Lettinga et al., 2001; Lew et al., 2009).

The separation of HRT and SRT can be facilitated through the use of baffling. Incorporation of baffles into passive treatment systems has been found to improve hydrodynamic performance and increase mixing (Peña et al., 2003; Langenhoff and Stuckey, 2000). Horizontal baffles, which produce a lane system creating 'side to side' flow, reduce hydraulic short circuiting and therefore promote sedimentation and particulate retention (Muttamara and Puetpaiboon, 1997). In contrast, vertical baffles create 'up-and-under' flow, which provides greater biomass contact and has been demonstrated to separate the stages of anaerobic digestion along the reactor length, with acidogenesis observed in the compartments closest to the inlet and methanogenesis further down the reactor in anaerobic baffled reactors (ABRs), increasing acidogenic and methanogenic activity by up to a factor of four (Barber and Stuckey, 1999). The incorporation of baffles into anaerobic reactors has led to the development of high-rate anaerobic ponds with 0.5 day HRTs (Peña et al., 2003), and ABRs with typical HRT <1 day, and as low as 1 hour (Barber and Stuckey, 1999).

Recently, further understanding of high-rate upflow anaerobic sludge blanket reactors (UASBs) has identified benefit can be delivered through inclusion of an anaerobic pre-treatment stage, in order to decrease solids loading onto the UASB and provided a more acidified substrate (Elmitwalli et al., 1999; Van Haandel et al., 2006). This has led to the development of two-stage high-rate anaerobic reactors, where downstream

UASBs have been preceeded by septic tanks (Luostarinen and Rintala, 2005), anaerobic filters (Sawajneh et al., 2010), and lower-rate UASBs (Sayed and Fergala, 1995; Halalsheh et al., 2005). Whilst it has been identified that, especially at low temperatures, two-stage anaerobic designs are essential for both maximising solids retention and degradation in the first stage, and providing preferential substrate to the second stage (Lettinga et al., 2001; Van Haandel et al., 2006), two-stage designs have not been applied to low-rate technologies to date.

Accordingly, the current study reports on the operation of a pilot scale staged anaerobic pond (SAP) over four HRTs, decreasing from 2.3 days to 0.5 days, to assess the potential for two-stage passive anaerobic treatment at higher loading rates than traditionally applied. The specific objectives of the study were:

1. Compare the performance of a staged AP to a single control AP over four HRT to determine differences in key indicators: hydrodynamic efficiency and flow characteristics; removal efficiency, specifically of carbon fractions; sludge accumulation and where it is retained; biogas production quantity and quality
2. Identify the effect of decreasing HRT on the APs for the above indicators, to determine optimal loading rates for APs at low temperature and its impact on AP operation for effluent quality, sludge management and energy generation

5.2 Materials and Methods

5.2.1 Experimental reactor design

The reactors were constructed of 12 mm uPVC sheeting and sealed with PVC hot welding. The internal dimensions were 1.5 m x 0.5 m x 0.25 m, giving hydraulic volumes of 188 L. A 3:1 Length:Width ratio was used in accordance with recommended AP design (Mara and Pearson, 1998). The SAP was created by connecting two single stage ponds in series, with a horizontally baffled anaerobic pond (HBAP) located upstream of a vertically baffled anaerobic pond (VBAP). The HBAP contained two baffles, located at L/3 and 2L/3 along the reactor length, which extended the entire height of the reactor and 85 % of the reactor width (Peña et al., 2003). The VBAP contained four baffles located at L/5, 2L/5, 3L/5 and 4L/5, which extended the entire width of the reactor and 80 % of its height, alternating between sitting on the base of the pond and standing against the lid (Figure 5-1). A control pond (CAP) was constructed with the same specifications as the HBAP. The reactors were initially seeded with 7 % by volume anaerobic sludge (volatile solids, VS = 36 g L^{-1}) from a previous study (Chapter 3), filled with crude wastewater from the Cranfield University sewage treatment works and left in batch for one day. The reactors were operated for three months at each of four HRTs, with a 2.3 d HRT applied at start up, then subsequent HRTs of 1.5, 1.0, and 0.5 d.

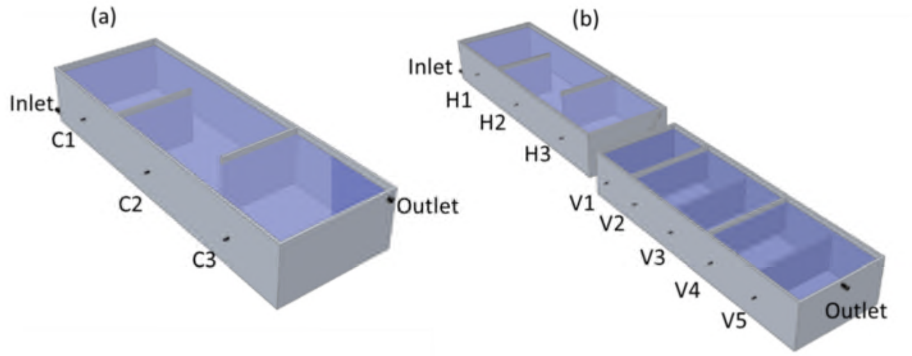


Figure 5-1 Layouts of the ponds used in the study

5.2.2 Analytical methods

Influent and effluent were analysed three times a week in duplicate, whilst liquid samples were also collected and analysed once a month from side ports in each of the chambers created by the baffles (Figure 5-1). Ambient and liquid temperatures were recorded at the time of sampling using a digital probe thermometer, with a sensitivity of ±0.05 ^{o}C. Samples were analysed for BOD_5, COD, TSS and VSS according to standard methods (APHA, 1998). Soluble COD (sCOD), particulate COD fraction (pCOD), and volatile fatty acids (VFA) were measured according to previously described methods (Chapter 4). Biogas was captured in gas-tight sampling bags and analysed for total volume and methane content. Gas volume was measured using a displacement method adapted from Mshandete et al. (2005), whilst methane content was measured with a Servomex 1440 gas analyser (Crowborough, UK). Sludge depth was measured at the end of each loading rate using a perspex tube graduated at 1 mm intervals. To enhance spatial resolution, a grid of 0.1 m x 0.1 m was used. ANOVA tests were performed on all data sets to determine statistical significance to 95 % confidence. The student t-test or Welch's t-test was applied to parametric data sets with equal or non-equal variances, respectively. Non-parametric data were examined for equal medians using the Mann-Whitney U test. Tracer studies were performed with Lithium (Li^+). A LiCl (>99 % reagent grade) solution of 306 g L^{-1} was prepared with de-ionised water, and a pulse signal was introduced to the influent. Effluent samples were taken at intervals of 5 % of HRT over a period equivalent to 3 HRT. Control samples were taken prior to dosing to analyse for background Li^+. Effluent Lithium concentrations were determined by atomic emission spectroscopy (Perkin Elmer, model AAnalyst 800, using an air-acetylene flame method at 670.80 nm) with a minimum detection limit of 0.05 mg L^{-1}.

Specific methanogenic activity tests were carried out on samples taken from each chamber of both ponds at the end of the study, according to previously described methods (Chapter 4). Quantitative real-time polymerase chain reaction (qPCR) was conducted on three methanogenic orders, *Methanomicrobiales, Methanobacteriales and Methanosarcinales*, and two families of the *Methanosarcinales* order, *Methanosaetaceae* and *Methanosarcinaceae*. Total DNA was extracted using a Maxwell automated nucleic acid and protein extraction system (Promega Corporation). Copy numbers of 16S rDNA genes were quantified with real-time qPCR assays using a LightCycler 480 (Roche Diagnostics). The qPCR cycling conditions and specific primer sets used were those described by Yu et al. (2005), with standard curves generated using the method described by Yu et al. (2006). Further details on the principles of qPR can be found in Appendix A.

5.2.3 CFD modelling

Three dimensional single phase CFD simulations were performed using the commercial software FLUENT v14.0.0 (ANSYS). Firstly, computational models, or geometries, were drawn to represent each of the reactors trialled in the experiments. These geometries were then modelled for steady state water flow using momentum equations and finite element analysis, to create velocity profiles of the flow within the reactors (Appendix A). Once the steady state flow had been calculated, a virtual tracer study was conducted on the computational model, in order to compare the computed tracer to the experimental tracer. Geometries for the four reactors designs were drawn in AutoCAD 2007 (AutoDesk, Inc.) and meshed using ICEM CFD (ANSYS). Meshes contained total elements of 2,234,971 for the CAP and 3,012,830 for the SAP. The laminar flow model was used as the Reynolds number of the ponds was calculated as less than 6. The fluid in the ponds was assumed to be incompressible and exhibiting Newtonian fluid properties of water with a density of 998.2 kg m^{-3} and a dynamic viscosity of 1.003×10^{-3} kg m^{-1} s^{-1}. The inflow boundary condition was defined as a mass-flow-inlet with a constant mass flow, according the volumetric loading of the HRT applied. The outflow boundary condition was defined as a pressure-outlet with a gauge pressure of 0 pascal. The tracer RTD analysis was performed by imposing a transient simulation of the tracer as a scalar on the velocity and turbulent fields obtained from the flow simulation using the method of Alvarado et al. (2012).

5.3 Results

5.3.1 Hydrodynamic comparison using experimental tracer studies and computational fluid dynamics

In both the experimental tracer studies and CFD simulations, lower dead space volumes were found in the SAP compared to the CAP at all HRTs. The differences between AP configurations became more pronounced with each step decrease in HRT (Table 5-1), indicating greater utilisation of pond volume in the SAP, especially at shorter HRT. Lower dead space volumes in the experimental studies compared to the CFD was influenced by the presence of the sludge in the tracer studies, with $HRT_a > HRT_t$ in the SAP at 0.5 d HRT demonstrating the interference of the high sludge volumes with the tracer. Interestingly, whilst higher S quotients in the SAP suggests a higher degree of short circuiting than in the CAP in the experimental tracers, this is only true at 2.3 d HRT in the CFD simulations, as higher CFD S values are produced in the CAP at the three shorter HRTs. Dispersion numbers were lower in the SAP in all

CFD studies, whilst N values were higher at all but 2.3 d HRT, with the differences in both parameters increasing with decreasing HRT, although no clear trends can be discerned from the experimental data. Whilst environmental impacts may have impacted the experimental results, the CFD trends suggest the overall flow characteristics of the SAP tending towards plug flow to a greater extent than the CAP.

Analysis of the local velocity profiles obtained in the CFD simulations suggest that the improved hydrodynamic profiles in the SAP can be attributed to the degree of recirculation that occurs between baffles. This is generated through the flow being forced back into the chamber by the small aperture created by the baffles, consequently utilising more of the chamber and thus reducing short circuiting (Figure 5-2). The recirculation is most pronounced in the front chamber of the ponds, where velocities are at their highest due to the jetting effect caused by the inlet. As this initial velocity is dissipated, the local velocities decrease through the second chamber such that the recirculation effect is lessened, creating preferential flow patterns which is most evident in the CAP. At higher velocities, such as in the first stage of the SAP, velocities in the second chamber are high enough to cause noticeable recirculation (Figure 5-2), improving the mixing profile within each chamber created, thereby reducing dead space whilst also creating an overall plug flow effect through the reactor.

Table 5-1 Hydrodynamic data calculated for a control anaerobic pond (CAP) and staged anaerobic pond (SAP), over four hydraulic loading rates. Data is shown for experimental data collected from tracer studies and for computational fluid dynamics (CFD) simulations

	Experimental data											
	2.3 d HRT			1.5 d HRT			1.0 d HRT			0.5 d HRT		
	CAP	SAP	Diff	CAP	SAP	Diff	CAP	SAP	Diff	CAP	SAP	Diff
HRT_a (d)	1.85	2.07	0.22	1.03	1.33	0.30	0.72	0.97	0.25	0.46	0.80	0.34
HRT_a/HRT_t (%)	80	90	10	69	89	20	72	97	25	92	160	68
Short circuiting quotient, S	0.40	0.47	0.07	0.25	0.36	0.11	0.16	0.34	0.18	0.46	0.79	0.33
Dead space volume (%)	20	10	-10	31	11	-20	28	3	-25	8	-60	-68
Variance, σ^2 ($days^2$)	0.90	0.76	-0.14	0.68	0.90	0.22	0.33	0.34	0.01	0.25	0.13	-0.12
Dispersion number, δ	0.16	0.10	-0.06	0.32	0.34	0.02	0.63	0.24	-0.39	0.19	0.12	-0.07
Tanks in series, N	5.96	7.06	1.10	4.82	2.76	-2.06	3.05	2.96	-0.09	3.93	1.89	-2.04
Tracer recovered (%)	100	100	0	48	55	7	35	52	17	40	88	48
Sludge volume (% of reactor)	13	14	1	11	12	1	18	19	1	46	38	-8
	CFD simulations											
	2.3 d HRT			1.5 d HRT			1.0 d HRT			0.5 d HRT		
	CAP	SAP	Diff	CAP	SAP	Diff	CAP	SAP	Diff	CAP	SAP	Diff
HRT_a (d)	1.33	1.48	0.15	0.89	0.98	0.09	0.63	0.69	0.06	0.34	0.36	0.02
HRT_a/HRT_t (%)	57	64	7	59	66	7	63	69	6	68	72	4
Short circuiting quotient, S	0.29	0.36	0.07	0.39	0.36	-0.03	0.41	0.38	-0.03	0.63	0.38	-0.25
Dead space volume (%)	43	36	-7	41	34	-7	37	31	-6	32	28	-4
Variance, σ^2 ($days^2$)	0.62	0.68	0.06	0.48	0.47	-0.01	0.37	0.36	-0.01	0.21	0.18	-0.03
Dispersion number, δ	0.13	0.12	-0.01	0.18	0.13	-0.05	0.23	0.17	-0.06	0.26	0.15	-0.11
Tanks in series, N	13.76	11.43	-2.33	9.77	10.03	0.26	7.31	7.46	0.15	5.67	7.75	2.08
Tracer recovered (%)	91	94	3	96	98	2	98	100	2	89	66	-23
Maximum velocity v_{max} (m s^{-1})	$1.47x10^{-2}$	$1.04x10^{-1}$	$8.90x10^{-2}$	$2.39x10^{-2}$	$1.55x10^{-1}$	$1.3x10^{-1}$	$3.76x10^{-2}$	$2.27x10^{-1}$	$1.9x10^{-1}$	$7.51x10^{-2}$	$4.00x10^{-1}$	$3.2x10^{-1}$
Minimum velocity v_{min} (m s^{-1})	$1.21x10^{-9}$	$7.22x10^{-8}$	$7.10x10^{-8}$	$7.22x10^{-8}$	$5.40x10^{-8}$	$-1.82x10^{-8}$	$1.74x10^{-9}$	$1.02x10^{-7}$	$1.00x10^{-7}$	$3.67x10^{-9}$	$1.71x10^{-5}$	$1.71x10^{-5}$

CAP –Control anaerobic pond; SAP – Staged anaerobic pond; Diff – Difference between CAP and SAP; HRT – Hydraulic retention time; HRT_a – actual (measured) HRT; HRT_t – theoretical HRT

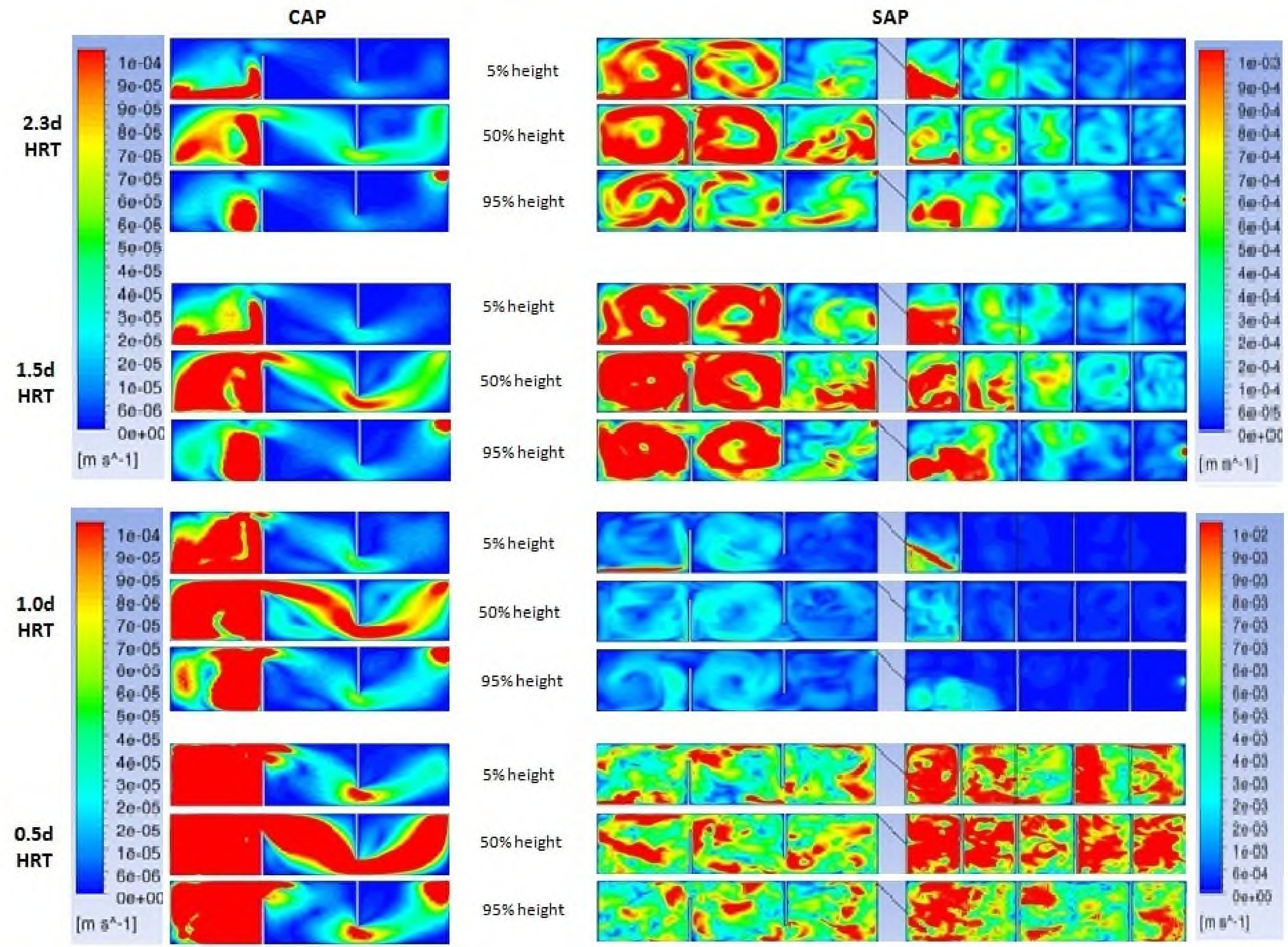


Figure 5-2 CFD generated velocity profiles for the control anaerobic pond (CAP), and the staged anaerobic pond (SAP)

5.3.2 Removal efficiencies over four HRTs from the staged and control anaerobic ponds

No significant difference between reactors was observed in relation to TSS or pCOD removal over all HRTs to a 95 % confidence level. In contrast, mean sCOD removals were statistically different, and were lowest at 1.5 d HRT in both reactors at -40 % and -44 % for the CAP and SAP, respectively, with the highest removal observed at 0.5 d HRT, CAP -5 % and SAP 2 % (Figure 5-3).

However, these removal efficiencies correlate with the temperature profile in the ponds, with the highest mean effluent temperatures recorded at 1.5 d HRT (CAP 17.1 ^{o}C; SAP 17.0 ^{o}C) and the coldest temperatures observed during the 0.5 d HRT period (CAP 9.3 ^{o}C; SAP 9.1 ^{o}C), and therefore both temperature and HRT may have influenced sCOD removal. Removal efficiencies of VFA were similar to the sCOD trend, with the largest addition of VFA to the effluent occurring at 1.5 d HRT whilst removal efficiency increased in the shorter HRT periods. The VFA removal efficiency profile is likely to be significantly impacted by the rate of hydrolysis – the conversion of organic carbon to VFA, which is strongly temperature dependent (Pavlostathis, 1991).

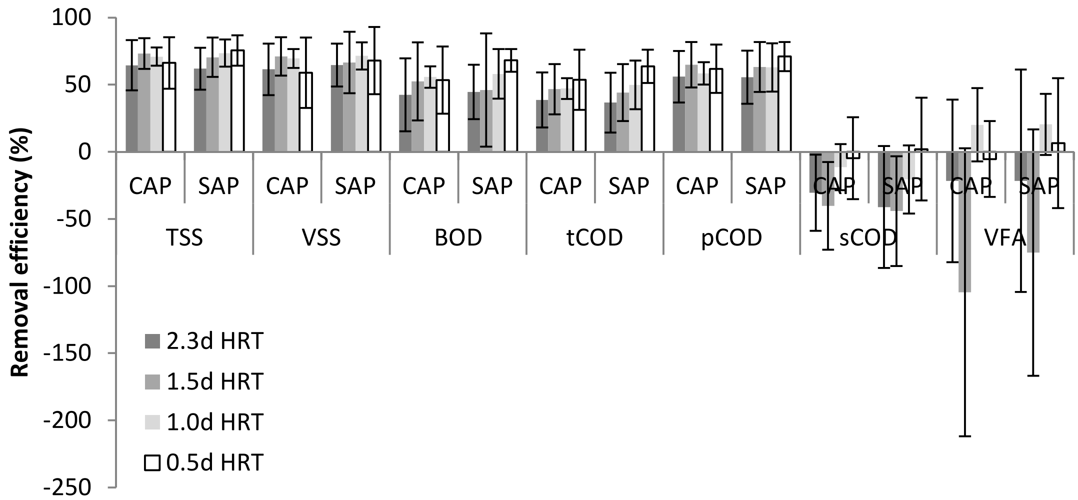


Figure 5-3 Removal efficiencies from the pilot scale trials on a horizontally baffled anaerobic pond as a control (CAP) and a staged anaerobic pond (SAP)

5.3.3 Solids removal and sludge accumulation

Whilst no significant difference was found between removal efficiency of TSS over the study, a linear relationship was found between mass removal and loading rate for both ponds at each HRT trialled suggesting the both ponds are being operated below the maximum limit of solids loading rate (Figure 5-4). Variations in loading for a set HRT occurred due to variable TSS concentrations in the influent, whilst effluent concentrations were consistent in both the CAP and SAP. To illustrate, the range of TSS influent concentrations at 1.5 d HRT was 154 to 818 mg L^{-1}, whilst the CAP effluent range was 39 to 108 mg L^{-1} and 44 to 162 mg L^{-1} in the SAP. Whilst this relationship was found at each HRT applied, the relationship diminished with decreasing HRT. In the SAP the highest influent concentration at the 2.3 d HRT, 637 mg L^{-1}, translated to a load of 0.28 kg m^{-3} d^{-1}, with removal of 0.52 g m^{-3} d^{-1}. The lowest influent concentration at 0.5 d HRT, 187 mg L^{-1}, produced a similar load of 0.37 kg m^{-3} d^{-1}, but with lower removal at 0.12 g m^{-3} d^{-1}. Comparable loading rates at 1.5 d HRT, 0.32 kg m^{-3} d^{-1}, concentration = 478 mg L^{-1}, and 1.0 d HRT, 0.32 kg m^{-3} d^{-1}, concentration = 318 mg L^{-1}, produced removal rates of 0.40 kg m^{-3} d^{-1} and 0.23 kg m^{-3} d^{-1}, respectively. The linear relationship was weakest at the lowest HRT, 0.5 d, where more variable removal rates, particularly in the CAP, may suggest process instability.

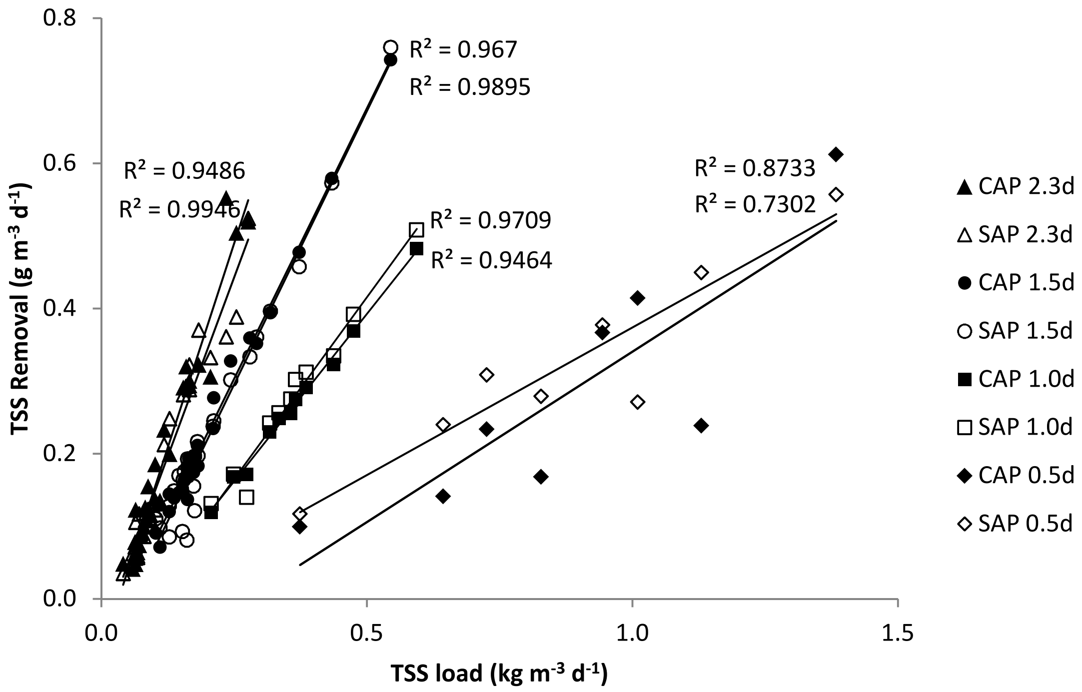


Figure 5-4 TSS removal from the CAP and SAP over the four HRTs applied during the study. Removal was found to vary with influent TSS over each HRT, although this relationship diminished with decreasing HRT

Solids accumulation rate within the ponds was found to be more dependent on temperature than loading. In both ponds, per capita normalised sludge accumulation rates were comparable at three of the HRTs studied. In the CAP, accumulation rates over the 2.3, 1.0 and 0.5 d HRT periods were 0.04 m^3 PE^{-1} y^{-1} (mean effluent temperature, T_{eff} = 10.5 ^oC), 0.04 m^3 PE^{-1} y^{-1} (T_{eff} = 13.9 ^oC) and 0.06 m^3 PE^{-1} y^{-1} (T_{eff} = 9.3 ^oC), respectively. In comparison, accumulation rates in the SAP were 0.06 m^3 PE^{-1} y^{-1} (T_{eff} = 10.5 ^oC), 0.04 m^3 PE^{-1} y^{-1} (T_{eff} = 13.7 ^oC) and 0.06 m^3 PE^{-1} y^{-1} (T_{eff} = 9.1 ^oC) were calculated for the same periods, respectively. However, during the warmest HRT period, 1.5 d, a reduction in total sludge volume was recorded in both ponds, with an

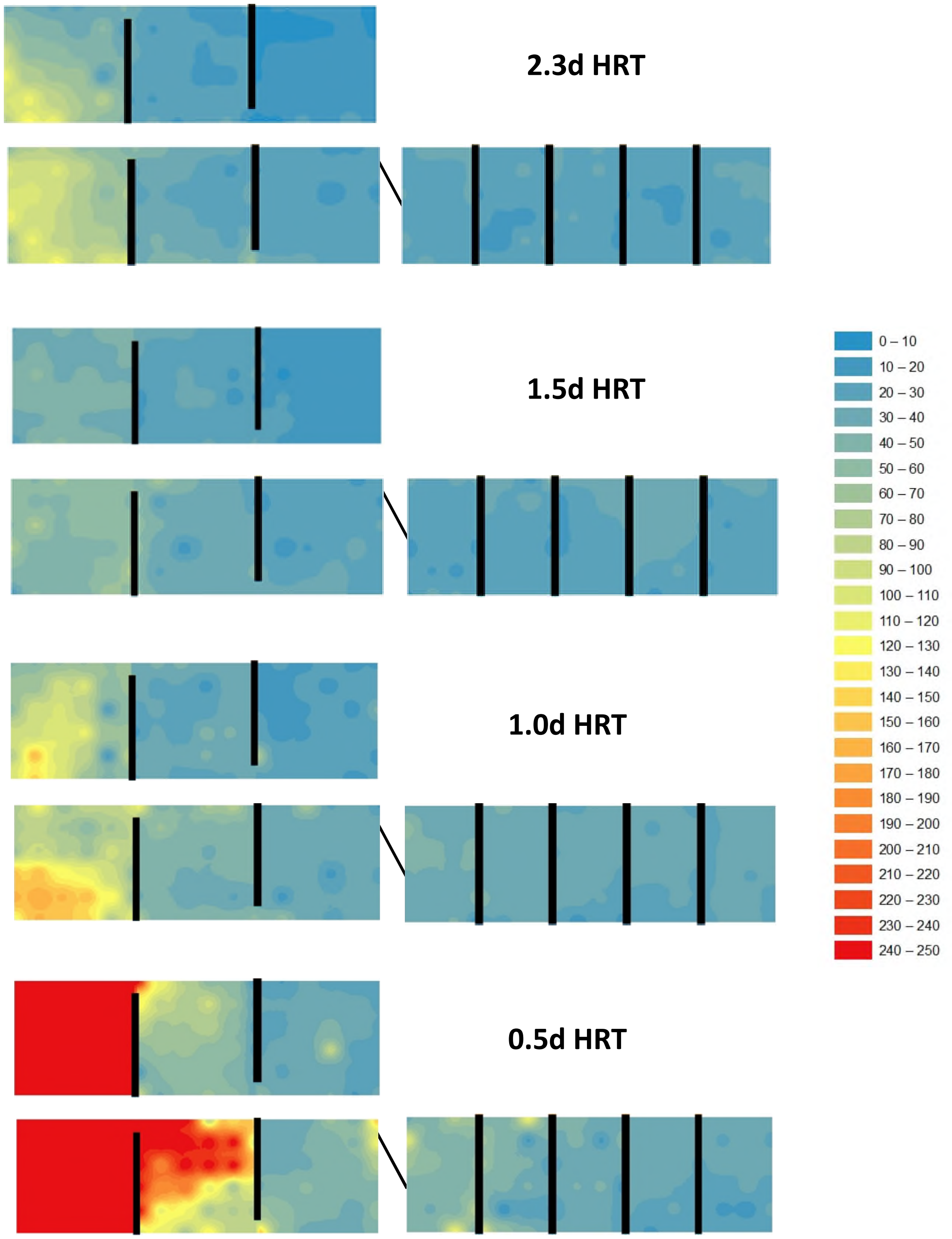


Figure 5-5 Sludge accumulation maps in the CAP and SAP at the end of each of the four hydraulic retention times applied

accumulation rate of -0.02 m^3 PE^{-1} y^{-1} for both ponds (CAP T_{eff} = 17.1 ^oC, SAP T_{eff} = 17.0 ^oC). Whilst the normalised accumulation rates were comparable across the decreasing HRT periods at low temperature, the higher loadings applied relate to higher absolute sludge volumes within the ponds. To illustrate, in the SAP the accumulation rate of 0.06 m^3 PE^{-1} y^{-1} during the 2.3 d HRT period related to an accumulated sludge volume of 16.11 L, or 3 % of total pond volume, whilst the 0.04 m^3 PE^{-1} y^{-1} accumulation rate over the 0.5 d HRT period related to an accumulated sludge volume of 73.11 L, or 14 % of total pond volume. Solids were mostly deposited in the front chamber of each pond (Figure 5-5), with 63, 49, 30 and 73 % of total CAP sludge volume found in this chamber after the 2.3, 1.5, 1.0 and 0.5 d HRT periods, respectively, whilst this chamber comprised only 33 % of total pond volume. In the SAP, sludge accumulation in the front chamber contained 39, 28, 37, and 43 % of total sludge volume, despite this chamber comprising only 17 % of total pond volume.

5.3.4 Biogas methane production and specific methanogenic activity of sludge

Rapid start up of methane biogas production was observed in both ponds, with mean flow normalised production of 3.86 LCH_4 m^{-3} wastewater treated (WWT) in the CAP and 5.40 LCH_4 m^{-3}WWT in the SAP during the first operational period, at 2.3 d HRT. The highest mean biogas production occurred during the second period, 1.5 d HRT, with mean flow normalised production of 5.40 LCH_4 m^{-3}WWT in the CAP and 8.82 LCH_4 m^{-3}WWT in the SAP, which coincided with the highest mean effluent temperatures (Figure 5-6).

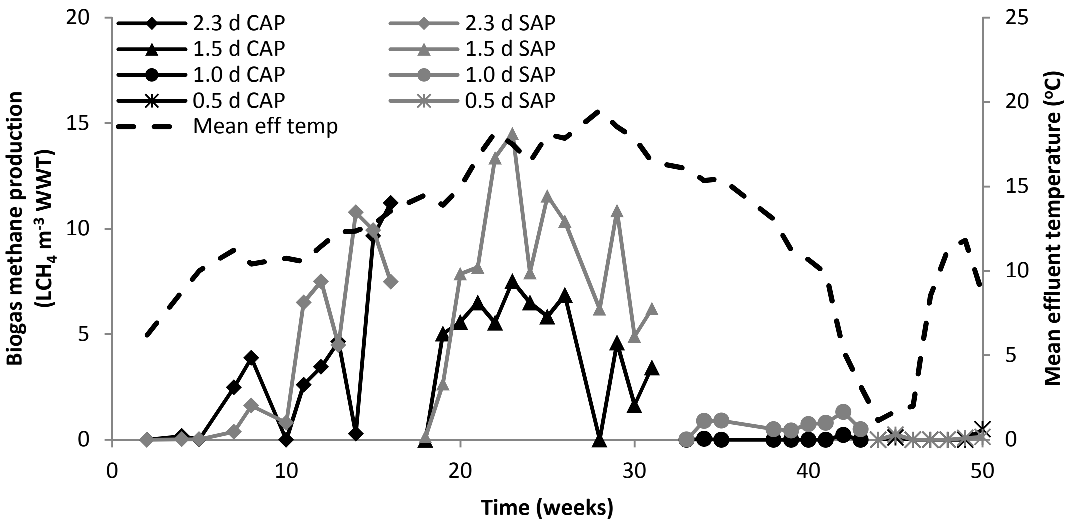


Figure 5-6 Mean flow-normalised biogas methane production in the control (CAP) and staged (SAP) anaerobic ponds.

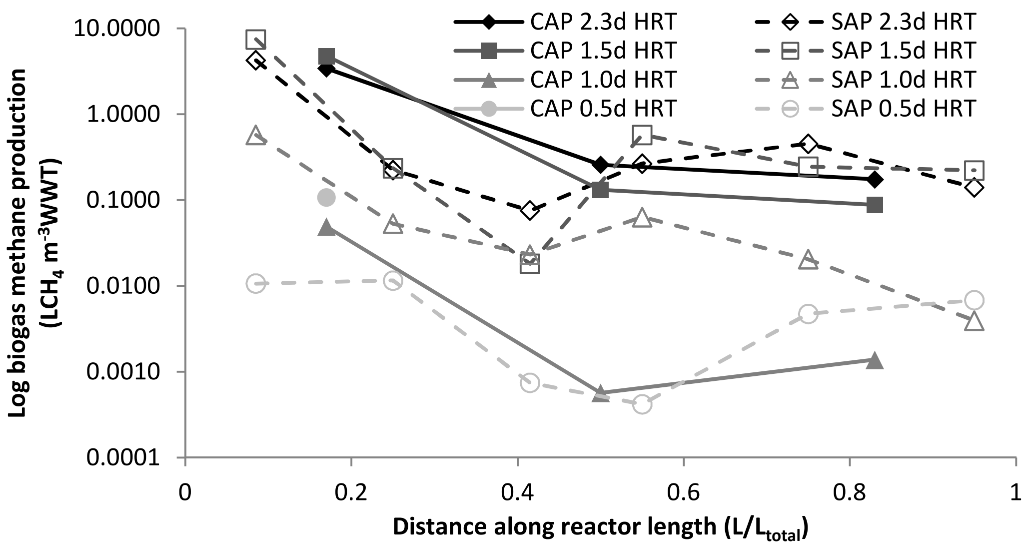
With decreases in both temperature and HRT, large reductions in biogas productions were observed for the final two operational periods, with mean production rates of 0.05 LCH_4 m^{-3}WWT and 0.11 LCH_4 m^{-3}WWT in the CAP and 0.74 LCH_4 m^{-3}WWT and 0.08 LCH_4 m^{-3}WWT in the SAP for the 1.0 and 0.5 d HRT periods, respectively. No statistical difference was observed in biogas production between the two reactors at 2.3 d HRT, nor at 0.5 HRT due to low production rates in both reactors. However, at 1.5 and 1.0 d HRT, biogas production in the SAP was significantly higher than the CAP to a 95 % confidence level.

The highest measured production rate was in the chamber closest to the inlet for both reactors at all four loading rates (Figure 5-7), with 95 and 84 % of total biogas CH_4 recorded in this chamber for the CAP over the entire study period for the CAP and SAP, respectively. In the CAP, production rates decreased in subsequent chambers at 2.5 and 1.5 d HRT, although at 1.0 d an increase was evident in the final chamber, suggesting production at the outlet may have been increasing respective to the centre of the reactor. Due to the low temperature during the 0.5 d HRT, no biogas was recorded in either chamber 2 or 3 for this final loading rate. In the SAP, biogas production decreased throughout the first stage reactor, but increased from the last chamber of the first phase to the first chamber of the second phase at 2.3, 1.5 and 1.0

d HRTs. To illustrate, mean biogas methane production rates at the outlet of the first stage were 0.08, 0.03 and 0.06 LCH_4 m^{-3} WWT *cf.* 0.26, 1.05 and 0.17 LCH_4 m^{-3}WWT at the inlet of the second stage at 2.3, 1.5, and 1.0 d HRT, respectively. This may be induced by both the jetting effect of the connection pipe between the two stages creating high mixing at the inlet of the second stage, and through a change in microbiological community found in the reactors. Specific methanogenic activity tests conducted on sludge at the end of the study period show activity rates were lower at the inlet of both reactors than the subsequent chambers (Figure 5-7), which is consistent with previous findings (Chapter 4). Hydrogenotrophic methanogenic activity was found to be over two orders of magnitude greater than aceticlastic activity, with mean hydrogen specific SMA of 1,001 $mgCH_4$ $gVSS^{-1}$ d^{-1} and 1,489 $mgCH_4$ $gVSS^{-1}$ d^{-1} recorded in the CAP and SAP, respectively, cf. 0.27 $mgCH_4$ $gVSS^{-1}$ d^{-1} and 1.36 $mgCH_4$ $gVSS^{-1}$ d^{-1} for acetate specific SMA. Interestingly, acetate specific SMA was over two orders of magnitude higher in the second phase of the SAP than the first, with mean acetate specific SMA of 0.01 $mgCH_4$ $gVSS^{-1}$ d^{-1} in the first stage *cf.* 2.71 $mgCH_4$ $gVSS^{-1}$ d^{-1} in the second stage.

Whilst this study has focused on the methanogenic communities at the final stages of the anaerobic digestion process, the finding that hydrogenotrophic methanogenesis is dominant suggests the earlier stages of the digestion process should also be investigated. During hydrolysis, acidification, and acetogenesis, complex organic compounds are broken down into intermediate products in the anaerobic digestion process, of which hydrogen and acetate are the two principle substrates generated that are required for the subsequent methanogenesis to occur (Appendix A). Acetate, $C_2H_3O_2^-$, contains hydrogen, and therefore during acidogenesis and acetogenesis competition for hydrogen between hydrogen forming bacteria and acetate forming bacteria can occur. Furthermore, hydrogen can have a toxic effect on some acetate forming bacteria, and therefore the balance of acetogenic activity is important for ensuring the stability of the overall digestion process (Shah et al., 2014). The increased levels of acetate evidenced in the VFA concentrations in this study, coupled with the dominance of hydrogenotrophic methanogens, suggest an imbalance in the microbial communities present for the anaerobic process. This may be evidence of a community still adapting to the environmental conditions, and further understanding of the balance of acetate and hydrogen forming bacteria in the reactors is required to understand the entire digestion process occurring.

(a)



(b)

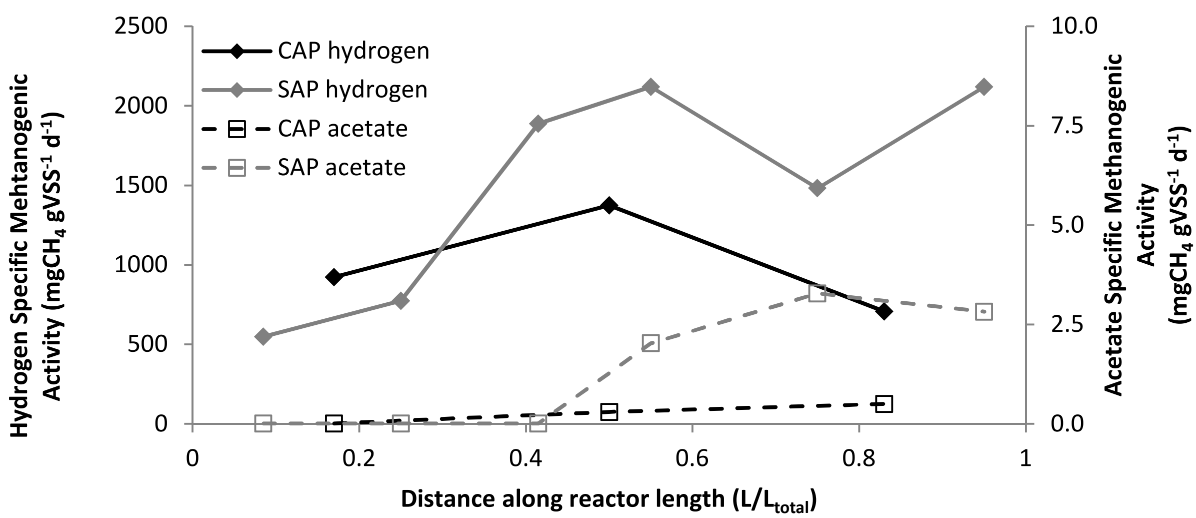


Figure 5-7 Mean flow-normalised biogas methane production (a) specific methanogenic activity (b) from sludge samples along the length of the control pond (CAP) and two-stage pond (SAP) at the end of the study

5.3.5 Microbial community profiling of methanogenic orders and families in the sludge

Microbial community profiling of methanogenic archea in sludge taken at the end of the study found the hydrogenotrophic order *Methanomicrobiales* dominant, producing a mean of 1.87×10^{7} copies from the qPCR process *cf.* 1.26×10^{6} copies of the aceticlastic *Methanosarcinales* order in the CAP, and 2.51×10^{7} copies *cf.* 4.53×10^{6} copies in the SAP (Figure 5-8). In addition, another hydrogenotrophic order, *Methanobacteriales*, was also present with mean 3.25×10^{5} copies in the CAP and 2.44×10^{5} copies in the SAP, increasing the dominance of hydrogen utilisers. The relative presence of these orders supports the SMA findings of hydrogen pathways dominating the anaerobic digestion process in both reactors. In the SAP, an increase in the *Methanosarcinales* order from mean 1.27×10^{6} copies in the first stage to 6.48×10^{6} copies in the second stage, also reflects the increase in acetate specific SMA found the in the second stage at the end of the study. Within the *Methanosarcinales* order, the *Methanosaetaceae* family was found to dominate the *Methanosarcinaceae* family in both reactors, with mean copy numbers 2.61×10^{6} *Methanosaetaceae cf.* 3.86×10^{4} *Methanosarcinaceae* in the CAP and 6.53×10^{6} *Methanosaetaceae cf.* 9.37×10^{4} *Methanosarcinaceae* in SAP (Figure 5-8). Interestingly, in the SAP the Methanosarcinales families were found in closest relative abundance in the first chamber, with 1.15×10^{6} copies of *Methanosaetaceae cf.* 4.85×10^{5} *Methanosarcinaceae*, with Methanosaetaceae dominating further along the reactor, particularly in the second stage, with mean 9.28×10^{6} copies of *Methanosaetaceae cf.* 3.26×10^{4} *Methanosarcinaceae*. The dominance of Methanosarcinaceae within the Methansarcinales order has been found to be consistent with low acetate concentrations and indicative of increased acetate oxidation, leading to hydrogenotrophic methanogenesis, rather than aceticlastic pathways (Karakashev et al., 2006).

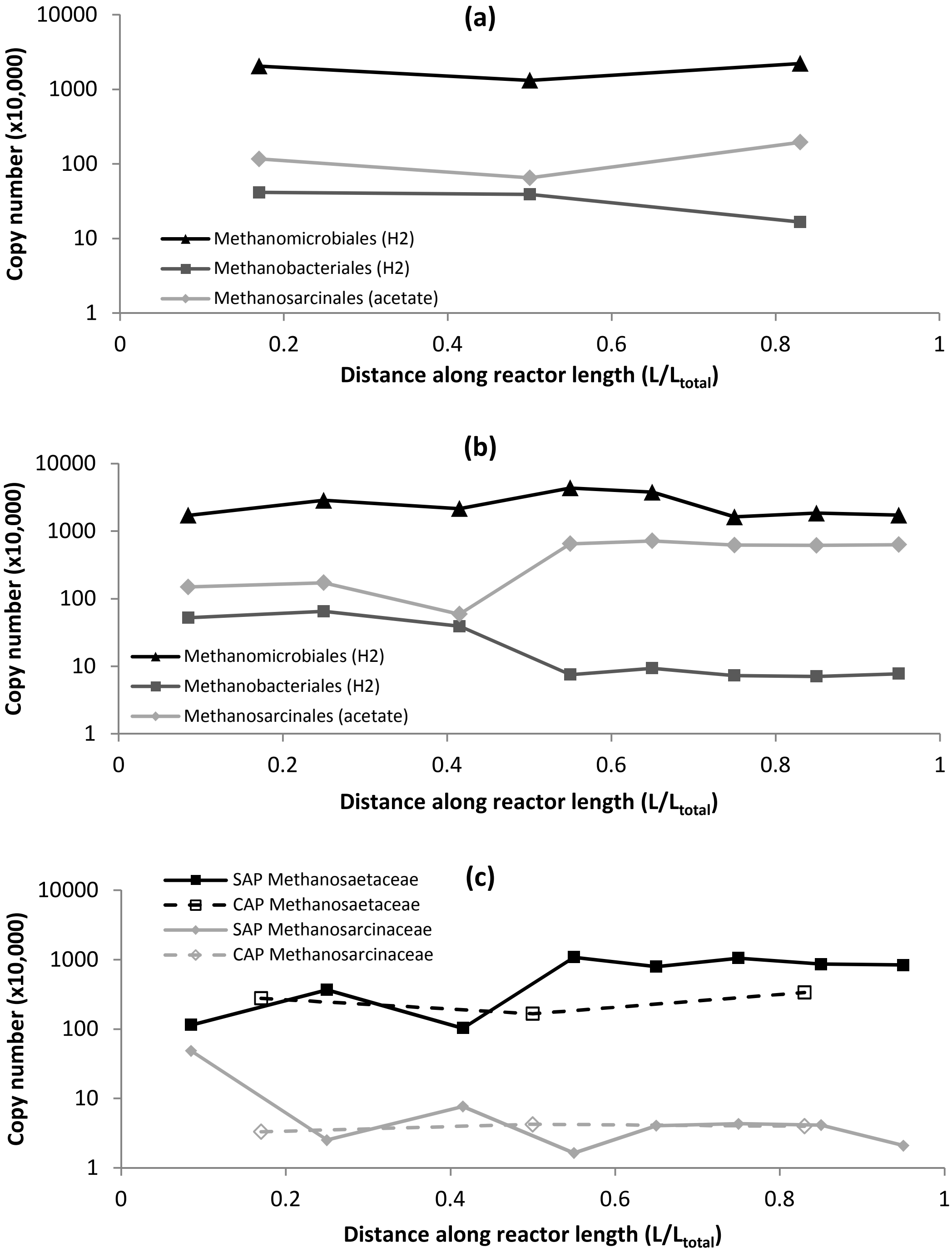


Figure 5-8 Microbial community qPCR data for three orders of methanogenic archea, two hydrogenotrophic and one acetoclastic, in the (a) CAP and (b) the SAP, and (c) two families of the methanosarcinales order in the CAP and SAP.

5.4 Discussion

Comparison of the proposed staged AP (SAP) to conventional ponds (CAP) revealed SAPs to enhance both biogas production and overall hydrodynamic efficiency. The latter was seen in terms of less short circuiting with associated less dead space. The enhancements where observed across all HRTs with velocity profiles demonstrating the increase recirculation between baffles (Peña et al., 2003) leading to greater utilisation of the pond volume. The CFD simulations were valuable in identifying the flow patterns within the ponds, and whilst more powerful modelling may be available in the future for more accurate representation of dynamic processes such as solids settling and biogas bubbling, it currently still presents a useful tool for complementing experimental results, and for preliminary selection of designs where extensive pilot trialling is not possible (chapter 4). Solids accumulations reduced the clarity of the impacts congruent with previous studies on unbaffled ponds (Peña et al., 2000; Alvarado et al., 2012). Further, vertically baffled systems can be particularly susceptible to channelling (Grobicki and Stuckey, 1992), as the flow is forced through the sludge layer at every 'hanging' baffle and thus optimisation of the baffling arrangement will be critical and as such is one of the key areas for further investigation. The differences in hydrodynamics did not manifest in terms of bulk removal which remained statistically similar for both ponds. This extended to soluble COD removal where improvements in removal were not observed to a statistically significant level. Improvements in gas production, however, were observed in the SAP, with increased SMA and an aceticlastic methanogenic community measured in the second stage suggests the spatial distribution of the anaerobic digestion was starting to occur (Barber and Stuckey, 1999; Paing et al., 2000). Furthermore, many of the advantages seen in the SAP over the CAP were more pronounced at the shorter HRTs indicating that the proposed design can provides a route to using APs with footprints more attractive to potential users.

Results from the reductions in HRT suggest APs can tolerate higher loadings than currently applied. Decreasing HRT in unbaffled ponds is known to increase short circuiting, however the results of the CFD simulations suggest that these impacts can be lessened in baffled system, and advantages can also be gained in reduce hydraulic dead space through the recirculation effect between baffles. As temperature profiles changed over the course of the study, the influence of temperature must be considered when comparing the HRTs applied. The removal of solids has been reported to be independent of operating temperature (Picot et al., 2003; Papadopoulos et al., 2003), and in this study a clearer relationship was found with loading rate. Consistent effluent TSS profiles down to 1.0 d HRT in both ponds reinforced the ability of the APs to handle shock loadings, whilst also confirming that solids loading rates are unlikely to be a restricting factor in AP design (Chapter 4). However, biological activity was clearly strongly associated with temperature (Toprak, 1995; Picot et al., 2003; Parissopoulos et al., 2003). Soluble carbon removal efficiency was linked with temperature rather than loading rate, and may be attributed to a reduction in soluble carbon generated in the digestion process (Chapter 3) rather than improvement of soluble degradation due to biological establishment (Paing et al., 2000; Picot et al., 2003). Sludge accumulation rates were also temperature dependent, with volume reduction occurring above 17 ^{o}C as suggested by Papadopoulos et al. (2003). The sludge reduction at warmer temperatures was linked to the highest biogas production rates, and supports previous evidence that APs can store particulate carbon in winter periods to be subsequently degraded in summer (Safley Jr. and Westerman, 1989; Papadopoulos et al., 2003; Picot et al., 2003). Therefore, in order to accurately estimate the effect of HRT on sludge accumulation and biogas production, studies must be conducted over an annual cycle. Furthermore, sludge accumulation rates have been found to lower, and biogas production rates increase, with extended AP operation (Paing et al., 2000; Picot et al., 2005b), and the minimum temperature at which methanogenesis occurs has been found to decrease with AP age as biomass acclimatises (Heubeck and Craggs, 2010). Therefore, it can be posited that these characteristics would improve from the current study over time.

Shorter HRTs can mitigate the largest single problem with AP uptake in reducing the land requirement, and therefore the cost (Agunwamba, 2001; Alexiou and Mara, 2003), and the results from this study suggest that shorter HRTs than currently recommended are feasible. The severe reduction in gas production at 0.5 d HRT is likely to be a cause of the temperature but also the loading, and whilst the sludge accumulation rate per capita was comparable to longer HRTs, the volume of sludge produced at this HRT would likely reduce the advantages APs can bring in reduced sludge handling (Chapter 3). To illustrate, whilst sludge accumulation rates in the SAP were 0.06 m^3 PE^{-1} y^{-1} at both 2.3 and 0.5 d HRT, desludging at 50 % volume would lead to a desludge frequency of 3.8 years at 2.3 d HRT, but 0.4 years at 0.5 d HRT. Therefore, extended trials of APs at 1.0 and 1.5 d HRTs are recommended, which would reduce AP volume by two to three times the current recommendations.

5.5 Conclusions

Trials on a two stage anaerobic pond design found advantages over a single stage control, whilst the opportunity to operate APs at shorter HRTs than currently recommended was also identified. Specific conclusions were:

- Whilst removal efficiencies between a two-stage and single stage AP were not statistically different, superior biogas production from the two stage pond, along with greater hydraulic efficiency, demonstrate the potential of two stage designs to deliver improved performance over a single stage pond
- The potential for shorter HRTs to be applied to APs, even at low temperature, has been demonstrated. This study shows that solids removal is not linked to temperature, and that stable effluent quality can be achieved even at HRTs of 1 day.
- From this study, it is recommended that APs should be trialled further at 1.5 and 1.0 d HRTs. Extended studies have to be carried out to determine how seasonal variations could affect biological activity, particularly with respect to soluble breakdown, sludge accumulation, and biogas production.
- Whilst shorter HRTs can infer advantages not only with process performance but also in capital cost savings through a reduction in land requirement, the potential for increased maintenance requirements, such as desludging frequency, must also be considered when designing an AP.

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6 Implications of the work

6.1 Key observations from the work

Through the research conducted in this thesis, key observations have been made to in relation to establishing effective guidelines for anaerobic pond design to deliver enhanced methane recovery and sludge management on low temperature domestic wastewater. Three themes have been identified as being integral to AP design, in order to address new operational requirements in improved biological breakdown and reduced footprint.

6.1.1 Hydraulic retention time and loading rate

- In the past anaerobic ponds (APs) have been used only for solids removal, and so long hydraulic retention times (HRTs) have been applied in order to capture particulate matter and degrade at a low enough rate that biogas emissions are sufficiently inhibited so as not to cause odour problems (Chapter 2)
- The low loading rates cause inefficient use of the reactor volume, sludge builds up mostly at the inlet, and the lack of mixing means digestion is not promoted (Chapters 2, 3, 4 and 5).
- Covering APs eliminates their biggest operational risk, odour emissions (Chapters 2 and 3), whilst also reducing their biggest design barrier, footprint, through permitting shorter HRTs (Chapter 5)
- APs can function effectively at shorter HRTs at low temperature, as APs are found to be resilient even to extreme low temperatures (Chapter 3), whilst extended solids retention time (SRT) means sludge that cannot degrade at the lowest temperatures can be stored for increased biological activity in warmer seasons (Chapters 3 and 5)

6.1.2 Baffling and staged designs

- The key to reducing HRT in APs is to decouple it from the SRT, which is best done through baffling to minimise solids washout whilst allowing higher liquid velocities (Chapters 4, 5).

- Different baffle configurations can promote different flow regimes, which can be used to achieve different aims – horizontal baffling for sedimentation and solids retention and vertical baffling for mixing and soluble degradation (Chapters 4 and 5).
- Staging the anaerobic process has been found to increase anaerobic activity, by promoting different stages of the digestion process along the length of a reactor (Chapter 5). Baffle designs could increase this further by generating preferential conditions in different areas of the ponds and generating a physical barrier between regions (Chapter 4, 5).

6.1.3 Dynamic processes and operational considerations

- Biological activity is strongly temperature dependent, and therefore seasonal variations need to be considered (Chapters 3 and 5). The building up of sludge in cold temperatures provides a biomass 'store' for increased activity at warmer temperatures, facilitating reduction in sludge volumes and higher biogas production (Chapter 3). However, this will affect the consistency of biogas productions and sludge accumulation, which may affect operational parameters such as the potential for year-round energy generation.
- Shorter HRTs didn't seem to have an effect on per capita sludge accumulation rates, but smaller pond volumes will need more frequent desludging (Chapter 5). For a specific AP design, the required balance between capital costs (footprint), operational costs (desludge) and organic breakdown/biogas production will be important in determining the HRT and sizing the pond (Chapters 3 and 5).

6.2 Incorporating anaerobic ponds into decentralised wastewater treatment

6.2.1 Introduction

The findings of this thesis have outlined the many benefits that APs present for full flow wastewater treatment. Opportunities for low impact sludge management strategies (Chapter 3) and renewable energy production (Chapters 3 and 5) have been highlighted, alongside traditional advantages of low energy demand and operation and maintenance requirements (Chapter 2). The opportunities APs could present to wastewater treatment flowsheets are especially suited to small decentralised treatment work, which pose unique challenges compared to larger centralised facilities. In the UK, treatment works serving < 2,000 PE account for 78 % of treatment works in the UK but only treat 4 % of the wastewater produced (Johnson et al., 2007). These works present the greatest risk of non-compliance with effluent quality requirements (Griffin and Pamplin, 1998), and have a disproportionately high burden on sludge management due to the need to tankering waste solids to centralised anaerobic digestion (AD) facilities (Chapter 3), and the associated infrastructure cost of ensuring suitable site access for these activities.

In order to quantify and examine the merits of incorporating APs into decentralised wastewater treatment, a flowsheet modelling approach has been adopted to assess the relative impacts against existing technologies. The aim of this study is to compare an AP flowsheet to a current standard decentralised flowsheet in order to determine the suitability of APs for incorporation into decentralised wastewater treatment, and identify where potential benefits and barriers may lie. This aim will be achieved through three objectives:

1. Modelling of energy balances for both flowsheets to determine energy requirements, both through on-site and off-site generation and demand
2. Carbon accounting of both flowsheets to assess carbon footprint, including direct impacts through fugitive emissions on site and indirect, through energy requirements and sludge transport

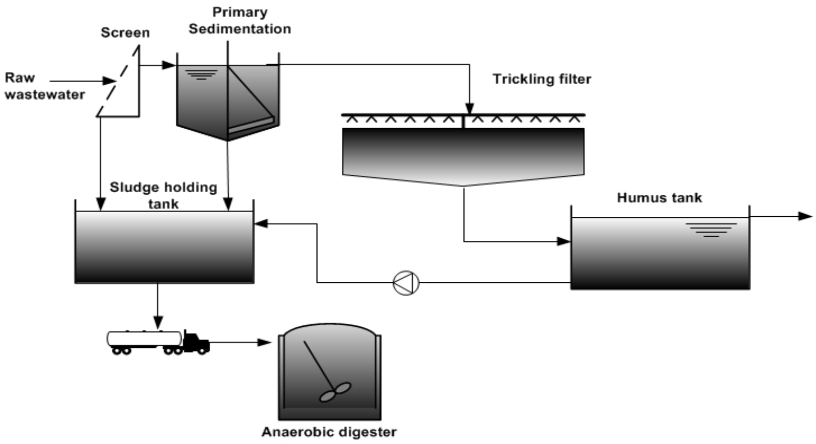
3. Life-cycle cost assessment (LCCA) for both flowsheets, to incorporate the energy demands and carbon footprint from objectives 1 and 2 with capital and operation costs

6.2.2 Materials and methods

Two flowsheets were chosen to be modelled to compare a proposed AP treatment works with current established technologies (Figure 6-1). The base case flowsheet reflected a standard decentralised flowsheet, comprising a course screen followed by a primary sedimentation tank (PST), trickling filter (TF) designed for BOD removal and nitrification, and humus tank (HT) as final clarifier. An on-site sludge holding tank (SHT) was designed for 30 day sludge retention before sludge was exported to a centralised mesophilic AD. The second flowsheet modelled the AP, with secondary TF and HT, with additional on-site infrastructure of a micro combined heat and power (CHP) unit for conversion of biogas collected from the AP. Despite findings suggesting successful operation of APs at low HRTs (Chapter 5), the AP was designed for a 2.3 d hydraulic retention time to incorporate the more comprehensive data set collected from this thesis over a year-long study period (Chapter 3). The flowsheets were designed to meet an effluent quality of <10 mg L^{-1} BOD, <30 mg L^{-1} TSS and <3 mg L^{-1} NH_4-N.

To reflect a decentralised UK treatment works with combined sewerage, a 2,000 PE was chosen with a per capita flow rate of 200 L d^{-1} and a weak strength wastewater as characterised by Tchobanoglous et al. (2003). Modelling was undertaken in Microsoft Excel assuming steady state conditions. Sludge held on site for 30 days was assumed to degrade in situ in accordance with the findings of Chapter 3, and transportation distance to AD was set at 15 km. (McAdam et al., 2011). Biogas yields and energy requirements for centralised AD have been attributed to sludge imports by normalising standard AD values per cubic metre sludge. Further parameters and assumptions for the energy and carbon modelling can be found in Table 6-1.

(a)



(b)

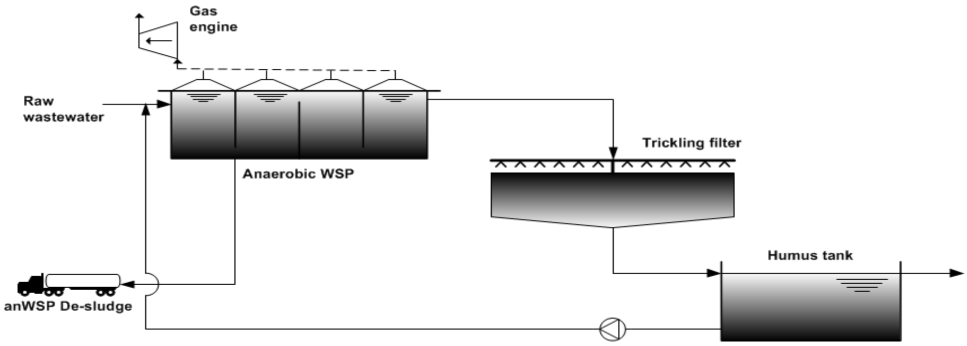


Figure 6-1 Model flowsheets for the (a) a conventional decentralised treatment works, and (b) a decentralised works incorporating an anaerobic pond

Life-cycle cost assessment (LCCA) was conducted on the two flowsheets assuming a 20 year M&E asset life. Costs were calculated in British Pound Sterling (£), using costs sourced from the UK wherever possible. Where costs were quoted in alternative currencies conversions were made at the current exchange from XE.com. Capital expenditures (CAPEX) were not depreciated (Norris, 2001), and final disposal costs could not be estimated so were excluded for all assets. The PST, TF, AP and SHT were all assumed to be excavated reinforced concrete, with the HT above ground reinforced concrete. An intermediate pump was included to account for the additional pressure head required for the HT on both flowsheets, with an additional 15 % added to capital infrastructure costs to account for miscellaneous fittings and 40 % for installation costs (Young et al., 2012).

Table 6-1 Summary of parameters and assumptions for flowsheet energy and carbon modelling

Design parameter	Units	Value	Notes	Reference
Screen				
Energy demand	kWh m^{-3}	0.0023		(McAdam et al., 2012)
Fugitive emissions	$kgCO_2e\ t^{-1}$RDS	0.3		Czepiel, 1993
Primary sedimentation				
Hydraulic retention time	h	3.0		(Foley et al., 2010)
Area	m^2	12.5	Assume 4 m depth	
Sludge generation	$m^3\ d^{-1}$	1.18	60% solids removal	(Tchobanoglous et al., 2003)
Energy demand (scraper)	kW d^{-1}	1.0	Assume 0.18 kWh $PE^{-1}\ y^{-1}$	Thöle. 2008
Anaerobic pond				
Hydraulic retention time	d	1.5		
Area	m^2	150	Assume 4 m depth	
Sludge generation	$m^3\ d^{-1}$	0.03	Assume 0.06 $m^3\ PE^{-1}\ y^{-1}$	Chapters 3 & 5
Biogas energy yield[a]	kWh d^{-1}	6.4	Assume 8 $LCH_4\ m^{-3}$WWT	Chapter 3
Trickling filter				
Organic loading rate	kg BOD $m^{-3}\ d^{-1}$	0.2		(Tchobanoglous et al., 2003)
	g TKN $m^{-2}\ d^{-1}$	0.6	Assume 20 mg L^{-1} TKN	
Area	m^2	98	Assume 2 no. 3 m depth	(Tchobanoglous et al., 2003)
Sludge generation	$m^3\ d^{-1}$	0.02		
Energy demand	kWh d^{-1}	1.6		(Tchobanoglous et al., 2003)
Humus tank				
Upflow velocity	m h^{-1}	1.5		(Tchobanoglous et al., 2003)
Area	m^2	11	Assume 3 m depth	Tchobanoglous et al., 2003
Sludge generation	$m^3\ d^{-1}$	0.003		
Energy demand (scraper)	kWh d^{-1}	2.3	Assume 0.42 kWh $PE^{-1}\ y^{-1}$	Thöle. 2008
Sludge holding tank				
Area	m^2	40	For 30 d holding, 3m depth	
Fugitive emissions	$kgCO_2e\ d^{-1}$	2.4	Assume 57 $mgCH_4\ P^{-}E^{-1}\ d^{-1}$	Chapter 3
Anaerobic digester				
Hydraulic retention time	d	15		
Biogas energy yield[a]	kWh m^{-3} sludge	7.7		
Energy demand[b]	kWh m^{-3} sludge	2.2		Tchobanoglous et al., 2003
Emissions for grid electricity	$kgCO_2e\ kWh^{-1}$	0.484		(McAdam et al., 2012)
Emissions from sludge tankering	$kgCO_{2e}$/t/km	0.114		(McAdam et al., 2011)

[a]Assumed methane conversion of 10 kWh_e/m^3 and on-site electrical conversion efficiency of 20%, centralised electrical conversion of 40%

[b] Includes energy for sludge dewatering, thickening, AD mixing and heating

In house data for CAPEX and OPEX were provided on a confidential basis by a UK water utility. The price of the CHP engine was provided by the in-house data and includes built-in biogas scrubbing, however this cost is typically tailored to site-specific usage and therefore is only an estimate. Whilst energy and associated emissions costs were calculated for AD per cubic metre sludge imports from the flowsheets, capital assets for AD were assumed to be existing and therefore not included. Operational expenditures (OPEX) included an emissions cost set at the UK carbon floor price for 2014-15 confirmed by the UK Treasury (Ares, 2013) in order to incorporate environmental impacts into the economic assessment. Maintenance schedules were estimated after consultation with a UK water utility, with site visits occurring weekly for the TF flowsheet, and monthly for the AP flowsheet. All further parameters and assumptions for the LCCA can be found in Table 6-2.

Table 6-2 Summary of parameters and assumptions for the LCCA

Parameter	Units	Value	Notes	Reference
CAPEX				
Land	£ m^{-2}	1.84		(RICS, 2013)
Excavation	£ m^{-3}	5.30/3.50	First 200 m^3/additional	(SEERAD, 2001)
Reinforced concrete	£ m^{-3}	187/163/92	First 4 m^3/next 20 m^3/additional	(SEERAD, 2001)
Intermediate pump	£	5,200		In house data
CHP engine	£	6,000		In house data
AP cover	£ m^{-2}	20		(Aardvark EM Ltd., 2009)
TF media	£ m^{-3}	83	Assume 10 year replacement	www.alibaba.com
OPEX				
Civils maintenance	£ y^{-1}	3,250	Maintenance every 5 years	In house data
M&E maintenance		Maintenance once a year, 2% of capital costs		(Young et al., 2012)
Maintenance visits	£ d^{-1}	41.80		In house data
Energy	£ kWh^{-1}	0.14	Same price for buy-back	(McAdam et al., 2011)
Sludge transport	£ t^{-1} km^{-1}	0.14		(Jeanmaire and Evans, 2001)
Emissions cost	£ $t^{-1}CO_2e$	9.55		(Ares, 2013)

6.2.3 Results

6.2.3.1 Energy balance

Energy balances were calculated by subtracting the energy generated, both on and off site, from the overall energy demand of the flowsheets. Negative energy balances were calculated for all flowsheets, demonstrating that additional energy would be required in all cases (Figure 6-2). The AP flowsheet required the least additional energy demand, with 1.7 MWh y^{-1}, with energy demand of 4.1 MWh y^{-1} offset by 56 % by the on-site energy generation. Whilst the TF flowsheet had a similar total energy balance to the AP, at 2.0 MWh y^{-1} required, the energy demand was offset by centralised AD, therefore the site requirements of the works would be 5.4 MWh y^{-1}.

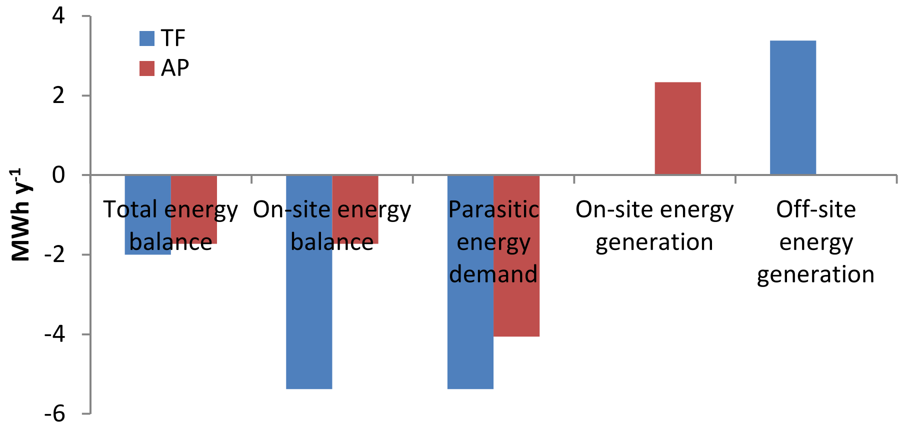


Figure 6-2 Energy balance trickling filter (TF) and anaerobic pond (AP) flowsheets

6.2.3.2 Carbon footprint

Carbon accounting for each of the flowsheets was divided into three categories: emissions generated from net energy required; fugitive emissions calculated by release of greenhouse gases from the treatment processes, and emissions associated with the transportation of sludge from site to centralised sludge management facilities (Figure 6-3). Emissions from energy requirements formed the largest proportion of the AP and flowsheet, accounting for 93 % of total calculated emissions. Fugitive emissions primarily arose from on-site sludge storage, which not only has an environmental impact but also negatively affects the value of the sludge once imported to AD (Chapter 3). For the TF flowsheet, emissions from sludge transportation were the most significant, comprising 36 % of total calculated emissions, and highlighting the impact of sludge management at decentralised sites. The desludge frequency calculated for the AP was 2 years, reducing tankering visits to site from 240 for the TF to 10 for the AP over the 20 year period.

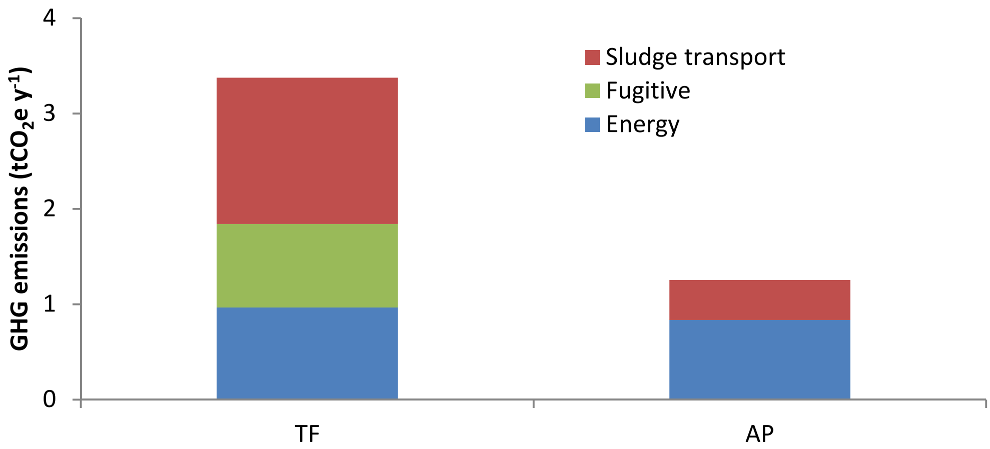


Figure 6-3 Greenhouse gas (GHG) emissions, expressed as carbon dioxide equivalents, from the trickling filter (TF) and anaerobic pond (AP) flowsheets

6.2.3.3 Life cycle cost assessment

Over the 20 year LCCA, the TF and AP flowsheets were very similar in costs, at £240,481 and £252,749, respectively (Figure 6-4). Higher CAPEX for the AP infrastructure, notably the size of the pond and the additional costs for biogas collection and utilisation, were offset by lower OPEX in maintenance requirements and sludge transport. In the AP flowsheet, CAPEX was actually higher than OPEX, with capital costs over three times the operational costs over the 20 year period. Interestingly, the CAPEX costs in the AP flowsheet were dominated by the infrastructure costs rather than the traditional assumption that land costs are prohibitive for extensive systems. The cost of land comprised 0.1 % and 0.2 % of the total costs for the TF and AP flowsheets, respectively, indicating cost of land was not a significant factor, whilst carbon costs also comprised less than 1 % in both flowsheets. Infrastructure was found to be the largest component, comprising 46 % and 76 % of total costs for the TF and AP, respectively.

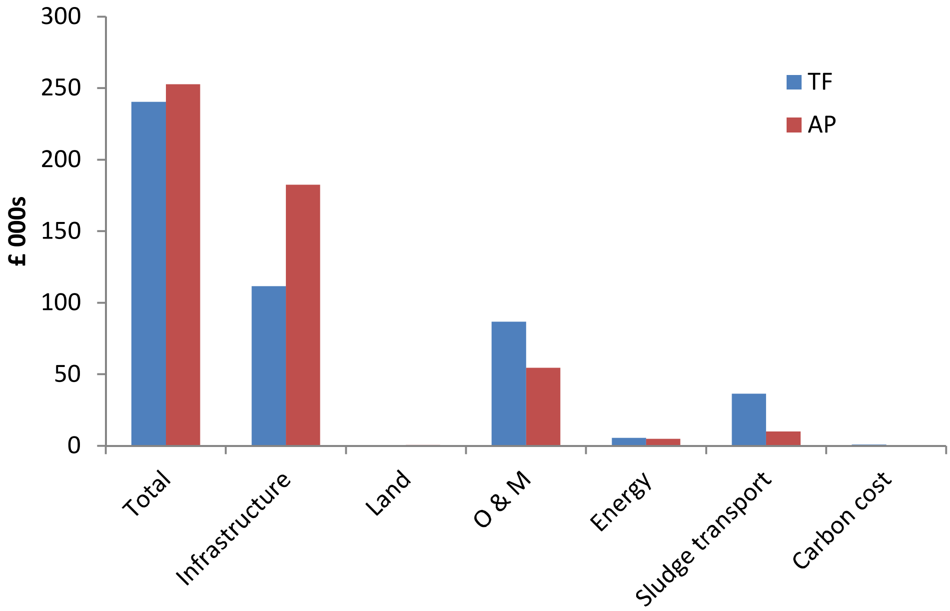


Figure 6-4 Costs calculated for the 20 year life cycle cost assessment (LCCA) for the trickling filter (TF) and anaerobic pond (AP) flowsheets

6.2.4 Discussion

Flowsheet modelling of an AP flowsheet demonstrated the potential advantages of incorporating this technology into decentralised WWT flowsheets. Compared to a current standard aerobic example flowsheet, APs present opportunities for decreasing energy demands, particularly on-site, and lowering GHG emissions, whilst providing competitive whole-life costing. Whilst biogas produced from the AP was not able to cover the entire energy demand of the site, the small difference remaining of 1.7 MWhr y^{-1} could potentially be provided by renewable energy such as solar or wind, enabling an off-grid treatment works. If feasible, this would not only reduce carbon emissions and electrical costs further, but also eliminate the need for a grid connection, a significant capital cost which was not considered in this modelling exercise (Richards, 2014). Whilst the practicality of an entirely off-grid energy works would depend on the natural resources of the location, this potential further enhances the case of an AP flowsheet that is largely self-sufficient and requires little input, for energy or operation and maintenance. Furthermore, UK energy prices for medium sized industrial users have risen 5 % since 2008, whilst the UK has the poorest progress towards its renewable energy targets of any of the EU-27 countries (DECC, 2012). These additional drivers towards renewable energy and reducing reliance on grid-bought energy make pursuing the feasibility of off-grid WWT even more attractive. Additionally, the extended sludge storage time on site lead to a desludge frequency of 2 years. Whilst monthly sludge tanker visits would require the construction and maintenance of a permanent access road, a temporary access surface could be used for the AP desludge, eliminating another significant infrastructure cost (Richards, 2014).

Whilst the AP flowsheet demonstrated the potential to cut carbon emissions, the economic gains from these reductions were not significant on an individual site basis. This is due both to the low emissions for such small works, and the economic cost of carbon as currently recognised in the UK. However, the government 'floor price' initiative will see significant increases in the price of carbon in subsequent years , with prices rising from £4.94 $t^{-1}CO_2e$ in 2013/14 to the 2014/15 price used in this study, £9.55 $t^{-1}CO_2e$, up to an indicative rate of £24.62 by 2017/18 (Ares, 2013). This 398 % rise in carbon costs in 4 years will further the case for carbon savings from WWT works (Figure 6-5), alongside the current requirement of water utilities to report the associated emissions from their commercial activities as a sustainability indicator (Water UK, 2012).

Whilst the AP flowsheet included in this assessment demonstrate the potential for the AP to generate energy through a micro-CHP engine, an alternative option would be to flare the biogas on-site. Whilst this would eliminate the potential of energy generation from the AP, the benefit of low energy demand is still realised and the potential for off-grid energy from other renewable sources is still possible. The benefits of gas flaring would be a simpler on-site process requiring less operation and maintenance, whilst maintaining low air pollution and GHG emission. Additional resource recovery options, such as nutrient recovery from secondary treatment (Vohla et al., 2011) or bioplastic production (Ben et al., 2011) from the VFA-rich effluents from the AP could be explored in the future to complement the sustainability and resource recovery potential of the AP flowsheet.

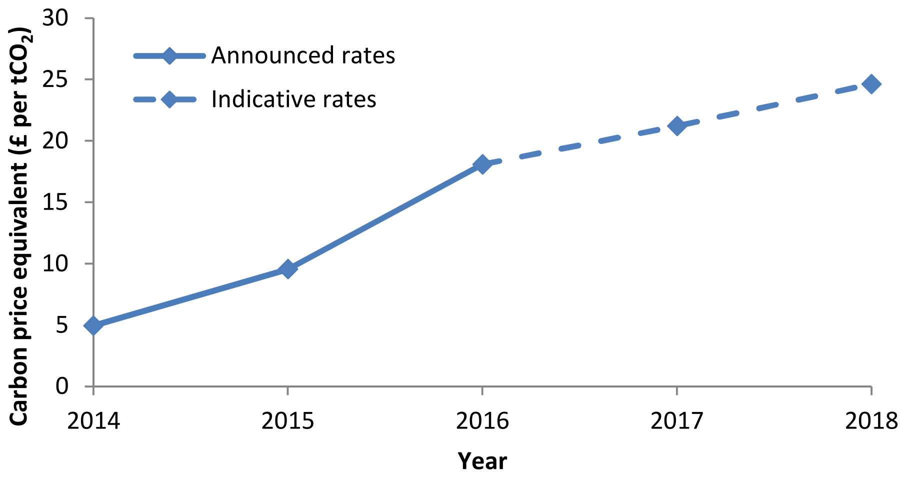


Figure 6-5 Carbon price equivalents announced by the UK Treasury, with set rates until 2016 and indicative rates until 2018 (adapted from Ares, 2013)

Surprisingly, the commonly cited prohibitive factor of APs, the costs associated with extended land requirements, were found to be negligible for the case of rural bare land sites. The land price used for modelling, £1.82 m^{-2}, was a U.K. average for rural 'bareland' (farmland without buildings), with regional averages ranging from £1.11 m^{-2} in Scotland to £2.22 m^{-2} in North West England (RICS, 2013). Whilst prices have risen sharply in recent years, around 134 % since 2007, these increases are largely attributable to large holdings being purchased for commercial and residential development, whereas small holdings, where available, command much lower prices (RICS, 2013) and would be adequate for small WWT works. Previous studies have already determined that land costs are not prohibitive for the development of facultative pond systems in the UK (Mara, 2006; Johnson et al., 2007), and with the decreases in HRTs possible in APs (Chapter 5), the LCCA implications of land requirement are not significant. However, these costs only relate to new bareland sites, and in many situations water utilities will look to refurbish or retro-fit existing assets rather than purchase additional land. Therefore, the possibility of retro-fitting APs to existing infrastructure, such as PSTs or SHTs, should be explored, and reduction in HRT could be decisive in determining the feasibility of both the retro-fits and the possibility of constructing APs on land already owned. Importantly, in the case of the AP, the CAPEX was greater than OPEX, and so if a LCCA was conducted over a period greater than 20 years the AP flowsheet would present further reductions in whole-life cost. If refurbishment of existing assets is a strategy for water utilities past the standard 20 year asset life, then initial investments in APs may provide greater payback in the long term.

Traditionally perceived benefits of APs in reducing operation, maintenance, and sludge handling requirements, were support by the LCCA. The UK Water sustainability drivers to reduce sector GHG emissions and energy requirements, whilst increasing renewable energy utilised (Water UK, 2012; Chapter 1), provide a strong case for the consideration of APs for decentralised WWT. These drivers are also reflected economically in the LCCA, where rising energy and carbon prices will continue to put pressure on the water sector to find alternative solutions for WWT in decentralised areas, and the large number of these small works require a new approach in order to reduce the current burden of maintenance and sludge handling requirements.

6.2.5 Conclusions

The potential advantages of incorporating APs into decentralised WWT flowsheets has been demonstrated through flowsheet modelling against current standard options.

- Whilst neither of the flowsheets modelled could achieve full energy self-sufficiency, either on-site or as a total balance, the AP provided the closest balance to energy neutral, thereby reducing energy costs and associated emissions, and providing the opportunity for renewable energy sources to be explored to enable off-grid WWT.
- The AP flowsheet a lower carbon footprint compared to the standard flowsheet, with reductions from in fugitive emissions, energy requirement, and sludge transportation. Whilst current carbon prices do not present a strong economic incentive for carbon reductions when incorporated into a LCCA, significant rises in carbon pricing are expected in coming years, and non-economic incentives in reducing carbon emissions are strong.
- The cost of additional land for an extensive treatment system, commonly identified as a significant barrier to APs and other natural processes, was found

to be largely insignificant when considered in the LCCA. However, in many scenarios retro-fitting or refurbishing of existing assets will be preferred to purchase of new bareland sites, and the potential of APs for these applications should be explored.

- Overall LCCA over a 20 year period found the AP to be competitive with a standard flowsheet. Significant savings were identified in OPEX, and therefore longer operational periods than 20 years would further improve the economic viability of the AP flowsheet.

6.2.6 References

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7 Conclusions and future work

7.1 Conclusions

The major conclusion of the work is that anaerobic ponds (APs) are feasible for the treatment of low strength, low temperature wastewaters, and indeed present an attractive alternative to current primary treatment options, particularly for small decentralised works. The decoupling of hydraulic retention time (HRT) from solids retention time (SRT) mitigates the challenges related to organic breakdown presented by the low temperature and weak influent concentrations, and also enables HRT to be reduced by 200 – 300 % without a decline in effluent quality.

Specific conclusions were:

1. A review of current AP design and practice found that most APs are currently underloaded, largely to avoid odour complaints, but this underloading leads to unnecessarily large footprint and inhibits the digestion process through restricting biomass/substrate contact. Current design guidelines are not suitable for recent developments and uses, as the covering of APs prevents odour escape and enables higher organic loading rates, whilst the use of baffles can improve mixing to enhance organic degradation (Chapter 2, Objective 1)
2. APs can be effectively applied to low temperature, weak strength wastewaters. Even an unoptimised design trialled over an extended period demonstrated the potential for methane generation at extreme temperatures, provided adequate primary treatment, and recorded a sludge accumulation rate comparable to APs operated at higher temperatures. The extended solids retention time and acclimatisation of biomass was able to compensate for the low temperatures and weak wastewater, and can provide an attractive alternative to current primary treatment options, through reducing fugitive greenhouse gas (GHG) emissions and providing less frequency desludging requirements (Chapter 3, 6) (Objectives 2, 5)
3. The use of baffles should be recommended in all APs, due to advantages to be inferred in facilitating mixing patterns between baffles, therefore enhancing

biomass/substrate contact, whilst providing an overall plug flow effect through the entire pond, enabling the retention of biomass. Furthermore, the removal mechanism with the pond can be manipulated with use of baffles, with different orientations generating different flow patterns and therefore creating conditions preferential for greater solids settlement and capture, or mixing and contact (Chapters 4, 5) (Objective 3)

4. Hydraulic retention times can be decreased from current design guidelines, and provide benefits in reducing footprint and increasing mixing without compromising effluent quality, especially with the inclusion of baffling. For weak strength wastewaters trialled, an HRT of 1 - 1.5 days could be recommended (Chapters 2, 3, 4, 5) (Objectives 3, 4)
5. Microbial communities within APs adapt to operating conditions, providing improvements in methane production over time. Hydrogen pathways for anaerobic digestion were found to dominate, and microbial community profiles changed with physical distance along the pond. By using different baffling structures in different regions of an AP, preferential conductions for specific communities could be engineered to facilitate staged anaerobic digestion (Chapters 3, 4, 5) (Objectives 2, 4)
6. A compelling case can be made for inclusion of APs for decentralised wastewater treatment, due to advantages gained in decreasing sludge management requirements whilst providing suitable primary treatment, with additional potential benefits in renewable energy generation, which could increase both with improved biogas yields and the option of combining with other renewable technologies. In some circumstances, it may be possible for an AP flowsheet to operate entirely off-grid, eliminating the need for costly infrastructure such as permanent access roads and national electrical grid connection (Chapter 6) (Objective 5)

7.2 Future work

Building on the findings from this thesis, area for further work have been identified for both gaining further knowledge of APs to feed into improved design, as well as in scaling up the work to promote the implementation of APs at full scale, especially in the UK water sector.

7.2.1 Furthering the fundamental understanding of AP processes

The dominance of hydrogen pathways in the anaerobic digestion during the pilot trials has not been reported in APs previously, possibly due to its occurrence at low temperatures (Collins et al., 2005; Connaughton et al., 2006), and should be further investigated. The competition for hydrogen, particularly at the front end of the AP, should be explored by further analysis of microbial communities and their relative abundances along reactor length, whilst the establishment of aceticlastic methanogens in the latter sections of the staged AP may increase VFA utilisation if it can be targeted in design. Better understanding of the microbial communities present, their changes within the AP, and their preferential growth conditions may lead to an optimisation design through baffling, and flow rates, for distinct sections of the pond. A sampling regime capturing spatial changes in microbial community and its activity, measured through the use of quantitative polymerase chain reaction (qPCR) and specific methanogenic activity (SMA) tests such as those utilised in Chapters 4 and 5, should be extended through sampling from start-up of a pilot trial and through regular time intervals to determine the development of the community with time. This would be complemented by the use of TRFLP fingerprinting to determine population diversity and support community profile conclusions from the qPCR analysis. The data collected would be combined with further literature study of the kinetics and preferred conditions of the bacteria and archaea identified to determine whether engineering design interventions can be used to enhance the digestion pathways utilised.

The development of the two stage AP should be continued to both improve the design of the separate sections for their intended purpose, whilst consideration should also be given to the relative sizing of each stage. Whilst in the staged AP operated in this thesis equal sizing and retention time was applied to both stages, greater understanding about the relative retention times and requirements for the separate stages, should identify ways to optimise both the individual sections separately as well as the overall process (van Haandel et al., 2006; Sawajneh et al., 2010). Analysis of results from the microbial ecology trials suggested could develop guidelines on optimisation criteria for separate stages of the digestion process, and different combinations of relative HRT, through different sizing of the respective stages, could be conducted at pilot scale to identify optimal loading rates for synergy between hydrolysis and methanogenesis.

The extent and impact of dissolved methane is likely to be significant for both the reflection of true methane yields from APs, and how its potential recovery could improve the energy balance of the process and subsequent flowsheets (Cookney et al., 2012). Furthermore, changes in temperature will affect the level of methane saturation in the liquid phase, and therefore impact the perceived seasonal variations in methane production. Whilst effluent dissolved methane concentrations were not been measured in this work, its potential impact has been identified and warrants further investigation. Dissolved methane concentrations can now be measured accurately (Cookney et al., 2012), and this analysis should be incorporated into future APs trial to enable calculation of true methane yields.

7.2.2 Implementation of APs at full scale

Trials in this thesis were conducted at bench scale, and succeeded in establishing the effectiveness of APs at low temperature, whilst furthering understanding of AP principles through observation in a controlled environment. However, the findings of this work must be scaled up, for verification of the findings at true pilot scale or at a small, full scale works. Furthermore, extended operation at larger scale is recommended to observe AP operation over seasonal variations at the recommended shorter HRTs, such as 1 and 1.5 d, whilst also benefitting from more consistent liquid and sludge temperatures through earth insulation provided in an excavated pond. These trials should reflect the methodology of Chapter 3, with the addition of dissolved methane and microbial community analysis described in Section 7.2.1. This would lead to the establishment of true methane yields to determine energy self-sufficiency of sites or additional requirements for off-grid renewables. The practicalities of full scale operation, such as gas storage and utilisation at scale, and economies of scale for equipment including potential use of methane stripping in effluent, should be determined in order to present a full business case for implementation of APs in the UK wastewater sector. This work should be conducted in collaboration with industry, in order to engage with stakeholders in the final implementation of the technology, whilst also facilitating knowledge exchange between the design developments and scientific understanding of APs and the practical aspects of costing and constructing wastewater treatment assets.

Finally, with extended trials and larger data sets, sensitivity analysis around the impact of alterations to HRT, and therefore volume, should establish guidelines for designers on dimensioning APs for the relative specific requirements of sludge holding, methane production, treatment performance, and physical footprint. This work should build on the flowsheet modelling conducted in Chapter 6, but incorporate more complex explorations around alterations in design that would be made possible by data collected from larger scale and longer running trials. This will enable AP design to evolve from a single set of empirical equations, to being adaptable to specific site requirements and a designer's desired outputs.

7.3 References

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Appendix A – Theory behind basic principles discussed in the thesis

Anaerobic digestion

Anaerobic digestion (AD) is the biological process of degrading organic matter into primarily methane and carbon dioxide. The AD process is a complex relationship between many different microbial consortia that work successively to produce intermediary substrates for subsequent consortia, with each ecological community dependent on a variety of environmental factors such as pH, temperature, and the presence or absence of chemical compounds that are either necessary for growth and metabolism or that have an inhibitory effect.

Broadly, AD can be categorised into four main stages (Figure A-1). The first two stages are hydrolysis and acidogenesis, in which complex organic matter such as lipids, proteins and carbohydrates are broken down into long chain fatty acids, amino acids, and monosaccharaides. The steps of hydrolysis and acidogenesis are carried out by strict anaerobic bacteria, primarily the families *Bactericides* and *Clostridia*, as well as the facultative family *Streptococci.* The next step is acetogenesis, in which the products of acidogenesis are further degraded into short-chain fatty acids, such as acetic, proprionic, butyric and valeric acids, as well as hydrogen and carbon dioxide. Acetogenic bacteria include *Acetobacterium woodii* and *Moorella thermoacetica*, and have an optimal operating pH range between 5 and 6. In the final stage of AD, methanogenesis, the short chain fatty acids, and hydrogen, are converted to methane and carbon dioxide. Methanogenic archea include *Methanosarcina* spp. and *Methanothrix* spp, which utilise acetate as a substrate, and *Methanobacterium* and *Methanococcus,* which utilise hydrogen along with formate. Methanogenic archea operate in an optimal range of 6.8 to 7.2, and are sensitive to pH below 6 – a major cause of instability in AD systems is 'souring', when acidogenesis and acetogenesis dominate and lower the pH to the point of inhibiting methanogenesis.

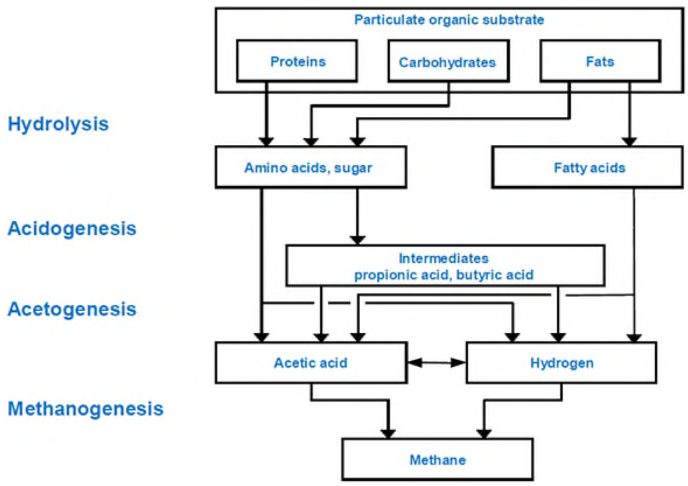

Figure A-1 Main pathways in anaerobic digestion. From WtERT, 2009.

Alongside pH, the AD process is highly temperature dependent. In general, at lower temperatures the metabolic processes of the microbes involved in AD slow down, and therefore digestion time is longer. However, different microorganisms at each stage of the AD process have different optimal operating temperatures, and are typically classified into three classes: psychrophilic, with an optimal temperature range between 10 and 20^{o}C, mesophilic, with an optimal range between 30 and 45^{o}C, and thermophilic, with an optimal range of 55 to 70^{o}C (Figure A-2). During temperate AD, such as studied in this thesis, a psychrophilic microbial community would yield the highest metabolic rates. However, psychrophilic anaerobes are rarely observed and measured, and often cold-adapted communities develop, where mesophilic anaerobes adapt to the colder temperatures, and whilst they may still be effective at low temperature – and increasingly so with time and increase adaptation, the growth rates are likely to be lower than those of true psychrophilic communities (Figure A-2).

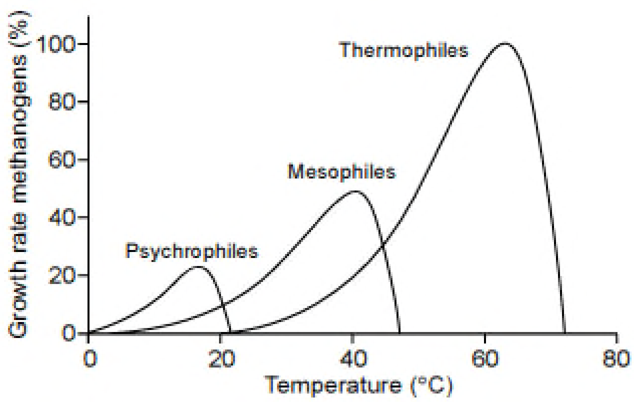

Figure A-2 Relative growth rates of methanogen classes. From Lettinga et al., 2001.

Quantitative polymerase chain reaction

Quantitative polymerase chain reaction (qPCR) is an extension of the original polymerase chain reaction (PCR) technology first developed by Kary Mullis in 1983. The PCR process is an amplification process of specific pieces of DNA, which over the course of a number of heating and cooling cycles can amplify the target DNA from a single copy to thousand of millions. The method first involves identified the target DNA that is to be amplified, and developing primers – short DNA fragments – to isolate the target DNA, which is then used as a template for replication onto a DNA polymerase, which is an enzyme that assembles new DNA strands from the replicated target DNA. This initiates a chain reaction, which is driven by thermal cycling – repeated heating and cooling of the sample in order to sequentially denature (break down the existing DNA to access the target strand isolated by the primers), anneal (the process of attaching the isolated target DNA to the polymerase) and elongation (the final synthesis of the new DNA strand between the target and polymerase, to complete the reaction). For each thermal cycle, the amount of target DNA will double due to the duplication process, and therefore the target DNA count will grow exponentially.

The PCR process was originally used to generate large quantities of duplicates of DNA samples that may be found only in small numbers in a sample. However, due to the duplication process with each thermal cycle, qPCR was developed as a method of quantifying the number of DNA strands present in the original sample, by counting the growth rates of the DNA count with each thermal cycle. For the qPCR process, alongside the primers to isolate the target DNA, an additional fluorescent dye, or probe, is included, which reacts only with the target DNA strand. Therefore, all primers and probes have to be developed specifically for the exact DNA strand targeted. As the thermal cycling process occurs, the fluorescence of the sample is measured, with the increased fluorescence after each cycle attributable to the increase in the target DNA count (Figure A-3). In this way, with the use of standards of known quantities, it is possible to calculate the number of target DNA strands initially present in the sample.

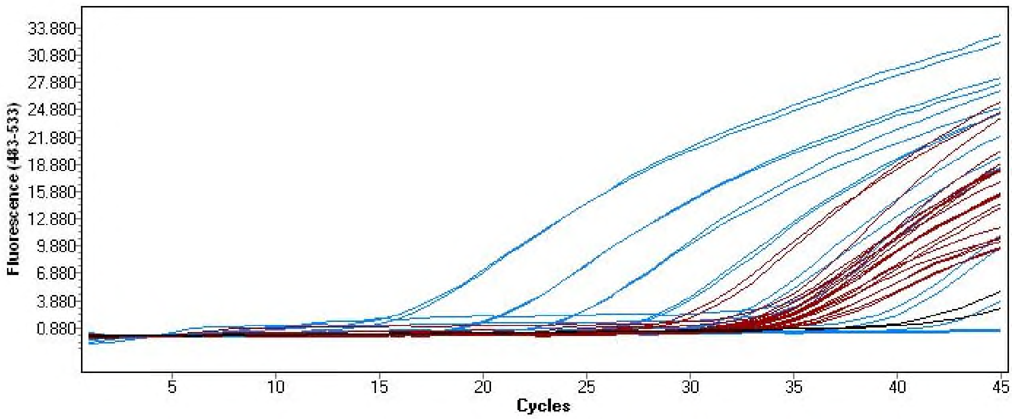

Figure A-3 Fluorescence readings from target DNA for the methanogenic order *Methanosarcinaceae*, as measured from a sample during this study

The target DNA strand is selected to be unique to the microorganism to be identified, and therefore the concentration of the target microorganisms in the sample can be determined.

Computational Fluid Dynamics

Computational fluid dynamics (CFD), is a form of modelling utilising fluid mechanics principles and finite element analysis, a discipline of mathematics that divides a larger model into many, often millions, of small fractions in order to resolve flow equations for each element, combining to provide a complex, comprehensive simulation of the original model. As computer processing power has developed CFD has broadened, originating from simple, one phase (ie a single gas or liquid), two dimensional models with a small number of elements, to current models that can simulate multi-phase, dynamic conditions in 3D with highly evolved models and large numbers of elements. The practice is applied to many scientific disciplines, such as aerodynamics, meteorological modelling, biomedical engineering, and in the water industry for a number of applications such as contact times in mixing tanks and optimisation and investigation of reactor design, such as used in this thesis.

The emergence of CFD for pond modelling has the potential to be a valuable tool in pond hydraulics. Whilst tracer studies, the traditional form of hydraulic assessment in ponds, are useful, they can also be costly and time consuming, and do not provide any input for the design stage of a pond as it already needs to be constructed before the studies can be undertaken. CFD offers the potential to predict the performance of ponds in the design stage, as well as model dynamically for changes in pond conditions. A further advantage CFD has over tracer studies is the ability to analyse the flow within the pond itself, rather than the more general picture developed through tracer analysis of the effluent. CFD can provide graphical representations of pressures, flows, velocities, and temperatures at any point within the model, whereas empirical data is often limited to what can be collected by grab samples.

There is a danger, however, in placing too much importance on CFD. The wealth of data CFD simulations can provide make it tempting for designers to base all design assumptions on the model outputs. However, as with all modelling, the model is limited by the information provided, and this is often incomplete, and can be difficult to simulate dynamic conditions external to the pond, such as temperature and wind variation, or the often unpredictable nature of microbial community development and adaption to conditions. Therefore, where CFD is used to model existing systems, validation must always be carried out with empirical data to corroborate the accuracy of the models, and identify deviations to be aware of model limitations. Where CFD is used in the initial design phase, it can be used as a roughing measure to test a number designs – which is often not practical experimentally – in order to shortlist promising outcomes for further investigation.

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