Can micro-scale anaerobic digestion be viable?

An investigation of the techno-economics and flexibility of micro-scale anaerobic digestion for future routes to the market

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Abstract

In the developed world, anaerobic digestion (AD) is commonly installed on a large scale (over 500 kWe) and far less at micro scale (less than 50 kW). One reason for this discrepancy is that there is an economy of scale. However, micro-scale AD has potential advantages in social, technical and environmental terms. This thesis aimed to evaluate and quantify these advantages, through a combined experimental and techno-economic approach.

Flexible biogas production can enhance AD profitability and was investigated experimentally. Two experimental streams were run with different loading patterns and the operation and stability were studied under the different conditions. Under a variable load pattern, the test digester showed better volatile solids degradation, a more pronounced immediate response to feeding events, and a higher methane production rate than a digester fed at a continuous load.

An operational 2 m^3 micro-scale AD plant in London, UK was monitored for a year whilst running on local food waste. The plant averaged a processing rate of 12.6 kg day^{-1} and achieved a specific methane production of 132.4 m^3 CH_4 $tonne^{-1}$ wet waste with an average biogas methane content of 60.6%. Signs of ammonia toxicity were successfully addressed by the addition of a trace element solution. The plant had a simple payback period of 148 years due to low revenues.

A technoeconomic analysis was performed for a theoretical micro-scale AD plant with a yearly input of 119 tonnes of food waste and 6 tonnes of vegetable oil. The simple payback time of the scenarios ranged from 5.4 to 11.9 years. The best solution included a biogas boiler and a composting system, adding cardboard and green waste to the digestate output. A sensitivity analysis showed that the simple payback time was most affected by the value of compost, the value of electricity and by government initiatives such as the Renewable Heat Incentive.

Keywords: Micro-scale anaerobic digestion, flexible biogas production, techno-economic analysis, case study.

Declaration

The candidate confirms that the work submitted is her own, except where work that has formed part of jointly authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

The work that appears in chapter 5 was published in a jointly authored publication (with Helen Theaker as a co-author), the details of which are as follows:

> WALKER, M., THEAKER, H., YAMAN, R., POGGIO, D., NIMMO, W., BYWATER, A., BLANCH, G. & POURKASHANIAN, M. 2017. Assessment of micro-scale anaerobic digestion for management of urban organic waste: A case study in London, UK. *Waste Management,* 61**,** 258-268.

The author would like to declare that although chapter 5 is largely her own work, there is a significant amount of overlap with the jointly published paper noted above. A full statement of the author's contribution is provided in Chapter 5.

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ADNet Early Career Researchers conference, 1st-2nd July 2017: Presentation 'Camley Street Micro-AD plant'. Awarded Best Overall Platform Presentation.

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Abbreviations and acronyms

AACE	Association for the Advancement of Cost Estimating International.
AD	Anaerobic digestion.
BMP	Biological methane potential.
CHNS	Carbon, hydrogen, nitrogen and sulphur (test).
COD	Chemical oxygen demand.
CSTR	Continuously-stirred tank reactor.
DCFROR	Discounted cash-flow rate of return.
DM	Dry matter, in %. Also referred to as Total Solids (TS).
EU	European Union.
FIT	Feed-in Tariff.
FOG	Fats, oils and greases.
HRT	Hydraulic retention time.
kWe	Kilowatts of electricity.
kWth	Kilowatts of thermal power.
LCFA	Long-chain fatty acid.
LCOE	Levelized cost of energy.
MTOE	Mega-tonnes of oil equivalent.
NNFCC	National Non-Foods Crop Centre.
NPV	Net present value.
NREAP	National Renewable Energy Action Plan.
OECD	The Organisation for Economic Co-operation and Development.
OFMSW	Organic Fraction of Municipal Solid Waste.
OLR	Organic loading rate.
P&ID	Piping and instrumentation diagram.
PV	Photovoltaic.
RHI	Renewable Heat Incentive.
RTFC	Renewable transport fuel certificate.
RTFO	Renewable transport fuel obligation.
SBP	Specific biogas production.
SFW	Synthetic food waste.
SMP	Specific methane production.
SPT	Simple payback time.
TAN	Total ammoniacal nitrogen.
TEA	Techno-economic analysis.
TPA	Tonnes per annum.
TS	Total solids content.
UASB	Upflow anaerobic sludge blanket.
VFA	Volatile fatty acid.
VS	Volatile solids content.

Contents

1 Introduction

1.1 The World View of Energy

Throughout the world, energy is an essential resource to enable people to live comfortable and productive lives. As the need for energy increases (figure 1-1) and global warming becomes a pressing issue, concerns grow over the reliance on finite, polluting resources such as coal, oil, natural gas and nuclear fuel.

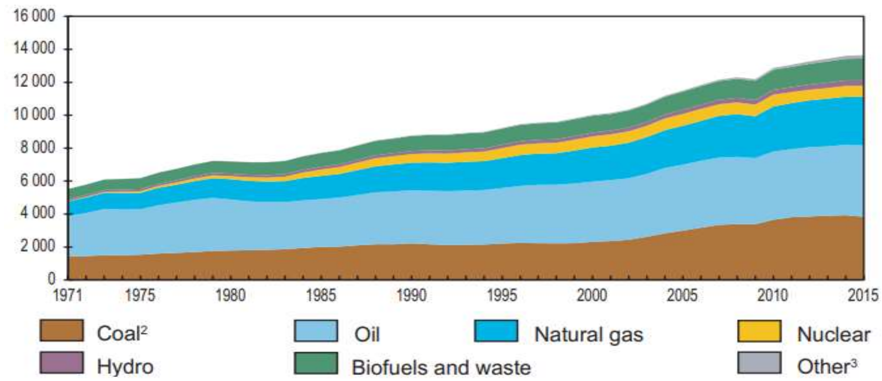


Figure 1-1: World Total Primary Energy Supply from 1971 to 2015 by fuel (MTOE) (International Energy Agency, 2017a)

Governments are increasingly using policy to encourage the development of renewable energy sources. The European Renewable Energy Directive was published in 2009 and set a goal for increasing the market share of renewable energy to 20% by 2020 (European Union, 2009). Subsequently, EU member countries were obliged to produce a national renewable energy plan (NREAP) to fulfil the requirements that this directive set out (European Commission, 2018). Renewable energy goals have now been set by 164 of the 196 countries of the world (International Renewable Energy Agency, 2015). The targets set out in these plans were further strengthened in importance by the signing of the Paris Agreement in 2015. This agreement aims to limit the raising of the global temperature to 1.5°C of pre-industrial levels by reducing global carbon dioxide emissions (United Nations, 2015).

In the UK, the NREAP targets a 15% share of the total energy requirement of the UK to be supplied by renewable energy by 2020, and breaks this down further by energy type, setting the following targets:

- 30% of electricity demand, including 2% from small-scale sources
- 12% of heat demand
- 10% of transport demand (GOV.UK, 2010).

Of these, the UK has already achieved the target for electricity (GOV.UK, 2020a) but so far has not achieved the targets for heat and transport, which have a market share of 7.2% and 8.5% respectively (GOV.UK, 2020a; Business Energy and Industrial Strategy (BEIS), 2019).

1.2 Anaerobic digestion as a renewable energy resource

Anaerobic digestion (AD) is a natural process that occurs when organic matter such as animal slurry, food waste or sewage sludge decomposes in the absence of oxygen (figure 1-2).

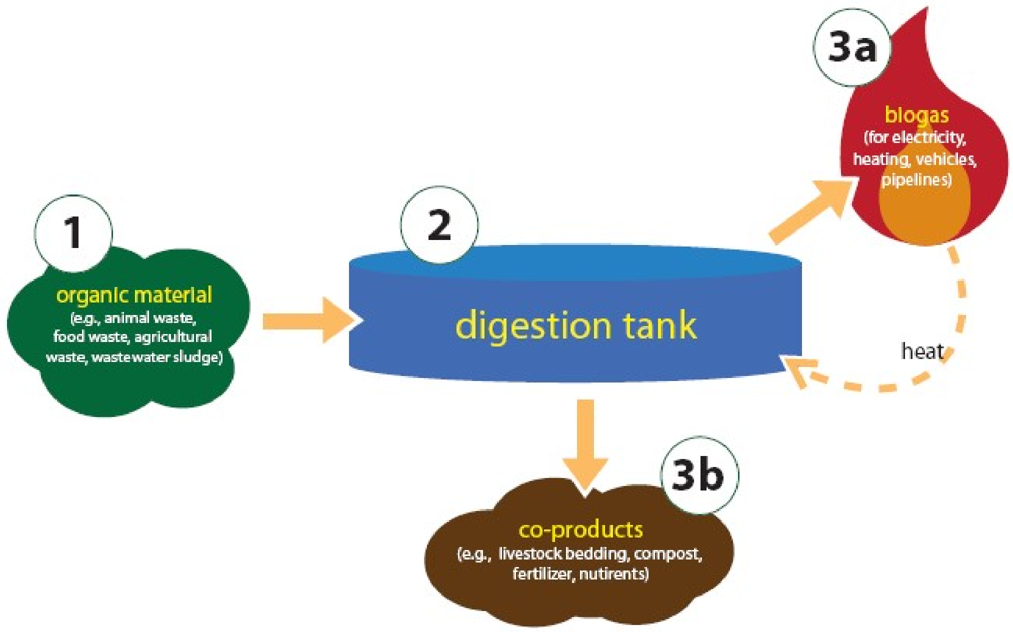


Figure 1-2: The Anaerobic Digestion process simplified (American Biogas Council, 2018).

The process uses microorganisms to break down the organic material into biogas (made up of methane and carbon dioxide) and digestate, an organic, nutrient-rich slurry which can be used as a fertilizer. The biogas created by the process can be burned to create heat, used to run a generator to make electricity, or 'upgraded' into biomethane to be used as a transport fuel or injected into the gas grid.

In recent years, the number of anaerobic digestion plants has increased significantly across the UK (Business Energy and Industrial Strategy (BEIS), 2016a) and Europe (figure 1-3). This is due to concerns about carbon emissions and the implementation of financial incentives for generation of renewable electricity and heat (for example, Feed-in Tariffs and Renewable Heat Incentives).

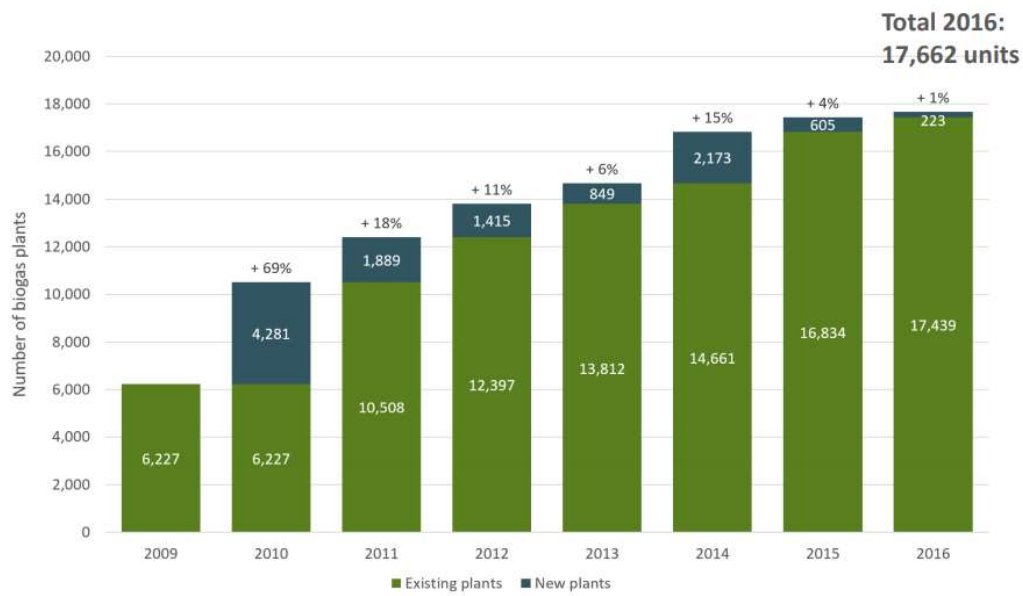


Figure 1-3: Total number of biogas plants in the EU from 2009 to 2016 (European Biogas Association, 2018)

Additionally, recent legislation has been made in the EU and UK that discourages the disposal of waste in landfill sites (European Commission, 2008; GOV.UK, 2011), turning attention towards alternative disposal routes. As waste disposal can now be very costly, alternatives such as anaerobic digestion are becoming more attractive.

Anaerobic digestion creates a storable fuel, which could be used to mitigate the variability of electricity supply from weather-dependent renewable energy sources such as solar PV and wind (Hochloff and Braun, 2014). The biogas from anaerobic digestion could also be used directly to generate heat to assist the decarbonisation of the heating sector (Lauer and Thrän, 2018) or upgraded to produce a vehicle fuel, both of which are key priorities in the Renewable Energy Directive.

It is estimated that anaerobic digestion could fulfil up to 7.5% of renewable energy requirements by 2020 (NNFCC, 2018), but this could increase if marginal feedstocks such as rough grassland were used (Ecotricity, 2016).

1.3 The Circular Economy

The circular economy is a systems thinking perspective on the subject of how to relieve the increasing pressure on the world's resources that has recently gained increasing attention (Ghisellini, Cialani and Ulgiati, 2016),. The concept focuses on the recycling and reuse of resources within an economic system as an alternative to the current 'linear' model, where resources are extracted, used, and thrown away. This encourages a system where economic growth is not directly connected to resource use and environmental pressure (figure 1-4).

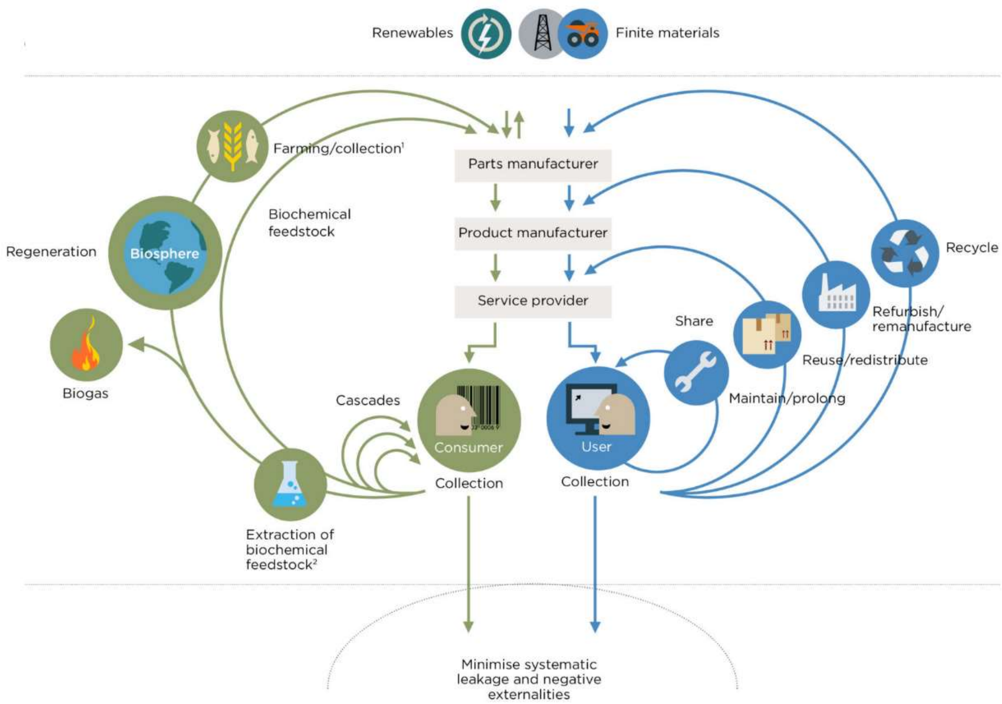


Figure 1-4: Overview of a circular economy (The Ellen Macarthur Foundation, 2017).

Anaerobic digestion can form a key part of this structure by repurposing organic waste into a useful resource, allowing nutrients to be recycled back into the system as fertilizer and by generating biogas.

The circular economy concept has become increasingly important to worldwide government policy in the last ten years. For example, China has adopted the circular economy as an ideology at a macro (government) level, its implementation being included in the 11th and 12th five-year plans in 2006 and 2011 (Naustdalslid, 2014). The EU published its 'Circular Economy Package' in 2014 and 'Circular Economy Action Plan' in 2015 (Lieder and Rashid, 2016; European Environment Agency, 2015), which contain actions and legislation proposals to work towards a circular economy.

1.4 Micro-scale AD

The size of an AD plant can range from a few cubic metres to thousands of cubic metres, depending on the location and purpose. In Europe and the UK, plants tend to be large, with the average size of an AD plant in the UK being 500 kW (Business Energy and Industrial

Strategy (BEIS), 2016a). This size of plant requires a volumetric capacity of about 2-3000 cubic metres, and an input of over 15 tonnes of feedstock per day.

The biological process of anaerobic digestion is the same at all scales, but as the size of plant changes, the technology differs in the plant design, economics, and operational techniques.

Micro-scale AD is widely employed in developing countries, generally at a 'household' size, and is used to process human waste, animal manure and food waste (Hou *et al.*, 2017; Surendra *et al.*, 2014). AD has been promoted by the Chinese and Indian governments and consequently as of 2011 there were 4 million biogas plants in India and 27 million in China, being mostly domestic plants, up to 5 cubic metres in volume (Bond and Templeton, 2011).

In developed countries, AD is employed to extract energy from organic waste and purpose-grown crops to produce electricity and heat, helping to sanitize the waste in the process. However, the uptake of micro-scale plants in developed countries is limited (Fuldauer *et al.*, 2018). Large-scale plants are generally more cost-effective to build (Yaman, Theaker and Walker, 2017), but have disadvantages in that they have a large footprint, and require planning permissions and large amounts of feedstock, placing an increased burden on the transport network. Increased installation of micro-scale AD plants in the developed world would cut down on the carbon emissions associated with transporting organic waste to centralised AD plants (Patterson *et al.*, 2011). Localised organic waste processing could make available feedstocks that are not currently economic to transport to larger plants.

Research on community based resource management found that if resources were managed locally, then this could result in more sustainable behaviour in terms of resource use (Campbell and Sallis, 2013). Therefore, the installation of micro-AD in a community could encourage behaviour change that fits into the pattern of the circular economy.

1.5 Variable feeding in micro-scale AD

The market share of renewable energy is increasing, with a prediction that by 2050 it will make up 97% of electricity generation (Lemmer and Krümpel, 2017). With the greater penetration of intermittent weather-dependent renewable energy sources such as solar and wind, the ability to control the output of an energy source is becoming more attractive, as it can help to ensure that the energy supply stays constant (Hahn *et al.*, 2014b). As a result, flexible feeding for anaerobic digestion has been gaining interest in recent years, particularly in relation to the stability of the plant.

AD plants can be used as a flexible source of energy by using biogas storage or by varying the organic loading rate.

In micro-scale AD, the ability to use a variable feeding rate or supply a variable biogas production rate without causing process instability would provide a number of advantages. The plant could accept new waste streams when they become available, would be more resilient against variations in feedstock supply, and would be able to balance shortfalls or overproduction of heat or electricity in its supply area. The plant could also take advantage of 'premium-rate' electricity tariffs – higher electricity feed-in prices at times of higher demand (Hochloff and Braun, 2014).

1.6 Food waste

Each year, about one third (1.3 billion tonnes) of food production is lost as waste, with 89 million tonnes of food waste being generated within the EU (Xu *et al.*, 2018; Curry and Pillay, 2012).

Food waste is estimated to contain about 2.3 MWh of energy per dry tonne, and the food currently wasted globally could generate approximately 894 TWh of electrical energy, approximately 4% of the global electricity demand of 2016 (Curry and Pillay, 2012; International Energy Agency, 2017b). The recycling or avoidance of food waste therefore has great potential to reduce both energy waste and carbon emissions. Organic waste in the waste stream is a significant global contributor to greenhouse gases. It is estimated that 13% of anthropogenic methane emissions in 2000 was caused by emissions from organic waste in landfill (Ren *et al.*, 2017).

In the UK, a 2012 report (WRAP, 2012b) stated that 7.0 million tonnes of waste were produced by households in the UK, with up to 5.4 million tonnes of this being avoidable (figure 1-5).

Figure 1-5: Household food waste in the UK (WRAP, 2012b).

The increasing concern over the wastage of food has led to the introduction of government policies to encourage better use of organic wastes (Zhang *et al.*, 2014). In 2018, the EU published the circular economy package, which demands that all EU countries have separate biowaste collections by December 31st, 2023 (Moore, 2018). This legislation was followed in 2018 by the UK's Waste and Resources legislation (DEFRA, 2018), which set out a plan to both reduce the production of food waste in the UK and recycle more. This would be achieved by introducing separate collections of food waste across 100% of the UK, and would increase the amount of food waste available to be put to further use. Landfill regulations have also become gradually more stringent. In the UK, a landfill tax was introduced in 1996, and has increased to £88.95 per tonne in 2018 (HM Revenue and Customs (HMRC), 2018; 360 Environmental, 2018) (figure 1-6).

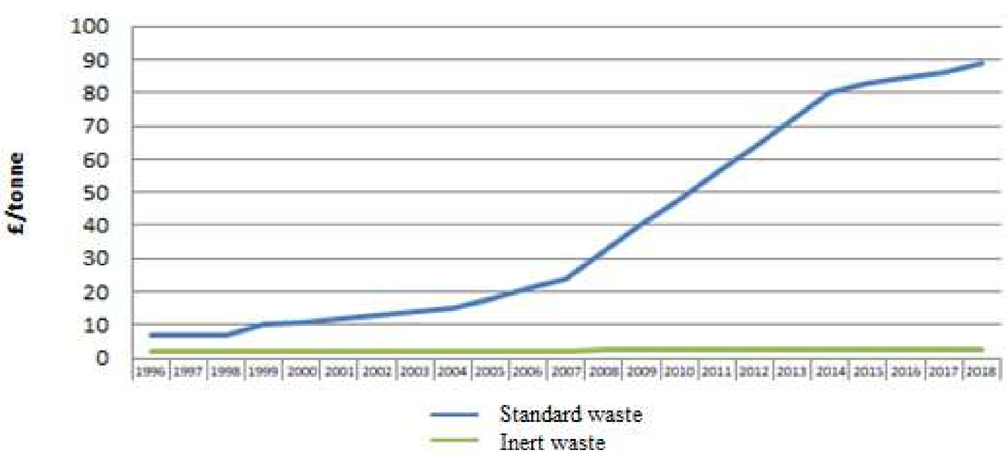


Figure 1-6: Landfill tax rates from 1996 to 2018 in the UK (360 Environmental, 2018).

Across the world, urbanisation is increasing. In 2012, 3.5 billion people lived in urban areas, and this is expected to increase to 6 billion people by 2050, with 50% of this growth occurring in developing countries (Curry and Pillay, 2012). Therefore, the issue of organic waste is a pressing concern that is likely to increase in severity in the future, particularly in urban environments.

1.7 Aims and objectives

The aim of this thesis was to make advances in the area of micro-scale anaerobic digestion of food waste by the following streams of investigation:

Experimental study: Variable-rate feeding for anaerobic digesters

The goal of this work was to investigate the effect of a variable rate of feed on the anaerobic digestion process. To achieve this, a dual-stream automated lab-scale AD system was designed and constructed, and then operated for a period of 9 months under different feeding regimes. The system was monitored for effects on alkalinity, biogas methane content and biogas production to show stability and volatile solids destruction, biological methane potential and magnitude of feed response to show performance. These data were gathered by both automatic readings and offline sampling and testing.

The overall aim was to gather information about the flexibility of the AD plant, and whether this flexibility was improved by the application of a variable feed regime.

Case study: Micro-scale AD in London, UK. A case study of a micro-scale AD plant in the urban environment.

The goal of the study was to present an account of the operation of a micro-scale anaerobic digester in context and show the effect of a variable feed in terms of the stability, indicated by the alkalinity and ammonia content, and the performance, indicated by the biogas production rate and methane content. The subject of the case study was a demonstration micro-scale AD plant in London, UK, that operated from 2013 to 2019, with data for analysis of the system collected for 319 days in 2014.

The case study also gathered and collected information on the energy use and production in the plant and the income and expenditure. This data was gathered with a goal of presenting a brief techno-economic analysis of the plant, to be used and expanded further in the desktop TEA study.

Desktop study/modelling: Micro-scale AD techno-economic analysis

The goal of this work was to produce a study of the feasibility of an anaerobic digestion plant in a micro-scale setting (at a university), and compare different scenarios to find the most viable, in terms of economics, safety and quality of output.

An assessment was made of the inputs, outputs and scope of the system and a model of the system was constructed. Different elements of the system were then added, modified or removed to find the most suitable scenario.

The model was further used to determine the cost factors that had the greatest influence on the payback time, with the aim of describing the optimal design for an AD plant of this type and scale, and providing a sensitivity analysis that could be used to inform the development of similar systems in the future.

1.8 Thesis structure

The subsequent content of this thesis contains the following sections:

Chapter 2: Literature review: Information that was important to this study from other researchers, about the AD process, plant design, food waste and micro-scale AD.

Chapter 3: Methodology: Design of an automatically controlled laboratory-scale AD plant: description of the design, build and commissioning process. Methods and techniques used in the analysis of the practical laboratory work. Experimental setup for the laboratory work, including any choices made regarding feedstock. Methodology of the process modelling and the techno-economic analysis.

Chapter 4: Flexible feeding of an anaerobic digester: Experimental study of the flexibility of biogas production that can be achieved in the AD plant and how variable feeding affects the stability of the plant.

Chapter 5: Case study: A case study of micro-scale AD plant in London, UK. This chapter was published as a paper in January 2017. The details of the publication are as follows:

> WALKER, M., THEAKER, H., YAMAN, R., POGGIO, D., NIMMO, W., BYWATER, A., BLANCH, G. & POURKASHANIAN, M. 2017. Assessment of micro-scale anaerobic digestion for management of urban organic waste: A case study in London, UK. *Waste Management,* 61**,** 258-268.

Chapter 6: Techno-economic analysis: A mass balance, model and study of the technical and economic aspects of a (theoretical) micro-scale anaerobic digestion plant at a university.

Chapter 7: Conclusions: A summary of conclusions drawn from the thesis.

Chapter 8: Discussion: A discussion of the conclusions in the broader context and further work.

Appendix A: Mass balance calculations: Full details of the equations used in the mass balance for the laboratory rig.

Appendix B: Techno-economic analysis calculations: Full details of the equations used in the TEA, with details and references of any data assumptions that were made.

2 Literature review

2.1 Micro-scale anaerobic digestion

2.1.1 The anaerobic digestion process

Anaerobic digestion (AD) is the decomposition of organic matter without the presence of oxygen. It is brought about by a consortia of bacteria working symbiotically in a series of interconnected biochemical reactions that systematically reduce the organic matter from large complex molecules into smaller molecules. Anaerobic digestion is a food chain, where each species depends on the proper operation of processes by the species before and after it in the sequence to be able to operate effectively itself (figure 2-1).

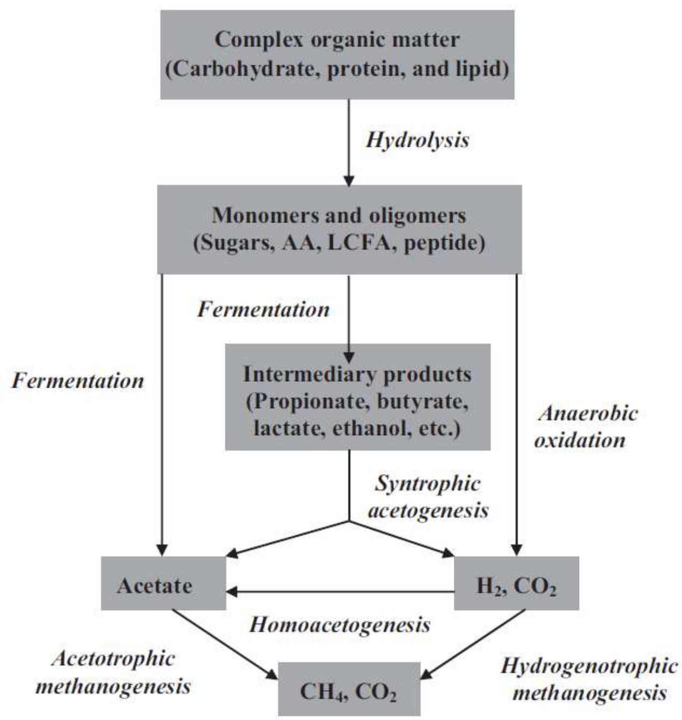


Figure 2-1: Major stages of the anaerobic digestion process (AA = amino acids; LCFA = long chain fatty acids) (Surendra et al., 2014).

There are four major 'stages' of the anaerobic digestion process. Firstly, the process of hydrolysis breaks complex substrates such as carbohydrates, lipids and proteins into simpler substrates such as sugars, fatty acids, alcohols and amino acids. These then undergo further breakdown via acidogenesis (fermentation in figure 2-1) into smaller substrates such as volatile fatty acids (acetate, propionate, butyrate, iso-butyrate, valerate), hydrogen and carbon dioxide. The longer-chain acids that were formed are similarly broken down further into carbon dioxide, hydrogen and acetate by the process of acetogenesis. The final stage is methanogenesis, in which acetate, carbon dioxide and hydrogen are used by a number of species of methanogens to synthesise methane (Gerardi, 2003a).

Each stage of the anaerobic digestion process is characterised by a reaction rate, and the first and final stages, hydrolysis and methanogenesis, are known to have a slower rate than the middle stages, acidogenesis and acetogenesis (Mata-Alvarez, Macé and Llabrés, 2000). A build-up of intermediates in the process is a useful indication of instability as it shows that the different reactions are not balanced properly.

2.1.2 What is micro-scale AD?

The process of AD is the same at any scale. The size of anaerobic digestion plants is normally defined by the power rating of the generator that could be run from the maximum output (in kWe), or sometimes by the mass of feed added per day or per year, in kg or tonnes, but can also be rated by the working volume of the digester. Micro-scale AD has no standard definition of size.

Micro-scale AD as described in the literature normally ranges from household-size to the size of a large institution such as a university (Chanakya, Sharma and Ramachandra, 2009; Hou *et al.*, 2017; BRE/WRAP, 2013), processing up to 1 tonne of waste per day, which would produce up to about 10 kWe. The UK government classification system groups together all AD plants under 250 kWe as 'small AD'. However, there are large differences in the economics, design and operation when comparing a 10 kWe to a 250 kWe AD plant, so there is potentially an argument that an extra 'micro' size category should be added.

In one reference, a micro-scale plant is defined as having a feed input of 100 kg to 1 tonne of fresh matter per day, which for the type of waste added in this paper (OFMSW) would translate as 0.7-7 kWe (Chanakya, Sharma and Ramachandra, 2009). In discussing micro-scale AD, this paper adds a context – decentralised treatment of waste at an 'institutional' level. A feasibility study by BRE/WRAP of a micro-AD plant (BRE/WRAP, 2013) showed the input as up to 138 tonnes/year or 0.38 tonnes/day. An earlier paper supports this, giving its definition of micro-scale AD as under 5 kW (Ackermann, Andersson and Söder, 2001). Given that an average household in the UK uses an average of 0.44 kW (GOV.UK, 2014a), a micro-scale AD plant would be one suitable for supplying electricity to 1-15 households. However, when micro-scale AD is reported on in more commercial contexts, the range of size is generally larger, for example 'less than 50 kWe' (Savills UK, 2017), and 'household-scale (6 litres of food waste per day) to 80kWe' (Biogas World, 2017). This is possibly because the commercial world does not yet view micro-scale AD as a viable proposal in terms of economics. These references are summarised in table 2-1.

Table 2-1: Summary of micro-scale AD size definitions

Size (feed in tonnes/day)	Size (kWe)	Reference
0.1 to 1 tonne/day	0.7-7 kWe	(Chanakya, Sharma and Ramachandra, 2009)
0.38 tonne/day	5.74 kWe	(BRE/WRAP, 2013)
	< 5kWe	(Ackermann, Andersson and Söder, 2001)
	<50 kWe	(Savills UK, 2017)
	<80 kWe	(Biogas World, 2017)

The Renewable Heat Incentive (RHI) offered by the UK government separates AD sizes into <200 kWth, 200 to 600 kWth and 600 kWth and above small, medium and large plants respectively (GOV.UK, 2019b).

2.1.3 Micro-scale AD in the developed world

There is very little discussion of AD specifically at a micro-scale for the developed world in peer-reviewed literature, although there is an interest in developing this technology (Decisive 2020, 2018; BRE/WRAP, 2013). There could be several reasons for this lack of research:

- The biological process for any scale of AD is the same, and research is generally focussed on the process rather than the application of the technology

- The technology is still at very early stages and therefore has not generated research interest yet

- Micro-scale AD is generally understood to be not as profitable or easy to implement as larger-scale AD, and the lack of commercial interest means that it is a less attractive proposition for research

A few small projects have been developed or investigated and reported upon (Curry and Pillay, 2012; Walker *et al.*, 2017; The Waste Transformers, 2016; Curry, 2015). There also exist a number of companies developing micro-scale off the shelf solutions (Moran, 2017; Qube Renewables, 2017).

A case study for small-scale anaerobic digester design in Canada gave details of the amount of waste that the digester would process, the system design, safety considerations, the amount of biogas and energy that would be produced, and capital cost of the system (Curry and Pillay, 2012). This paper provided a useful outline for the design of anaerobic digestion technology at this scale. The design was a two-stage system where hydrolysis was separated from the rest of the AD process in a primary tank, and the hydrolysate was then fed into a secondary tank. A second paper by the same author (Curry, 2015) expanded on the design of the system by adding heating from an air source heat pump located in a greenhouse, and found that the digester could be heated from the greenhouse, heat which would otherwise have been made by burning 15% of the biogas produced by the digester. However, this paper did not give details of calculations for the amount of energy used to run the heat pump and so it is difficult to conclude whether an energy saving was made.

A later paper (Walker *et al.*, 2017) described an operational micro-scale AD plant in London, UK. The digester was a 0.37 kW plant with a CSTR-type digester, processing 5.23 tonnes per annum of food waste. The paper included almost a year of operational data, key performance indicators, an energy balance, emissions savings and a full predicted and actual CAPEX and OPEX costing for the project. The designers were able to capitalise on the small footprint of the plant by situating it in a greenhouse, which reduced the heating requirements of the plant by 49%. The paper highlighted critical areas in which efficiency made a large difference to the energy requirements of the plant (for example, a high proportion of the total energy to the plant was consumed by the monitoring system; in larger plants this is a fraction of the total energy use). The paper also provided a number of observations on the difficulties encountered that were specific to the size and location of the digester, and therefore acted as a useful reference case study for future projects.

A project in operation in Amsterdam, Netherlands (The Waste Transformers, 2016) collects waste from a group of commercial properties in a retail and entertainment park. The website provides details of waste collection and the operation of the plant and shows its size (about 3 m^3) but does not give details about how much energy is produced or how the biogas is used. The project emphasises the advantage of processing waste 'on site', so that the plant is visible to its users, encouraging users to be responsible for their organic waste and demonstrating its value.

A feasibility study (BRE/WRAP, 2013) provided a design and costing for a potential micro-scale AD plant in Watford, UK. The study highlighted some advantages and disadvantages of urban micro-scale systems and provided a decision-making framework for future projects. The authors conclude that the cost-benefit of the installation could be improved if the plant was run as a practical demonstration and possibly by applying AD in parallel with other renewable technologies or additional thermal storage to capitalise on the heat produced.

In summary, there is certainly interest in implementing anaerobic digestion at a micro- scale, but it has not yet gained the universal acceptance and understanding that has been achieved by other renewable technologies or by large-scale anaerobic digestion.

2.1.4 Micro-scale AD in the developing world

There are many more installations of micro-scale AD in developing countries (Bond and Templeton, 2011), and consequently a lot more research is available. Two examples are a floating cover digester (figure 2-2) and a bag digester (figure 2-3). The floating cover and bag digesters are very commonly used in rural China and India (Gunnerson and Stuckey, 1986; Surendra *et al.*, 2014).

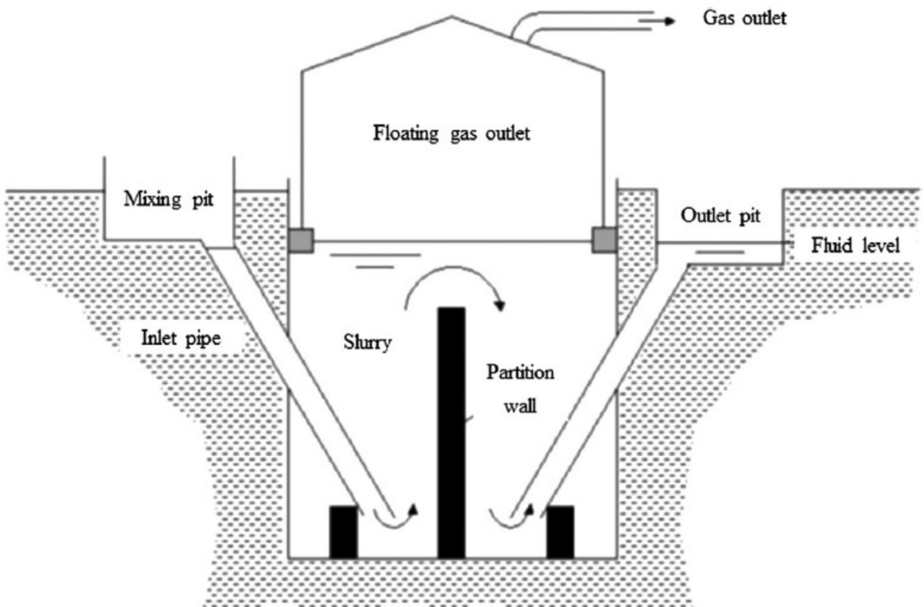


Figure 2-2: Diagram of a floating cover digester design (Surendra et al., 2014)

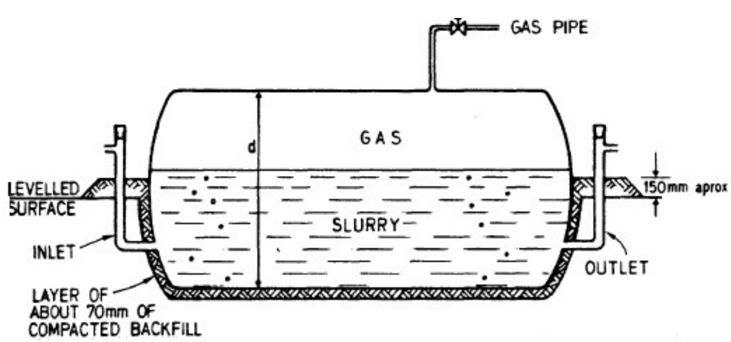


Figure 2-3: Diagram of a bag digester (Gunnerson and Stuckey, 1986).

The digesters that are commonly built in the developing world are usually household-scale digesters (Surendra *et al.*, 2014). The digester size is given as of 5-10 m^3 (Hamad, Abdel Dayem and El Halwagi, 1981), and similarly in a later paper as 2-10 m^3 (Surendra *et al.*, 2014). The design is shown similarly in a range of older and newer sources of literature, indicating that there has not been a great deal of development in the technology (Surendra *et al.*, 2014; Gunnerson and Stuckey, 1986; Hou *et al.*, 2017). The digester would use feedstocks such as animal manure, kitchen waste and human waste (Singh and Kaushal, 2016; SSWM, 2017). The bag digester is built on a slope (2-5%) to create a very slow 'flow' from the inlet to the outlet (Singh and Kaushal, 2016).

Compared to micro-scale AD in developed countries, the technology is much simpler, with no control systems, mixing or heating (Surendra *et al.*, 2014), so there is little engineering-based research of direct relevance to this thesis. However, there are operational lessons to note. These digesters suffer from a high failure rate, with the average amount of functional digesters being around 50% in a given region when they were revisited after several years of operation (Bond and Templeton, 2011). However, a 100% success rate in the Sirsi region of India was reported in the same paper. This was attributed to a competitive market in which companies provided maintenance for the digesters and the area was known to have a high literacy rate (Bhat, Chanakya and Ravindranath, 2001). The relevance to the developed world would be that expert intervention, maintenance and training are therefore essential to a plant's functional and economic success.

2.2 Variable biogas production

Anaerobic digestion is unusual among renewable energy technologies in that it produces a fuel that can be stored (Szarka *et al.*, 2013). Additionally, the operator can control the biogas production from a plant by increasing or decreasing the input rate of the feedstock. Using storage and feed variation, AD plants can be viewed as a flexible source of energy (figure 2-4).

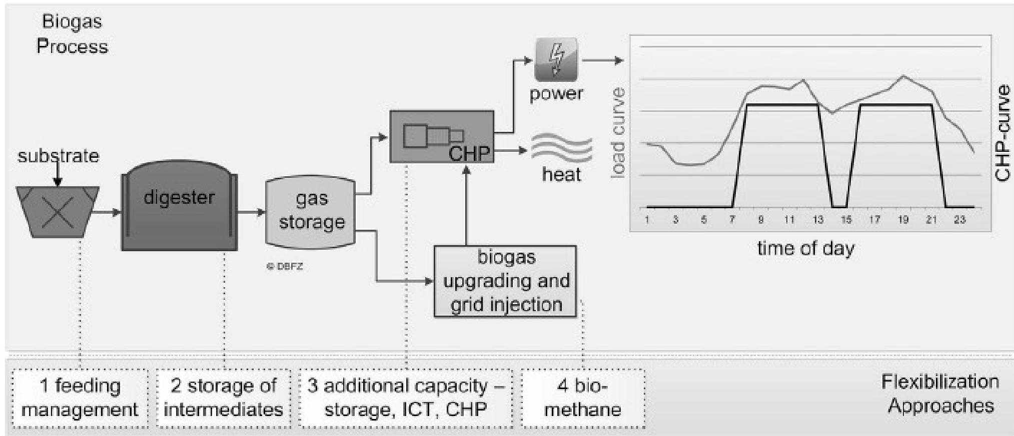


Figure 2-4: Different technical approaches for increasing biogas plant flexibility. CHP = combined heat and power unit, ICT = information and computer technology (Szarka et al., 2013).

The share of renewables in the electricity market in the UK was 29.7% at the end of 2017 (GOV.UK, 2018a), and the market share has been growing each year (table 2-2).

Table 2-2: Percentages of electricity derived from renewable sources (GOV.UK, 2017a)

	2012	2013	2014	2015	2016
International Basis (1)	11.3	14.9	19.1	24.6	24.5
Renewable Obligation (2)	12	15.5	19.8	26.1	26.2
2009 Renewable Energy Directive (3)	10.8	13.8	17.8	22.3	24.6

1 All renewable electricity as a percentage of total UK electricity generation
2 Measured as a percentage of UK electricity sales
3 2009 Renewable Energy Directive measured as a percentage of gross electricity consumption

As the market share of renewable energy sources grows, the ability to control the output of a renewable energy source is becoming more important. This is because it can help to ensure that the energy supply stays constant when intermittent weather-dependent renewable energy sources such as solar and wind are installed. This assertion is supported by the introduction of 'premium' rates in Germany for energy produced at peak times (Hahn *et al.*, 2014b; Szarka *et al.*, 2013). Flexible energy production can also reduce the total cost of a country's power supply system, as it reduces the need for storage (Lauer and Thrän, 2018).

2.2.1 How should flexibility be quantified?

A review of concepts surrounding the use of AD to match demand for electricity in Germany (Hahn *et al.*, 2014b) defined flexibility in terms of the response time that can be achieved, with three classifications: primary (under 5 minutes), secondary (5-15 minutes) and tertiary (up to four hours). A second reference from the viewpoint of national energy planning (Papaefthymiou, Grave and Dragoon, 2014) defined short-term (minute by minute), medium-term (hourly and daily) and long-term (yearly) flexibility and presented the features of a power plant that would satisfy each type.

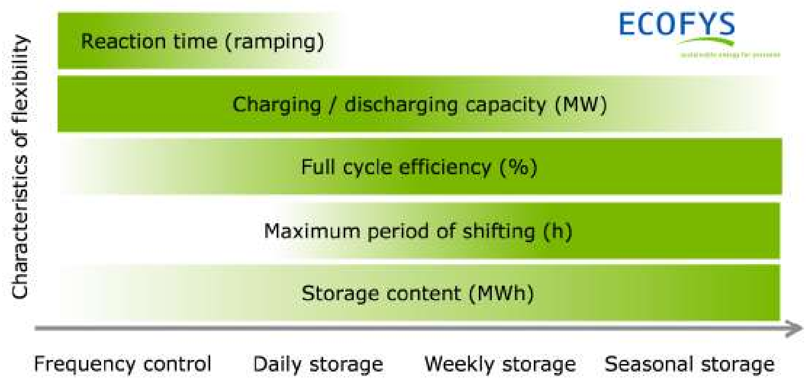


Figure 2-5: Basic characteristics of energy storage for providing flexibility and their significance in the different timeframes (Papaefthymiou, Grave and Dragoon, 2014).

A similar study presented the options available to provide flexibility in energy production, such as ramping up and down the CHP output, increasing biogas storage, or moderating the feed input, which have a different response times (Thrän *et al.*, 2015).

The interest in flexibility of energy production from biogas is relatively recent, with most papers being produced in the last 5 years. In earlier papers, flexibility is quantified by the percentage variation achieved (Mauky *et al.*, 2015; Laperrière *et al.*, 2017), which is relatively simplistic and does not provide a lot of scope for comparison between plants. A recent paper (Dotzauer *et al.*, 2018) looked into this question in greater detail and produced parameters by which the flexibility of a plant can be quantified, in terms of the profiles of two different outputs; the power generation of the plant and the biogas output (figure 2-6 and figure 2-7).

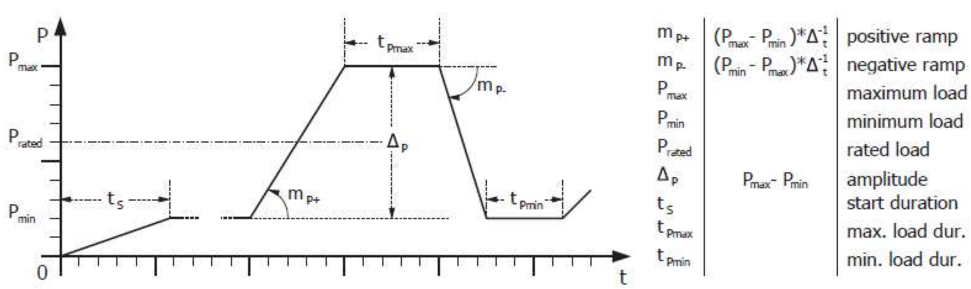


Figure 2-6: Indicators for flexible power generation by biogas plants (Dotzauer et al., 2018).

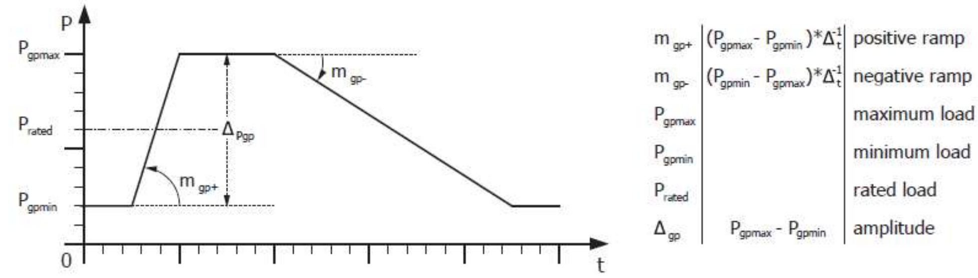


Figure 2-7: Indicators for flexibility of biogas production (Dotzauer et al., 2018).

This paper applies the framework to a number of reference plants and quantifies their biogas production and power generation flexibility. The authors note that the flexibility in power generation is not strictly linked to biogas production flexibility, because most AD plants include a 'buffer' in the form of a biogas holder. They also note that power generation flexibility in an AD plant gives faster-response, short-term flexibility, and the variation in biogas production can support this by providing longer-term flexibility. The most effective system of operating AD plants to respond to energy demand will incorporate whichever of these strategies best suits the needs of the situation – or both strategies at once.

2.2.2 Varying feedstock type for flexibility

'Variable substrate feeding' was studied, where the biogas production rate from different feedstocks was determined and used to control the biogas output (Hahn *et al.*, 2014b). The study tested corn, rye, beet, and cattle manure, and mapped the gas production from each (figure 2-8).

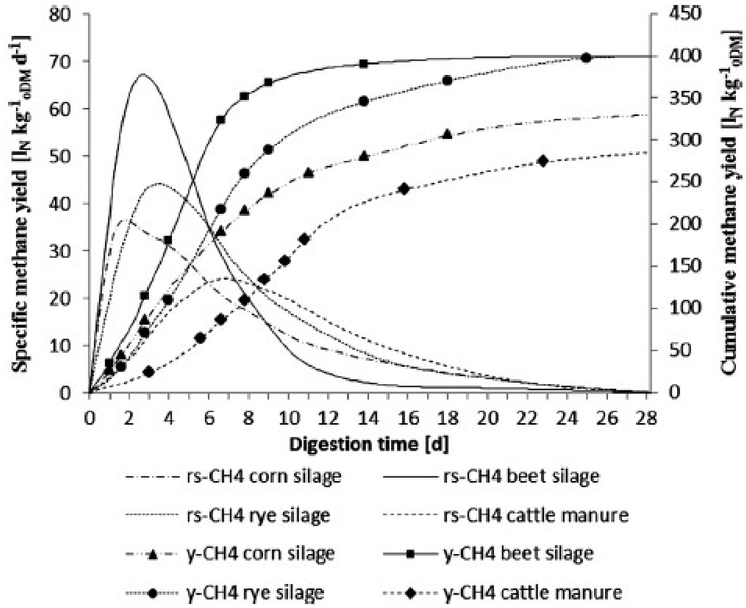


Figure 2-8: Specific methane yield and cumulative methane yield from the digestion of cattle manure, corn, rye and beet silage in batch experiments under mesophilic process conditions over a digestion time of 28 days (Hahn et al., 2014b).

The paper quantified the different rates of biogas production from different feedstocks. Although not tested, the paper cited glycerol as a very fast-degrading substrate, with a degradation time of hours as opposed to days for maize or grass silage. Drawing on these results, the paper presented a model simulation of a potential pattern on flexible biogas production over the time span of a week (figure 2-9).

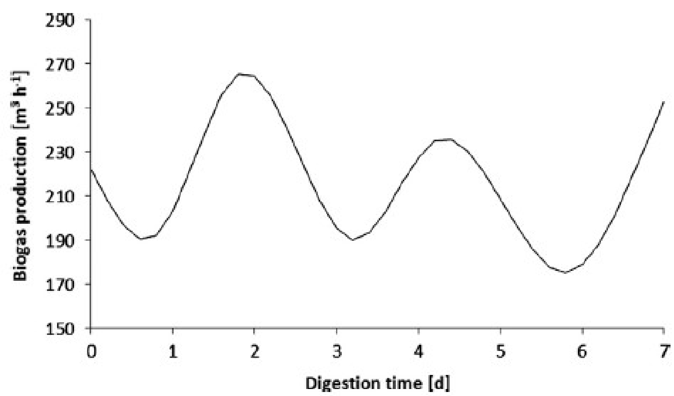


Figure 2-9: Simulated flexible biogas production during 1 week achieved by a variable feeding of manure, maize silage and shredded wheat of a mesophilic driven biogas plant with 500 kW of installed electrical baseload capacity (Hahn et al., 2014b).

This type of flexible biogas production would be classed as a tertiary (above four hours) response time (Hahn *et al.*, 2014b). This is limited in its usefulness but could provide longer-term, predictable variations, such as the variation in daily or seasonal energy use. A similar study found that flexible biogas production over the course of a day could be achieved by finding the degradation rate of different feedstocks (cattle slurry, maize and sugar beet) and designing the feeding pattern to generate biogas according to a demand (Mauky *et al.*, 2015). The results showed that production rate could be increased or decreased by a factor of four over two hours.

Another possibility for adding flexibility to biogas production by modifying the feedstock is to allow the feedstock to degrade (and therefore produce VFAs) before it is fed into the plant (Aichinger *et al.*, 2015). Similarly, disintegration techniques were investigated as a way of increasing flexibility in gas production (Hahn *et al.*, 2014b), and found that the techniques did result in a faster biogas production. A comparison of flexible biogas production with both disintegrated and non-disintegrated feedstocks might be a useful area of further investigation.

2.2.3 Varying feedstock input rate for flexibility

Academic studies of variable feeding patterns for anaerobic digesters have used an organic loading rate (OLR) of between 1 and 11 kgVS m^{-3} day^{-1} and 1 to 20 g COD m^{-3} day^{-1} with a variety of different loading patterns (table 2-3).

Table 2-3: Loading rates and patterns in studies of variable feeding of anaerobic digesters.

Reference	Loading pattern	Digester and feedstock
(Lemmer and Krümpel, 2017)	2 peaks over 24 hours – in 2 separate peaks or following the diurnal pattern of electricity use. Feeding 2 to 20 g COD L^{-1} day^{-1} OLR.	Anaerobic filter, fed with hydrolysate from maize- and grass-fed leach bed.
(Mauky *et al.*, 2015)	1.0 to 7.0 kgVS m^{-3} day^{-1} varied over the course of 300 days. Different regimes, between 1 and 7 feedings per day.	CSTRs, with substrates maize, sugar beet, cattle slurry, digestate from primary digester.
(Mauky *et al.*, 2016)	OLR 2.8-3.5 kgVS m^{-3} day^{-1} and 4.0 kgVS m^{-3} day^{-1} over 20 days. 5 feeds over 12 hours then no feed for 12 hours.	Operational plants (CSTRs) fed with cattle manure, maize silage, ground wheat grain, grass silage
(De Vrieze, Verstraete and Boon, 2013)	1 g L^{-1} day^{-1} for 24 days' 'start-up', then daily or every 2 days feeding. Stress test – feeding at OLR 2, 4, 6, 8 g L^{-1} day^{-1} over 4 days	CSTR fed on synthetic raw domestic sewage (SYNTHES)
(Laperrière *et al.*, 2017)	Base load of 1.5 or 2.5 kgVS m^{-3} day^{-1} then increase to 3 to 5.5 kgVS m^{-3} day^{-1} for 1 day	CSTR. Flexibility measured by gas production.
(Mauky *et al.*, 2017)	(1) 2 to 5 kgVS m^{-3} day^{-1}, 5 feeds over 12 hours then no feed for 12 hours	(1) CSTRs, with substrates maize, sugar beet, cattle slurry, digestate from primary digester.
	(2) 2 to 4 gVS L^{-1} day^{-1}, 5 pulses over 12 hours then no feed for 12 hours	(2) Operational plants (CSTRs) fed with cattle

		manure, maize silage, ground wheat grain, grass silage
(Mulat *et al.*, 2016a)	4 phases, phase 1-3 4 kgVS m^{-3} day^{-1}, phase 4 between 5 and 11 kgVS m^{-3} day^{-1}. Fed daily, every 2 hours or every second day	Two CSTRs, fed with distiller's dried grains with solubles.
(Lv *et al.*, 2014b)	4 kg VS m^{-3} day^{-1} for all digesters throughout. Fed once a day or twice a day.	Four CSTRs, fed with maize silage.

A model for flexible biogas production was described, which was able to predict the biogas production rate with only a 4-9% discrepancy against the observed biogas production rate (Mauky *et al.*, 2016). The model was applied to two industrial digesters, 150 m^3 (digester A) and 923 m^3 (digester B) in size, being fed in a variable pattern with cow slurry, maize, grass and wheat grain. A typical example of the intermittent feeding pattern that was applied to the digesters is shown in graph (a) of figure 2-10.

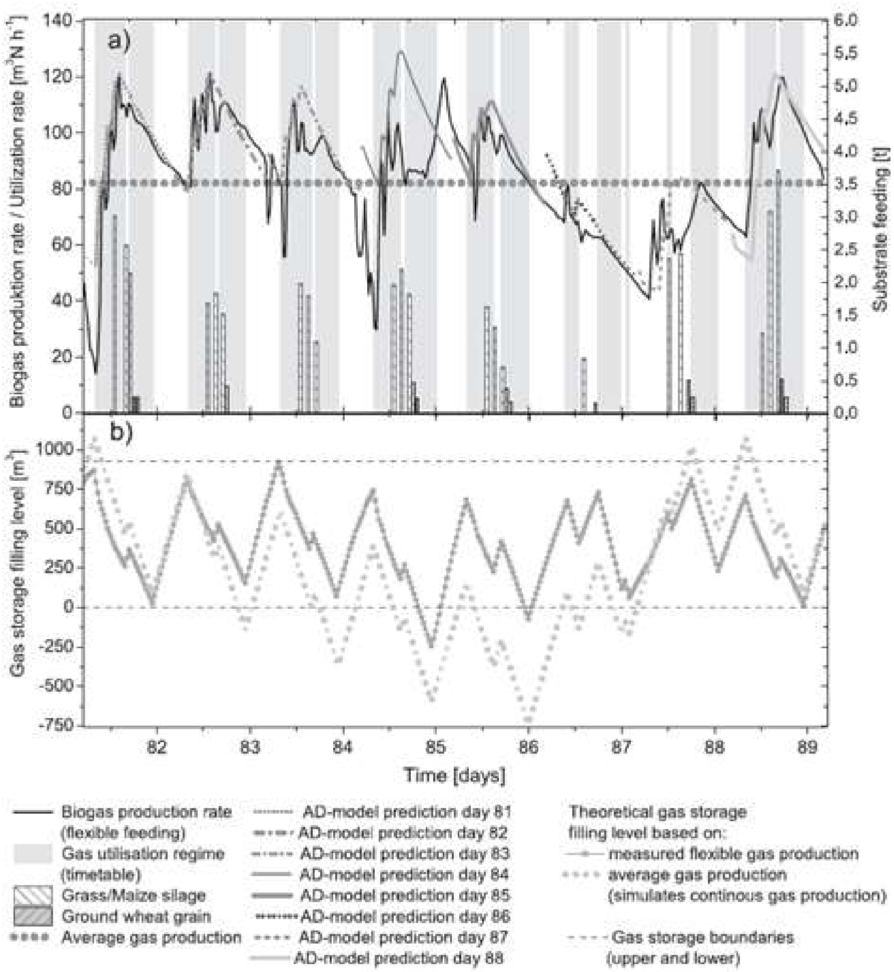


Figure 2-10: Flexible biogas production at research biogas plant B in experimental week 2. (a) Measured gas production rate of one week compared with the daily forecasts within the model predictive control of the particular day (b) Theoretical gas storage filling level based on flexible and continuous gas production (Mauky et al., 2016).

The variation in biogas production achieved by these digesters by intermittent feeding was 20% to 130% of the biogas production that would be achieved (according to the model) with feeding the same amount steadily. By feeding intermittently, with a view to supplying biogas when it is to be used, the authors estimated that the biogas storage requirement was reduced in size by 42% (digester A) and 45% (digester B) compared to steady biogas production.

2.2.4 Organic loading rates and biogas production

A variable biogas production study (Laperrière *et al.*, 2017) ran two laboratory-scale digesters fed with carrots and grass silage at an organic loading rate (OLR) of 1.5 gVS L^{-1} day^{-1} and then 2.5 gVS L^{-1} day^{-1}, then increased the OLR by 2 to 4 times in one of the digesters (figure 2-11).

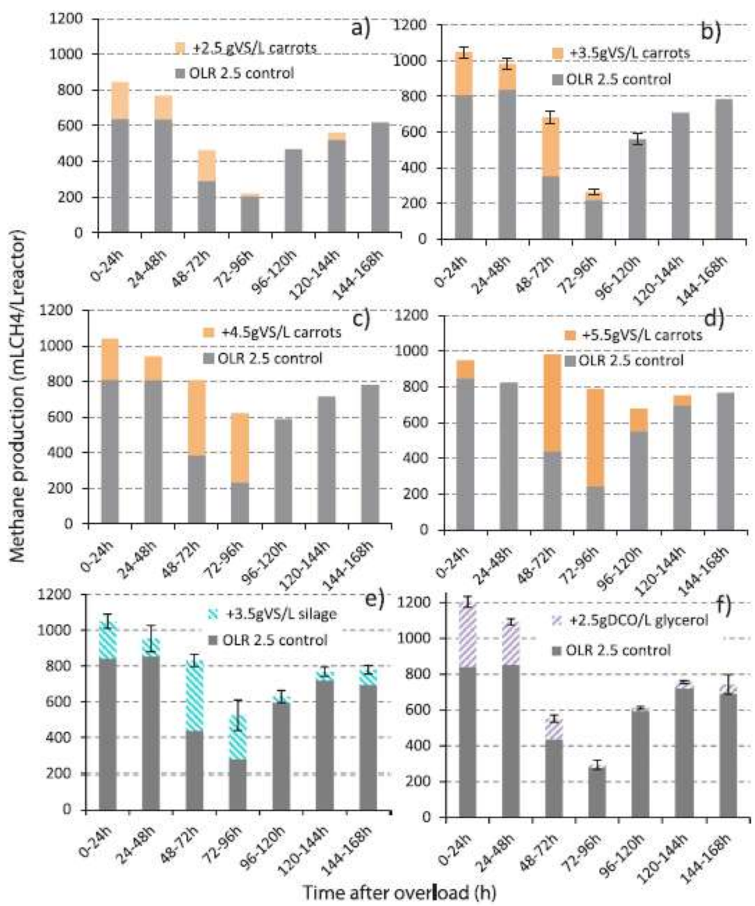


Figure 2-11: Comparison between methane overload production and control production on 25 gVS/L.day baseload with a) +2.5g VS/ overload b) +3.5 gVS/L overload c) +4.5 gVS/L overload and d) +5.5 gVS/L overload with carrots; e) overload of +3.5 gVS/L with maize silage and f) overload of +2.5 gVS/L with glycerol (Laperrière et al., 2017)

The test digester showed an increase of biogas production of between 18% and 180% compared to the control digester. The production increase was greatest when the digester started from the lower 'baseload' OLR of 1.5 gVS L^{-1} day^{-1}. The conclusion was that the digester starting from a lower OLR was not as close to the maximum capacity as it was when starting from a higher OLR, and therefore the digester could accommodate a greater increase in feed. Also reported on was the amount of time that the digester showed an 'overproduction' of biogas as a result of increasing the feed. The paper showed that the overproduction of biogas lasted for approximately 120 hours, with most of the overproduction occurring within about 72 hours of the increased feed (figure 2-11). This shows the responsiveness of the digester to changes in feed with different substrates.

The paper showed that glycerol had the fastest degradation kinetics, with an increased biogas production rate for the shortest length of time after the overfeeding incidents, with the overproduction occurring mostly in the first 48 hours. Finally the paper noted that at a higher baseload, the biogas overproduction period was slightly delayed when the overload was highest (graph (d) of figure 2-11), indicating a possible inhibition due to the overloading, however the authors did not support this theory with VFA testing.

2.2.5 Flexible biogas production using two-stage digestion

Flexibility in an AD plant can be achieved by separating the system into a two-stage process, although it is less favoured in industry as it is more complex than single-stage digester and has higher operating and investment costs (De Gioannis *et al.*, 2017; Mohan and Bindhu, 2008).

A 2014 paper presented two versions of a two-stage design (Hahn *et al.*, 2014b).

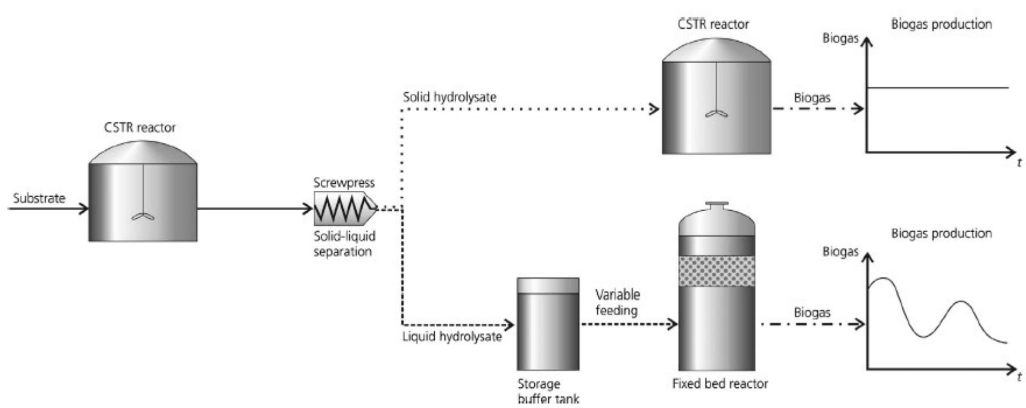


Figure 2-12: Flow chart of the ReBi biogas plant configuration for a flexible biogas production (Hahn et al., 2014b).

In the first design (figure 2-12), the feed is broken down hydrolysis and acidogenesis in a stirred-tank reactor, then separated. The liquid fraction is fed into a fixed-bed reactor, and the solid fraction is fed into a CSTR reactor to undergo acetogenesis and methanogenesis. The results show an approximately 20-fold increase of volumetric methane yield over a four-hour period (figure 2-13).

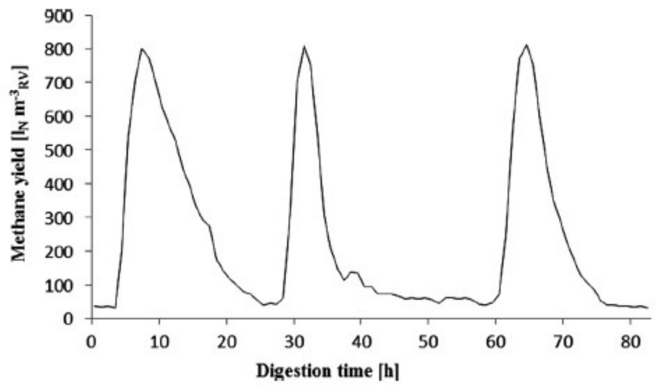


Figure 2-13: Flexible biogas production from the ReBi biogas plant under mesophilic process conditions through a variable feeding of press fluid from maize silage into a fixed bed reactor over a digestion time of 80 hours (Hahn et al., 2014b) (RV=Reactor volume).

A second two-stage configuration discussed by the paper is the leach bed – fixed bed hybrid system (figure 2-14).

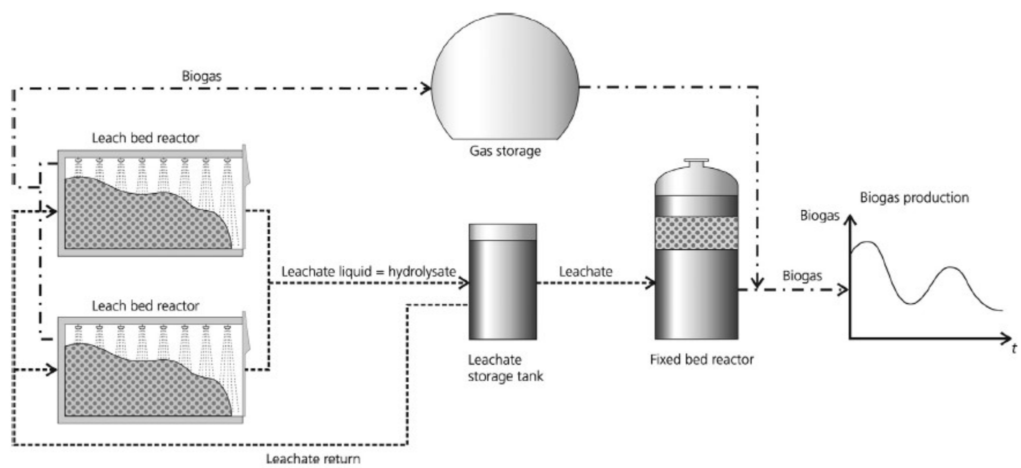


Figure 2-14: Flow chart of the double stage leach bed/fixed bed biogas plant configuration for a flexible biogas production (Hahn et al., 2014b).

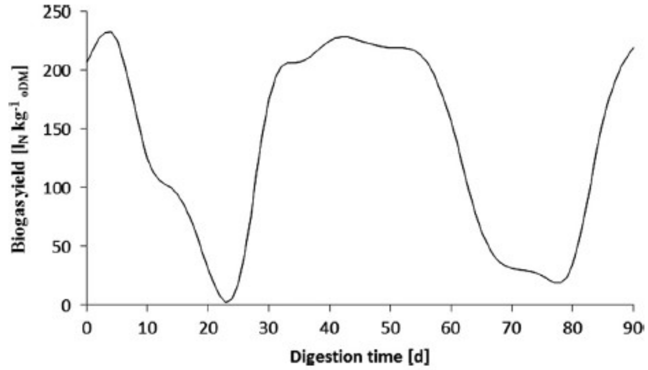


Figure 2-15: Flexible biogas production with the leach bed/fixed bed mesophilic driven biogas plant configuration achieved by a variable feeding of leachate from maize silage into the fixed bed reactor(Hahn et al., 2014b).

In this system, only the liquid generated from the first stage is used for biogas production. This configuration shows a slower ramp-up time – on the scale of days rather than hours. The two designs would make a similar liquid product so the slower digestion time is inconsistent. The variation in biogas yield in the second example is much greater, with the digester production being stopped at one point.

Flexible feeding using an anaerobic filter reactor was investigated (Lemmer and Krümpel, 2017), in which two different hydrolysates from leach bed reactors (substrates A and B) were fed in different patterns over the course of a day into an anaerobic filter. The study tested two different day-long feeding patterns: a 'demand' pattern similar to the normal grid demand (figure 2-16), and a 'peak' pattern with a low level of feeding interrupted by two 3-hour long sections of continuous high level of feeding (figure 2-17).

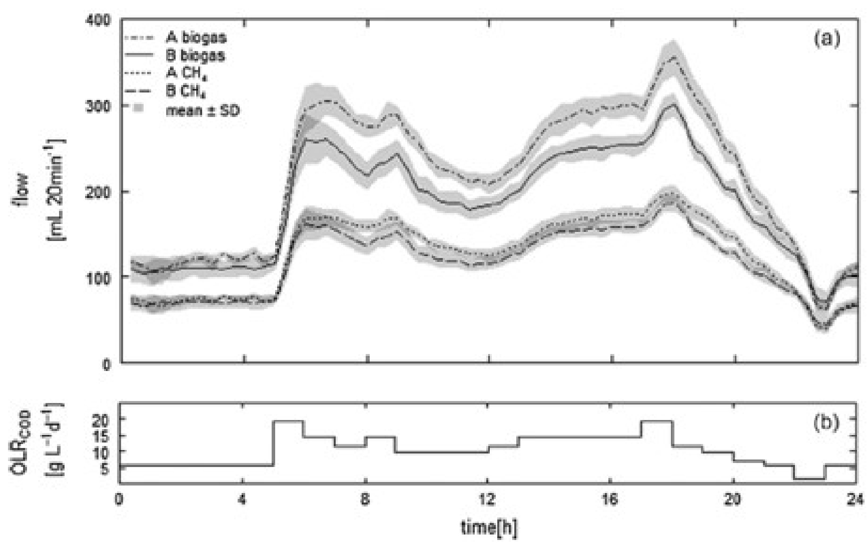


Figure 2-16: Compiled data of OLR-mode 'demand' for both substrates A and B (a) daily gas and methane production, (b) applied OLR (Lemmer and Krümpel, 2017).

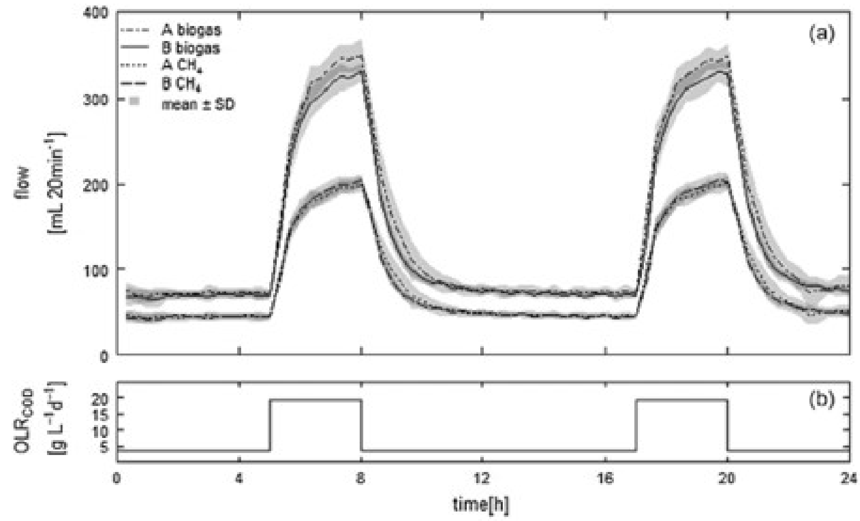


Figure 2-17: Compiled data of OLR-mode 'peak' for both substrates A and B (a) daily gas and methane production, (b) rate of increase in methane production r_{CH4}, (b) applied OLR (Lemmer and Krümpel, 2017)

The biogas production in both cases closely followed the feed pattern, with a response within minutes, indicating that an anaerobic filter is a suitable technology for flexible biogas production, either for peaks of demand or a diurnal 'predictable' demand.

2.2.6 Digester stability with flexible feeding

Dynamic feeding was investigated at both laboratory and full scale (digesters with a working volume of 165m^3 and 800m^3), using an OLR of 2 to 5 kgVS m^{-3} day^{-1} (Mauky *et al.*, 2017). The authors found that the stability was affected by the changes, but none of the plants became 'dangerously' unstable (they were not at risk of failing). In a study of flexible feeding in an anaerobic filter digester, it was also found that the stability of the digesters was not affected by the varying feed pattern (Lemmer and Krümpel, 2017).

The effect of intermittent feeding on the stability of a digester and the diversity of its microbial community was described by De Vrieze et al (2013). This paper compared two laboratory-scale digesters, one fed daily (A) and the second fed every two days (B), both at an OLR of 1 gCOD L^{-1} day^{-1}. Digester A was classed as a 'stable-feeding' digester, and digester B was classed as a 'dynamic-feeding' digester. The paper found that digester B developed a more diverse microbial community, and was more able to resist stress tests (ammonia toxicity and overloading up to an OLR of 8 gCOD L^{-1} day^{-1}). The results of the overloading test (figure 2-18) show that the 'dynamic' digester produced more biogas compared to the 'stable' digester.

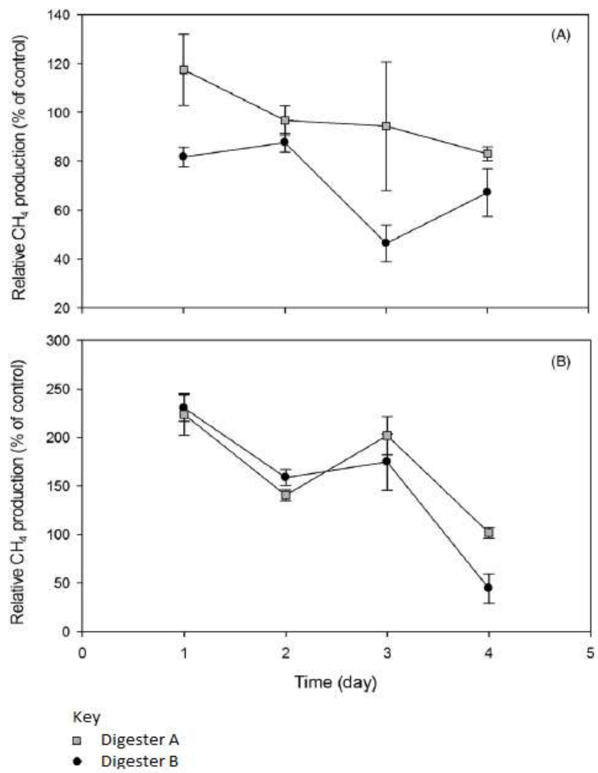


Figure 2-18: Results of the short-term stress tests in terms of the tolerance of the digesters to A) high concentrations of ammonium and B) elevated organic loading rate. Average values of the three replicates per treatment are represented together with the values of the standard deviations as error bars (De Vrieze, Verstraete and Boon, 2013).

Stability of the plants under variable conditions was reported, finding that the variation did not cause instability when the feed amounts were varied between 1.0 and 7.0 gVS L^{-1} day^{-1}, although the paper does not give a clear indication of for how long these 'upsets' were sustained (Mauky *et al.*, 2015). A more recent study reported the effect of discontinuous feeding (once a day) versus continuous feeding and found that a digester that had been intermittently fed was more resilient to overfeeding than a digester that was fed continuously (Bonk *et al.*, 2018). This paper examined the methanogen populations of the two digesters and found that in the discontinuously-fed digester, an unusually high population of *methanosarcina* developed, an archaea that is functional at high concentrations of acetic acid. This investigation explored further than previous studies, by ruling out experimentally the effect of increased pH, total microbial biomass concentration and bacterial (i.e. non-archaeal) populations as responsible for the improved performance, as there was no significant difference between the two digesters.

2.2.7 Economic effect of flexibility in biogas production

Flexibility of energy supply can be achieved in several ways; by managing the feed input, storing 'intermediates' of the process (i.e. operating a two-stage system), adding extra biogas storage, varying the loading rate, or by upgrading the gas and injecting it into the national gas grid (Szarka *et al.*, 2013). Each of these options has its own cost implication but has recently been made more economically viable in Germany by the introduction of a flexibility premium, in which energy has a higher value at times of peak demand (Szarka *et al.*, 2013; Hahn *et al.*, 2014a). An analysis of a number of plant configurations for medium to large-sized AD plants with flexible generation capabilities (Hahn *et al.*, 2014a) considered two demand scenarios; biogas demand for only 8 hours a day (scenario A), and biogas demand during weekdays only, with no demand at weekends (scenario B). The study found that for scenario A, the most cost-effective method was to add extra storage, but for scenario B, the best approach was 'flexible biogas construction configuration' – that is, a two-stage system. The two-stage system became even more profitable when the plant was linked with a technology called IFBB, or the Integrated solid Fuel and Biogas from Biomass technology, which produces a high-organics liquid feed as well as a solid fuel that can be burned, making the most of the fuel and achieving high efficiency of the system. This finding emphasises the importance of considering the design of the whole system, as this can increase profitability.

The extra investment required for flexible power generation is only economically advantageous if the plant owner will be paid a premium for peak power generation, and that the system is more profitable if it is larger scale (the scale considered is 0.6 MW to 2 MW) (Hochloff and Braun, 2014). The use of a premium rate was modelled in a hybrid CHP-biomethane upgrade system (O'Shea, Wall and Murphy, 2016) and found that the daily revenue could be increased by 52% in a system that operated a CHP for one hour at a peak time and then exported the biogas (upgraded to biomethane) for the remainder of the day.

A study in Germany modelled the best economic options for the future development of the AD industry in their country (Lauer and Thrän, 2018). The study found that if biogas plants were upgraded so that they could operate in a flexible way, AD would be a cost-effective alternative to other storage options.

2.3 Techno-economics of micro-scale anaerobic digestion

In the developed world, the profitability of an AD plant is normally the main factor that determines whether it will be built, and it is likely to be one of the main reasons that AD on a micro-scale is not widely implemented (Yaman, Theaker and Walker, 2017). The modelling and study of the financial aspects of a plant is therefore of direct relevance to micro-scale AD applications.

2.3.1 Methodology

Guidelines for conducting a TEA study have recently been published by the Global CO_2 Initiative (Zimmermann *et al.*, 2018) which present a simple methodology for the TEA process (figure 2-19).

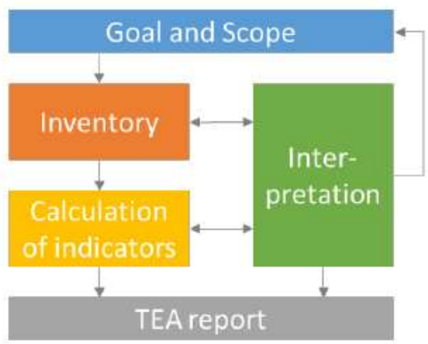


Figure 2-19: Phases of techno-economic assessment (Zimmermann et al., 2018).

This document was written to standardise the methodology for a TEA. It was written specifically for carbon capture and utilization (CCU) technologies, but can be applied across other technologies as it contains a full description and explanation of principles that are often used in other studies. For example, the concept of the functional unit and system boundary are also used in a separate study (Patterson *et al.*, 2011). The document is clear and comprehensive and draws from many other studies.

Anaerobic digestion was the subject of a life-cycle analysis (LCA) that included comparisons between centralised and distributed infrastructure (Patterson *et al.*, 2011). The study made convincing conclusions about the best use of the biogas created and the impact of a decentralised structure in terms of transport emissions. Although this study focused on minimisation of carbon emissions, it has useful principles for a techno-economic analysis. The process comprised:

1. Study and summary of previous studies.

2. Determination of the basic attributes of the system (such as how much waste it should process) and the system boundary.

3. Definition of inputs, outputs and indicators within the system,

4. Choice of which modelling software to use.

5. Running of the model under different scenarios to analyse the system and the impact of changing different parts of the system.

There are significant similarities between the process in this reference and the methodology in figure 2-19. A further TEA (Zamalloa *et al.*, 2011) supports these methodologies and describes clearly the TEA process, providing useful detail and following the same basic structure.

2.3.2 Feasibility studies

A feasibility study of an AD plant to process the output of the central food market in Barcelona (Mata-Alvarez *et al.*, 1992) provided information on many of the practical considerations such as the plant design (HRT, recirculation rate, biosolids destruction rate, mass balance), electricity demands and operating and capital costs. This paper found again that landfill costs had a significant influence on the financial viability of the plant. The paper was written a relatively long time ago but the study is applicable to a current setting.

A feasibility study based in the UK (BRE/WRAP, 2013) focused on micro-scale AD of food and garden waste, and included a plant design, regulation considerations, a cost-benefit analysis, outputs, funding resources, and health and safety considerations. The study provided useful information on UK-specific grants, financial incentives and legislation issues. The study concluded that in this situation, there was no scenario in which a micro-scale AD plant could prove to be economically viable, but that analysing the plant as part of a community or larger energy plan (such as a community microgrid) might improve the financial viability.

A further feasibility study of a large AD project (Moriarty, 2013) described a plant in the planning stage that would process food waste from a number of towns in Louisiana, USA. The result of the study was that the plant was not feasible, due to the low cost of landfill and energy in the area, and the high investment cost of the AD technology.

2.3.3 Economic effect of plant size

A study was reported of the relationship between the size of plant and levelized cost of electricity (LCOE) for AD plants processing cow slurry (Oreggioni *et al.*, 2017). The study found that the LCOE was 4.3p kWh_e^{-1} for a plant processing waste from 125 dairy cows, compared to 1.9p kWh_e^{-1} for a plant processing waste from 1000 dairy cows, which are both attractive compared with current UK average electricity supply price of 14.4 p kWh_e^{-1} (Choose.co.uk, 2018). This study included feed-in tariffs, which have since been removed for all renewable electricity providers except solar PV (GOV.UK, 2018b). A second paper, based in Canada, produced a techno-economic analysis of the effect of plant size on the digestion of household source-separated organic waste (Sanscartier, MacLean and Saville, 2012). The paper found that the most competitive size of AD facility would be one that processed over 30,000 tonnes per year, increases in size above 50,000 tonnes a year did not improve the relative carbon emissions savings, and that feed-in tariff rates at that time in Canada were not sufficient to make smaller scale plants economically viable. From these studies, it is reasonable to conclude that micro-scale AD plants are not innately financially viable as they are currently designed, and that a feed-in tariff of the correct level was required for smaller AD plants to be built.

2.3.4 Economic effect of pre-treatment

The pre-treatment of organic sludge by disintegration as a method of minimising costs was reported on (Winter, 2002). This study found that the biogas production could be increased by 20% but that the process was economically advantageous only if disposal costs were high. The use of this technique would therefore need to be decided on a case-by-case basis.

2.3.5 Economic effect of feedstock and feed processing system

This thesis focused on food waste as a feedstock, which is a waste stream and as such earns ‘revenue’, either in avoided disposal costs if the system is operating in-house, or gate fees if the system accepts waste from external clients.

In the case of a micro-scale AD plant, the feedstock source is expected to be local, with low transportation costs. A study of a life-cycle analysis for anaerobic digestion plants (Patterson *et al.*, 2011) compared centralised and distributed plants, and concluded that the transportation requirements of a centralised system versus a number of distributed plants had very little effect on the carbon emissions and cost of fuel used. However, this study used a small country (Wales) where transport distances would be relatively short anyway, and therefore the distances used to calculate the difference between the two systems were actually very similar – the average distance travelled in the centralised system was 36.7 km, and for the distributed system was 22.9 km. The centralised system had 5 AD plants, and the distributed system had 11. The justification for this similarity was that the authors considered a plant processing 6326 tonnes of food waste a year to be the smallest economically viable plant, which made the minimum distribution larger between plants – if the plants had been smaller, they would have been placed closer together. In the same paper, the authors suggest that 'a large increase in transportation requirement for the centralised infrastructure does produce a significant difference between the centralised and distributed infrastructures', so potentially modelling the system in a larger country, and including a larger number of micro-scale AD plants would have had a more significant effect on transportation costs.

2.3.6 Economic effect of digester and system design

The urban micro-scale AD plants that have been documented in the literature have been continuously stirred-tank reactors (Walker *et al.*, 2017; Curry and Pillay, 2009; Riggle, 2013) and commercial micro-scale AD reactors are also universally CSTR types (Decisive 2020, 2018; Qube Renewables, 2017; The Waste Transformers, 2016; Methanogen (UK) Ltd, 2019). There appears to be no research of application of two-stage reactors at this scale, even though they have been found to have a higher yield (Schievano *et al.*, 2014; De Gioannis *et al.*, 2017). This is a knowledge gap that would be worth further investigation.

Research into a 'systems' approach for anaerobic digestion (Stoknes *et al.*, 2016) described a system of prototype 1200-litre digester connected to a hydroponic growing system, as an example of a circular economy (figure 2-20).

Figure 2-20: Visualisation of the overall concept for the food-to-waste-to-food project (Stoknes et al., 2016).

The system optimises food cultivation by connecting it with the anaerobic digestion process, receiving carbon dioxide and heat from the biogas combustion, and nutrients from the compost made from the digestate, and the products are recycled into inputs. The authors provided detailed information about the nutrients gained from the digestate and concluded that the system would operate without any additional fertiliser. This is an important aspect of a circular economy; unless the commodity exchange in the system is quantified, it is not possible to gauge how effective the system is. Although no cost-benefit information was included, the paper found that the system was feasible and biologically beneficial.

The size of a micro-scale digester means that is possible to locate it in a greenhouse. This concept has been tested in the Camley Street AD plant in London (Walker *et al.*, 2017), where it was concluded that the solar gain and insulation properties of the greenhouse reduced the heating requirements for the digester by 49%. This calculation was made by taking daily single measurements of the inside and outside temperatures throughout the year, so does not allow for daily variations. A more accurate figure could be calculated by using a computer simulation to model the system throughout the year.

A model of a theoretical network of micro-scale anaerobic digestion plants was created with an aim to minimise the transportation distances for the feedstock (household food waste and green waste) and digestate (Thiriet, Bioteau and Tremier, 2019). This resulted in two models: a decentralised network of 273 micro-scale (<64 T y^{-1}) anaerobic digestion plants in a 534 km^2 area, or a network of 143 micro-scale AD plants with a single central treatment plant. However, although the study built a useful model for designing a micro-AD plant network, it did not include any modelling of the economics or quantify the social or environmental impacts, which would be a key progression to make the model useful for developers.

2.3.7 Economic effect of digestate use

Digestate is an output of anaerobic digestion that is produced in high volume and contains potentially valuable nutrients (Drosg *et al.*, 2015). A report from the IEA bioenergy group (Drosg *et al.*, 2015) compared the costs of different methods of processing digestate (screw separation, drying, centrifugation) with spreading to land and concluded that the costs were very site-specific but that drying and centrifugation were approximately three times more costly than screw separation or spreading to land.

The post-processing of digestate through hydroponics and algae cultivation in small-scale plants (producing less than 200L of digestate per day) was found to be financially viable (Fuldauer *et al.*, 2018). The economics could be further improved by sharing one digestate enhancement facility between several plants, increasing algal growth or using vertical hydroponics. The study concluded that financial support to further develop the technology would be required to make small-scale AD using these processes viable.

2.3.8 Composting

Composting can be used as a post-processing mechanism for anaerobic digestate, with some financial implications. Composting is an exothermic process, and it can satisfy the requirements for animal by-products sanitation if it attains a specified temperature for a predetermined amount of time to achieve pasteurisation. For a closed composting container, in the EU the time-temperature required is 60°C for 2 days or 70°C for 1 hour (EC-European Commission, 2003). The heat produced can be harnessed to provide a hot water supply, which can be more reliable and more cost-effective than other renewable sources such as solar hot water and ground source heat pumps (Irvine, Lamont and Antizar-Ladislao, 2010).

The process of composting releases a combination of latent heat (as increased water vapour) and sensible heat (as increased temperature), which form approximately 86% and 14% respectively of the energy released (Smith, Aber and Rynk, 2017). There are three methods of capturing the heat: direct utilisation of the heat and vapour (for example, making a compost heap in a greenhouse to heat the greenhouse), using a heat exchanger within the compost pile (figure 2-21), and using heat exchange with the compost vapour, with the last of these being the most efficient (Smith, Aber and Rynk, 2017).

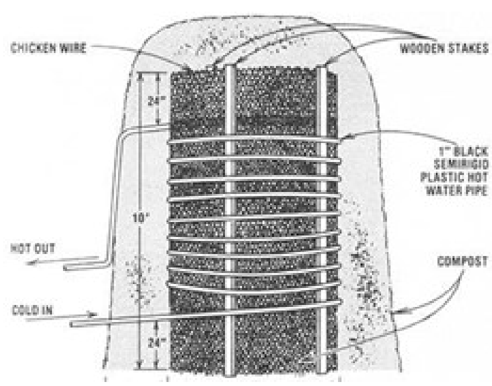


Figure 2-21: Jean Pain Composting (Mother Earth News, 1980).

The third method, a heat exchange system, requires a closed vessel, which is sealed and insulated, with air recirculation and heat-exchanging pipework mounted in the roof-space (figure 2-22).

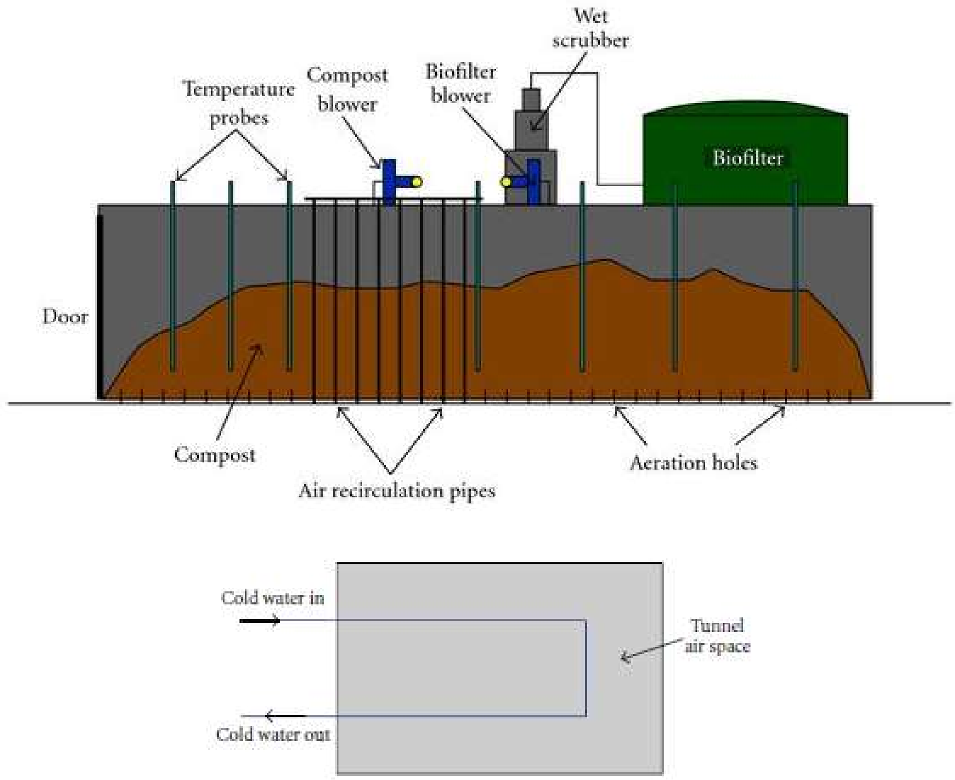


Figure 2-22: Cross-section of in-vessel composting unit, with pipework layout design (Irvine, Lamont and Antizar-Ladislao, 2010).

This technique is able to capture both the sensible and latent heat (by condensing the water vapour) and results in a greater potential for heat recovery (Smith, Aber and Rynk, 2017).

Energy recovery from compost is typically reported in kJ hr^{-1} or kJ $kgDM^{-1}$ (dry matter). This variation in the units used makes it difficult to compare different systems in published literature, and the Smith review rightly recommends that the energy recovered is universally measured in kJ $kgDM^{-1}$ (Smith, Aber and Rynk, 2017). The review presents figures from a number of different systems which range very widely in their heat recovery capability. The within-pile systems range from 89 to 27491 kJ kg^{-1} and the vapour condensing systems range from 148 to 10000 (theoretical) kJ kg^{-1}. These figures do not include systems that have stated their energy production in kJ hr^{-1} as they cannot be compared and appear to show higher energy being produced from within-pile systems. However, a system within the same review was reported to have upgraded its heat recovery system from within-pile to an exhaust vapour condenser, with an accompanying heat recovery increase from 4294 kJ $kgDM^{-1}$ to 11041 kJ $kgDM^{-1}$ (Smith, Aber and Rynk, 2017). Assuming that the data was collected in the same way from the same system for both heat collection methods, it is clear that heat recovery through vapour condensation is more efficient.

This review (Smith, Aber and Rynk, 2017) summarised very well the progress of compost heat recovery technology up to the point of its publication and made clear descriptions of each approach. However, despite the review being recent, many of the citations were not accessible. The paper stated calculated figures for the energy recovery in kJ $kgDM^{-1}$ for data that actually represent the energy recovery in kJ kg^{-1} when checked against the heat recovery rate in kg hr^{-1} and the feed input – an important distinction.

Co-composting of organic wastes with digestate was investigated (Arab and McCartney, 2017) by adding increasing proportions of digestate by % weight to the organic fraction of municipal solid waste (OFMSW). The ideal resulting moisture content and free air space were reported as approximately between 50% and 65% and >30% respectively (Christensen, 2011; Alburquerque *et al.*, 2008), and the materials in this study were adjusted to these values by adding water and woodchips as required. The research found that the optimal ratio was 3:7 digestate to OFMSW, i.e. 30% digestate. The research presented information on the changing values of the overall organic matter removal and the heat production over time, which was useful to understand further how the kinetics of the process were altered when different amounts of digestate were added. A secondary important finding from this paper was that at higher proportions of digestate, the amount of ammonia in the mixture (above 5000 mg kg^{-1}) became inhibitory. When using digestate as a co-composting material, it is therefore important to note the nitrogen content and the carbon-nitrogen ratio.

2.4 Feedstocks for micro-scale anaerobic digestion

2.4.1 Reported feedstocks in operational micro-scale AD plants

Micro-scale anaerobic digestion is a decentralized biowaste treatment system and therefore the feedstock is generally that which is available locally as it is convenient and minimises the need for transport. As with large-scale plants, a micro-scale plant is designed based on the feedstock available.

A common scenario for micro-scale anaerobic digestion in the developed world is a plant that processes organic waste for a collection of small local facilities, such as the plant in Lyon, France (Decisive 2020, 2018), which was designed to run on food waste from a small group of houses and catering facilities. Similarly, a micro-scale AD plant in Amsterdam, The Netherlands (The Waste Transformers, 2016) was installed to process organic waste from a site housing several small businesses such as restaurants and theatres, and in London, UK, a micro-AD plant ran on food waste collected by bicycle from businesses in the local area (Walker *et al.*, 2017). A review of urban anaerobic digestion facilities (Angeli *et al.*, 2018) listed a number of plants with their feedstocks, including food waste, sewage, garden waste and wastewater, with food waste being the most prevalent. Further studies of micro-scale anaerobic digestion in the developed world (BRE/WRAP, 2013; Curry and Pillay, 2012) also focussed on food waste as a feedstock.

Micro-scale AD plants in developing countries are generally located in more rural settings, and the feedstocks are typically animal and crop waste, human waste and food waste (Gunnerson and Stuckey, 1986; Singh and Kaushal, 2016).

This thesis is based on micro-scale anaerobic digestion in the context of the developed world urban environment, and from the studies listed above it is reasonable to view food waste as a likely feedstock, and therefore choose it as the feedstock to be used in the experimental section of the thesis.

2.4.2 Characteristics of food waste

Research into food waste digestion has shown that food waste is highly digestible compared to other waste streams (Appels *et al.*, 2011; O'Shea, Wall and Murphy, 2016; Browne, Allen and Murphy, 2013) with a typical VS destruction rate of 85% (Banks *et al.*, 2018; Paritosh *et al.*, 2017). Food waste digestion works well at mesophilic and thermophilic temperatures and can be enhanced by co-digestion (Zhang *et al.*, 2014; Banks *et al.*, 2018). It generally has a high moisture content compared to other feedstocks – between 70% and 82 % depending on the source (Banks *et al.*, 2018) but its composition can vary a lot with location (Zhang *et al.*, 2014; Izumi *et al.*, 2010; Radu *et al.*, 2016; Kuczman *et al.*, 2018). A summary of the characteristics of food wastes from different papers (Xu *et al.*, 2018) shows the high variability that exists (table 2-4).

Table 2-4: Generation, chemical composition, and methane potential of some food wastes (Xu et al., 2018).

Type	**TS%**	**VS (% of TS)**	**C:N ratio**	**pH**	**Carbohydrate (% of TS)**	**Protein (% of TS)**	**Lipids (% of TS)**	**Methane yield (m^3 $kgVS^{-1}$)**
Fruit and vegetable waste	7.4–17.9	83.4–95.3	15.2–18.9	3.7–4.2	–	10.5–17.8	0.8–5.2	0.16–0.35
Slaughter-house waste	2.0–28.3	82.7–93.6	3–6	–	0–27.7	2.0–38.9	1–40.5	0.20–0.50
Brewery waste	23.0–29.2	87.6–97	18.8–54.9	6.9	–	23.0–32.0	5.7–10.6	0.22–0.31

Dairy waste	0.1–7	–	11.4–13.6	6–11	3–43.8	1.4–33.5	0.1–11.6	0.1–0.85
Waste pet food	86–93	74.6–94.5	10–25	–	–	6.0–34.5	1.0–30.8	0.15–0.50
Fat, oil, and grease (FOG)	1.3–3.2	86.0–93.9	22.1	4.2–4.8	0.8	10.2	75.4–100	0.4–1.1
Household and restaurant food waste	4.0–41.5	88.7–95.1	11.4–36.4	3.3–5.7	3.3–59.0	1.4–22.8	4.0–41.5	0.46–0.53

The characteristics of a typical European food waste sample were summarized by (Banks *et al.*, 2018) (table 2-5).

Table 2-5: Model values for a typical European food waste (Banks et al., 2018).

Parameter	**Unit**	**Typical value**
TS	% fresh matter	24
VS	% fresh matter	22
TKN	g kg^{-1} fresh matter	7.4
Calorific value	MJ kg^{-1} VS	22
N	g kg^{-1} VS	31
P	g kg^{-1} VS	4
K	g kg^{-1} VS	13
C	% VS	52
H	% VS	6.9
O	% VS	38
N	% VS	3.4
S	% VS	0.3
Biochemical Methane Potential (BMP)	m^3 CH_4 $kgVS^{-1}$	450

The table contained an error - the BMP units are shown as m^3CH_4 $kgVS^{-1}$ when it should read LCH_4 $kgVS^{-1}$ or m^3CH_4 $tonneVS^{-1}$. Other than this the values are consistent with the values in table 2-4. The biological methane potential of standard household food waste is relatively high compared to other feedstocks (Appels *et al.*, 2011; Curry and Pillay, 2012).

2.4.3 Experimental use of synthetic food waste

To ensure that the content of the feedstock used is consistent over the course of the experiment, it is possible to use a ‘synthetic’ food waste. In a food waste AD study, a synthetic food waste recipe was derived from food waste statistics (WRAP, 2012b)(Radu *et al.*, 2016). This food waste recipe includes categories of vegetables (38%), Fruit (21%), Bakery (16%), Meat/fish (11%), Drink (10%) and Dairy (4%). For comparison, the food waste recipe from (Izumi *et al.*, 2010) is also provided (table 2-6).

Table 2-6: Synthetic UK food waste recipe (Radu et al., 2016).

(Radu *et al.*, 2016) g/kg of recipe		**(Radu *et al.*, 2016) Recipe % by weight**		**(Izumi *et al.*, 2010) Recipe % by weight**	
Potato	237	Vegetables	38	Vegetables	54
Onion	40				
Carrot	37				
Cabbage	26				
Lettuce	21				
Tomato	19				
Banana	114	Fruit	21	Fruit	25
Apple	96				
Bread	160	Bakery	16	Rice, noodles and bread	8
Beef	55	Meat and fish	11	Meat, fish and eggshells	5
Pork, Ham and Bacon	55				
Tea	100	Drink	10	Tea	8
Buttermilk	40	Dairy	4		
TOTAL	**1000**	**TOTAL**	**100**		**100**

There are some significant differences between the western and eastern diet that this highlights; Japanese food waste includes a much larger proportion of vegetable waste, whereas the UK food waste recipe is higher in bread/carbohydrates and meat/fish, and contains a proportion of dairy.

An alternative to using a synthetic food waste ‘recipe’ is to use dry dog food, as was used in a study on aerobic decomposition, with the carbon and nitrogen content tested as 44.6% and 5.3% of dry solids respectively, and the volatile solids content given as 89.5% of total solids (VanderGheynst, Gossett and Walker, 1997). These figures show a similar content to food waste, which has a carbon and nitrogen content of 52% and 3.4% by weight of dry solids respectively, and the VS makes up 92% of the total solids (table 2-5). The authors stated that dry dog food was used because it had a uniform physical consistency and a similar content to ‘standard’ food waste. A second study, also based on aerobic decomposition, used dry dog food as a co-composting substrate, but didn’t comment further on its similarly or differences to standard food waste (Lemus *et al.*, 2004). If dry dog food is used as a substitute in the experimental work, analysis such as biological methane potential and calorific value testing would be necessary to further compare it with food waste.

2.4.4 Known issues

Each feedstock for AD has specific known issues that the operators need to be aware of in order to anticipate and avoid potential problems. A recent review of food waste research (Ren *et al.*, 2017) noted that of particular interest in research was ammonia inhibition effects. Ammonia inhibition causes a build-up of VFAs (Chen, Cheng and Creamer, 2008). Problems with food waste digestion can be an imbalance of nutrients such as lipids, which can be overcome by managing the feedstock input or co-digestion with alternative feedstocks (Zhang *et al.*, 2014). A lack of trace elements can also be a long-term problem, again causing a build-up of ammonia (Zhang *et al.*, 2014; Walker *et al.*, 2017). Food waste can have a high nitrogen (TAN) content, which can cause ammonia toxicity at higher pH (Chen *et al.*, 2016). Food waste digestion can also experience process problems such as accumulation of VFAs caused by the rapidity of degradation of the feedstock (Xu *et al.*, 2018). Foaming in food waste digesters can be a problem, caused by high VFAs (Subramanian and Pagilla, 2015), the presence of surfactants (surface active agents) or sudden gas release (Xu *et al.*, 2018). Foaming issues can be treated by adding anti-foaming agents, reducing lipids in the feedstock, lowering the OLR, and better management of the digester to reduce process upsets (Xu *et al.*, 2018).

The addition of micronutrients (iron, zinc, selenium, manganese) have been proven to be beneficial to the long-term operation of food waste AD plants to prevent inhibition from ammonia build-up and nutrient deficiency (Banks *et al.*, 2012; Zhang and Jahng, 2012).

Contamination of the feedstock by foreign bodies such as plastic bags or metal cutlery can be an issue in food waste, as it comes from a number of different sources and so is more difficult to control. However, this can be mitigated by providing smaller containers (Banks *et al.*, 2018). Food waste contamination is lower if it is collected separately rather than being collected as a sub-fraction of municipal solid waste and if a good collection scheme is in place (Banks *et al.*, 2018).

2.4.5 Regulations

Due to its meat and dairy content, food waste potentially contains pathogens and is regulated in the UK by the Animal By-Products regulations 2003 and the EC Animal By-Products Regulation 1774/2002 (GOV.UK, 2013; Duckworth, 2005). The regulations state that any catering waste containing animal by-products (including meat) must be treated at 70°C for 1 hour or 60°C for 2 days in a closed reactor. There are guidelines specifically for composting (Duckworth, 2005) and for both composting and anaerobic digestion (GOV.UK, 2014b), which state what type of waste can be treated and how it should be stored and handled.

2.4.6 Digester design

An alternative to the commonly-used CSTR (continuously-stirred tank reactor) design is the two-stage design in which the hydrolysis and methanogenesis stages are separated. This type of design appears to be well suited to food waste digestion as it increases stability, which is important for a feedstock that degrades rapidly (Xu *et al.*, 2018; Bouallagui *et al.*, 2005). A two-stage design for food waste AD was compared to a one-stage system (De Gioannis *et al.*, 2017) and found that the two-stage system achieved better fermentation, resulting in a 20% higher methane yield. The two systems were also compared by (Mohan and Bindhu, 2008), who reported that the two-stage system could support a higher OLR (8 kg VS m^{-3} day^{-1} compared to 5.5 kg VS m^{-3} day^{-1}) , achieved higher removal rates of both COD and VS, and was more stable.

Food waste can be digested at mesophilic or thermophilic temperature, each having their own advantages. Digestion at mesophilic temperature has been shown to be more stable, whereas thermophilic temperatures increase the methane production rate, allowing a lower retention time and smaller digester (Curry and Pillay, 2012).

2.4.7 Alternative routes of disposal for food waste

Food waste disposal in developed countries can be managed via a number of different streams. In the UK, food waste that is not separated from the principal waste stream is referred to as 'Organic Fraction of Municipal Solid Waste' (OFMSW). The OFMSW can be separated in a Materials Recovery Facility, and then used as a feedstock for anaerobic digestion (Fei *et al.*, 2018). If the waste is sent to landfill, the energy it contains is lost unless the landfill is covered and the methane that results from any organic waste breakdown is captured and combusted. This is a common practice in the UK (Department of Energy and Climate Change (DECC), 2013). The solid waste can be incinerated, which includes some energy recovery (Nixon *et al.*, 2013), or be used in the process of pyrolysis or gasification, both of which create a combustible fuel (Jain *et al.*, 2018).

If the food waste is separated at source, it can be sent to a food waste-specific anaerobic digestion plant (ReFood, 2021). If it is collected as part of 'green waste' (mixed organic waste from the house and garden), it can be composted in a large waste facility (Wei *et al.*, 2017). Within the home, separated food waste can be disposed of via an in-sink macerator (known as a garbage disposal (US), food waste disposal unit (UK), or Insinkerator). The waste from the macerator goes to the sewer connection (Iacovidou and Voulvoulis, 2018). Alternatively, food waste can be collected as a separate waste stream by the household or business and composted with other waste streams such as cardboard and garden waste.

These food waste management techniques were described and compared qualitatively by a comprehensive report on food waste produced by the World Biogas Association (Jain *et al.*, 2018). The findings of the report are summarised in Table 2-7.

Table 2-7: Comparison of food waste management technologies, derived from (Jain et al., 2018).

Technology	Carbon emissions avoidance	Installation cost	Energy recovery	Nutrient recovery	Scale	Operation
Anaerobic digestion	High - produces renewable fuel or renewable electricity	High	60% more than combustion	Produces a nutrient-rich fertiliser	Most economical at large scale	Requires expert management
Composting - in-vessel and windrow	Avoids emissions from landfill	Low	No energy recovery	Produces high organic matter compost	All scales	Simple to operate, windrows cannot be used to treat animal wastes
Liquefaction	Avoids emissions from landfill	Low	Good if waste goes to AD	May produce digestate	Small scale	Requires waste water systems that can cope with the extra organic load
Gasification	Medium - generates heat/electricity	High	Depends on waste - can be negative if high moisture content	No nutrient recovery	Large scale	Complicated pre-processing needed; works better with homogeneous feedstocks
Incineration with energy recovery	Medium - generates heat/electricity	High (but lower than gasification and pyrolysis)	60% less efficient than AD	No nutrient recovery	Large scale	No separation needed
Landfill without gas collection	Negative impact on environment - releases greenhouse gases and toxic substances	Low	No energy recovery	No nutrient recovery	All scales	Can be extremely dangerous if not designed properly
Landfill with gas collection	Low - gas is poor quality but can be used to generate electricity	Medium	Yes, by methane collection, although some will escape	No nutrient recovery	Medium to large needed to generate sufficient gas	Potential toxins and variability of methane content in gas produced
Pyrolysis	Medium - generates heat/electricity, lower than AD	High - technology not mature	Yes, potentially more than incineration	No nutrient recovery	Can operate at small and large scale	Technology is less familiar as not mature
Mechanical biological treatment	Low	High	Yes, by AD of food waste fraction, but low overall recovery as the process is energy-intensive to run	Some recovery via anaerobic digestion	Large scale	High cost, energy intensive, complex

The report demonstrated that each method of food waste management has both advantages and disadvantages, and the technology used must be chosen based on the circumstances. For example, depending on whether the right expertise and capital are available, whether there are limitations due to the nature or amount of the feedstock, or whether there are incentives or policies imposed by the government that make one solution more advantageous than another. In this summary, anaerobic digestion was shown to be a technology that has great value as a method of food waste disposal.

2.5 Summary of gaps and opportunities in research

There is a general absence of research specific to micro-scale anaerobic digestion in the developed world due to its perceived lack of economic viability. A case study could therefore make a useful contribution to the body of knowledge by exploring the specific challenges to this scale of AD plant. A techno-economic analysis could quantify the feasibility of a micro-scale AD plant and report on ways of making it more economically viable.

In terms of variable biogas production, there is a lack of reported information available on the effects on digester other than the microbial population and digester stability. Therefore, a study of other system indicators such as the biogas production and composition, changes to the inoculum, and volatile solids breakdown could add useful data in this area.

3 Research methodology

3.1 Analytical methods

The feedstock composition and physical, chemical and biological attributes were determined using biological methane potential (BMP), total and volatile solids (TS/VS), and CHNS (Carbon, Hydrogen, Nitrogen and Sulphur testing). The chemical and physical attributes of the liquid digestate from the digester and the digestate tanks (overflow from the digesters) were tested using TS/VS and CHNS tests. Stability of the digestate was measured using pH, alkalinity and volatile fatty acids (VFA) analysis. The biogas was measured for methane and CO_2 content.

Table 3-1: Analysis schedule for laboratory work.

	Feedstock	**Digestate**	**Biogas**
BMP	•		
CHNS	•		
TS/VS	•	•	
pH		•	
Alkalinity		•	
VFA		•	
Microbial population		•	
Methane and CO_2 content			•
Biogas volume			•

3.1.1 General

Best practices were followed in the laboratory, with CoSHH assessments and Risk Assessments completed for the processes as required. Good standards of hygiene and secure access for trained personnel only were maintained throughout the testing period to ensure that there was no transfer of biologically active substances or chemicals to areas outside the laboratory.

3.1.2 Biological methane potential

A biological methane potential (BMP) test was performed on the feedstock (figure 3-1, figure 3-2) using purpose-built BMP testing equipment (AMPTS II, Bioprocess Control, Sweden).

Figure 3-1: The Bioprocess Control automated BMP equipment, during operation.

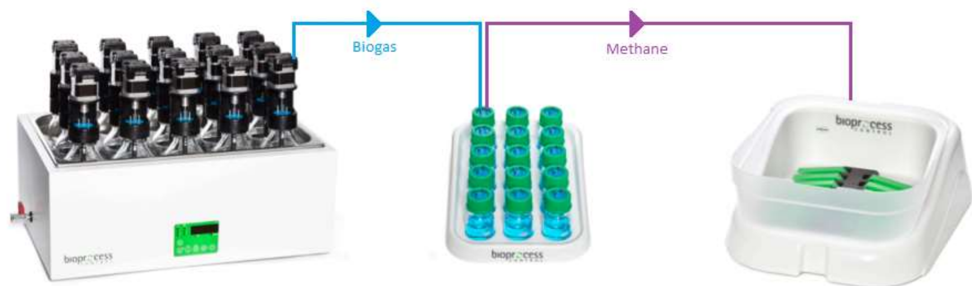


Figure 3-2: The Bioprocess Control 'AMPTSII' BMP automated rig: incubation unit (left), carbon dioxide absorption unit (middle) and flow cell array and DAQ unit (right).

At the start of the BMP test, approximately 10 L of anaerobic digestate inoculum was sourced from a local food waste digestion plant, sufficient to fill 15 test jars with 400 mL each, with some inoculum left over for testing. After collection, the inoculum was immediately filtered to ensure it was homogenous. The approximate ratio of inoculum VS to sample VS by weight was set at 3:1 to ensure that a measurable amount of methane would be produced during the test. For the purposes of setting up the test, the % VS (by weight) of the inoculum was estimated based on previous tests. The % VS of the substrate was then measured (section 3.1.3). Blank samples (inoculum only) and control samples (inoculum with cellulose, with a known methane potential) were set up at the same time.

For each jar, approximately 400mL of inoculum was measured into the jar and the weight of inoculum recorded to an accuracy of ± 5 g (SJ-12KHS, Cole-Parmer, UK). Then the required amount of substrate was weighed and added into the jar, to an accuracy of ± 0.001g (UY-

20000-33, Symmetry EC series toploading balance, Cole-Parmer, UK), and the jar was immediately sealed and set up in the test rig.

The jar was placed in an incubation unit and connected via Tygon gas tubing to a jar in the carbon dioxide absorption unit, which was then connected via Tygon gas tubing to the flow cell array. The headspace of the jar system was purged of oxygen by flushing with synthetic biogas (65% methane, 35% carbon dioxide, Calgaz, UK). The set up of the test jar was completed by starting the stirrer and starting the recording of gas production on the web-based experimental data record. Each test jar was prepared and started in this way before preparing the next, to minimise gas loss. When all the test jars had been set up, the incubation unit was filled with deionised water and set to 38°C.

The BMP of each sample was calculated (equation 3-1).

Equation 3-1

$$BMP = \frac{V_S - V_B \frac{m_{IS}}{m_{IB}}}{m_{VS,sS}}$$

Where: BMP is the normalised volume of methane produced per gram VS of substrate added (NL gVS^{-1})

V_S is the accumulated volume of methane produced from the reactor with sample (i.e., inoculum and substrate) (mL).

V_B is the mean value of the accumulated volume of methane produced by the three blanks (i.e., inoculum) (mL).

M_{IS} is the total amount of inoculum in the sample (mL).

M_{IB} is the total amount of inoculum in the blank (mL).

$M_{VS,sS}$ is the amount of organic material (i.e., volatile solids) of substrate contained in the sample bottle (gVS).

3.1.3 Total and volatile solids

Total solids (TS) and organic or volatile solids (VS) tests were performed on both the feedstock and digestate according to APHA standard methods (APHA, 2005). Crucibles to be used in the test were cleaned with detergent, rinsed with water and deionised water, dried in an oven (Thermo Heratherm OGS60, Thermo Fisher Scientific, Germany) at 105°C, then placed in a desiccator prior to use. The sample was homogenised before being measured to a sensitivity of ± 0.001g (UY-20000-33, Symmetry EC series toploading balance, Cole-Parmer, UK) using a metal spatula or pouring into a dry, weighed crucible and dried the oven for at least 24 hours at 105°C ± 1°C. The crucible was then left to cool for up to half an hour to room temperature in a desiccator before being re-weighed. The crucible was then transferred to a cool furnace (Elite BSF12/10A Box furnace, Elite Thermal Systems Ltd, UK) and heated at 550°C ± 5°C for 2 hours with a 14°C/min ramp up rate. This step left only the ash portion of the sample remaining in the crucible. The crucible was left to cool to room temperature in a desiccator then weighed for the final time. After the test, the crucible was washed using a detergent, rinsed with water and deionised water and then left to dry in the oven until the next TS/VS analysis.

Total and volatile solids were calculated according to the following formulas:

$$TS\ (\%\ by\ weight) = \frac{C - A}{B - A} \times 100$$ *Equation 3-2*

$$VS\ (\%\ by\ weight) = \frac{D - C}{B - A} \times 100$$ *Equation 3-3*

Where: A = Weight of the crucible.
B = Weight of the crucible with sample.
C = Weight of the crucible with dried sample.
D = Weight of the crucible after ignition in the furnace.

3.1.4 CHNS

CHNS analysis was performed using an elemental analyser (Flash 2000, Thermo Scientific, Germany). Calibration was performed by running the analysis on a standard tin capsule containing 10mg vanadium pentoxide (to facilitate the sulphur ionisation). Two reference samples were prepared, containing 10mg vanadium pentoxide plus approximately 5mg of 2,5-Bis (5-tert-butyl-benzoxazol-2-yl) thiophene (known as BBOT) through the analyser prior to running the analysis on the samples. All sample weights were measured to a sensitivity of ±0.001 mg and recorded (CPA2P balance, Sartorius, Germany). The feedstock to be tested was homogenised using a food mixer then dried for 24 hours at 105°C, ground using a pestle and mortar and then dried for a further 24 hours at 105°C. Three duplicate capsules were made for each sample using 10mg vanadium pentoxide with approximately 5mg dried feedstock.

The carrier gas was hydrogen delivered at a flow rate of 200 mL min^{-1}, enriched with oxygen at a flow of 300 mL min^{-1}. The samples were heated to a temperature of 900°C for 700 seconds and the CHNS content was quantified using a flame ionization detector and a capillary GC column of type CE Instruments CHNS/NCS PTFE 2m packed column.

3.1.5 pH

pH readings were taken using a benchtop pH meter (Accumet AE 150 benchtop meter with Accumet AE6 3-in-1 single junction gel pH/ATC electrode, Fisher Scientific, UK). The pH meter was calibrated weekly during the experimentation period using standard solutions of pH 4, 7 and 10 (Atlas Scientific, USA).

The pH probe was stored in a 3M KCl solution (Fisher Scientific, UK) to prevent deterioration. Before and after each use, the probe was rinsed thoroughly with deionised water.

3.1.6 Alkalinity

The alkalinity of digestate samples was tested according to APHA standard method 2320 B (APHA, 2005). A 5 ml aliquot of the sample was added to a 50 ml beaker and made up to 20 ml using deionised water. The sample was then titrated in an automatic digital titrator (Titroline 5000 titrator, SI Analytics, Germany). The sample was stirred using a magnetic flea while the titration acid (0.1N/0.05M H_2SO_4) was added. The end points were 5.7 and 4.3, to calculate partial alkalinity and total alkalinity respectively (Ripley, Boyle and Converse, 1986). The pH probe was calibrated using standard buffers of pH 4, 7 and 10 on a monthly basis as described in section 3.1.5.

Alkalinity was calculated according to the formulas (equation 3-4, equation 3-5 and equation 3-6).

$$PA = \frac{A \times N \times 50000}{V_s}$$ *Equation 3-4*

$$IA = \frac{B \times N \times 50000}{V_s}$$ *Equation 3-5*

$$Alkalinity\ ratio = \frac{IA}{PA}$$ *Equation 3-6*

Where: PA is the partial alkalinity, in mg $CaCO_3$ L^{-1}.
IA is the intermediate alkalinity, in mg $CaCO_3$ L^{-1}.
A is the volume in mL of H_2SO_4 added to reach intermediate endpoint (pH 5.7).
B is the additional volume in mL of H_2SO_4 added to reach final endpoint (pH 4.3).
N is the normality of the titrant, H_2SO_4.
V_s is the sample volume in mL.

3.1.7 Volatile Fatty Acids

The volatile fatty acids (VFA) content was measured using a gas chromatograph (GC) based on APHA standard method 5560 D (APHA, 2005). A 2mL sample of digester sludge was collected in a glass beaker where it was acidified to pH 4 using 96% formic acid (Sigma-Aldrich, UK) and diluted to 10mL with deionised water. 2 mL of the acidified sample was then centrifuged at 13000 rpm for 30 minutes. The resulting supernatant was filtered through a 0.45 µm-grade filter to obtain a clear sample.

10 µL of the sample was loaded in duplicate into a GC (Trace 1300, Thermo Scientific, Germany) for analysis. A standard solution mixture of formic, acetic, propionic, iso-butyric, n-butyric, isovaleric, valeric, hexanoic and heptanoic acids was used at concentrations of 10, 100 and 1000 mg L^{-1}. The VFA was quantified using a flame ionization detector and a capillary GC column of type Thermo TR-FFAP. The carrier gas was helium delivered at a flow of 50 mL min^{-1} and a split ratio of 50 to give a flow rate of 1mL min^{-1} in the column and a 5.0 mL min^{-1} purge. The GC oven was programmed to hold at 80°C for 1 minute, then increase to 200°C over 8 minutes, hold at 200°C for 6 min, then increase over 1.5 minutes to 240°C with a final hold time at 240°C of 4.1 minutes. The full programme was 20.6 minutes. The temperatures of the injector and detector were 200°C and 240°C respectively.

3.1.8 Gas composition

Biogas composition was measured continuously using infrared hydrocarbon sensors (MSH-PS/HC/NC and MSH-DP/HC/HCO2/NC, Dynament Ltd, UK).

The gas sensors were calibrated every two weeks by zeroing with air and calibrating against synthetic biogas (65% methane, 35% carbon dioxide, Calgaz, UK).

3.1.9 Gas volume

Gas volume was measured using an ultra-low flow gas flowmeter (µFlow, Bioprocess Control, Sweden). The resolution of the flowmeter was 10 mL ± 1 mL with a precision of 1%. The flowmeter cell volume was calibrated at the factory and this value entered into its processing unit. The gas measurements were automatically normalised by the flowmeter to 0°C and 1 atm (STP).

3.2 Operational issues

The gas chromatography column had been in use in a teaching laboratory for over two years and at the start of the testing period was found to sometimes be unreliable when repeating tests on samples. Several remediation tasks were performed such as shortening, cleaning and replacing parts before reproducible results were produced.

To ensure the quality of the results, a cleaning cycle with methanol was run on the chromatography column before every testing session, was calibrated with repeats of standard solutions at 0.1 mM, 1 mM, 5 mM and 10 mM and checked with blank samples (i.e. deionised water). The results were processed and assessed promptly so that they could be repeated if necessary.

4 Design, commissioning and operation of the lab-scale anaerobic digestion plant

An automatically controlled laboratory-scale anaerobic digestion (AD) plant was built for the experimental study, to mimic the workings of an industrial AD plant. This chapter describes the design, build and commissioning of the plant.

4.1 Design and construction

4.1.1 Design

The experimental rig consisted of two identical 'streams' of equipment with some shared items. One complete stream consisted of a feed tank, pump, digester, digestate tank, gas flowmeter, gas sensors and a National Instruments LabView CompactRIO control system (figure 4-1). Both digesters were fed from the same feed tank and were heated by the same water bath, which ensured that both the feed and heat to the digesters was the same.

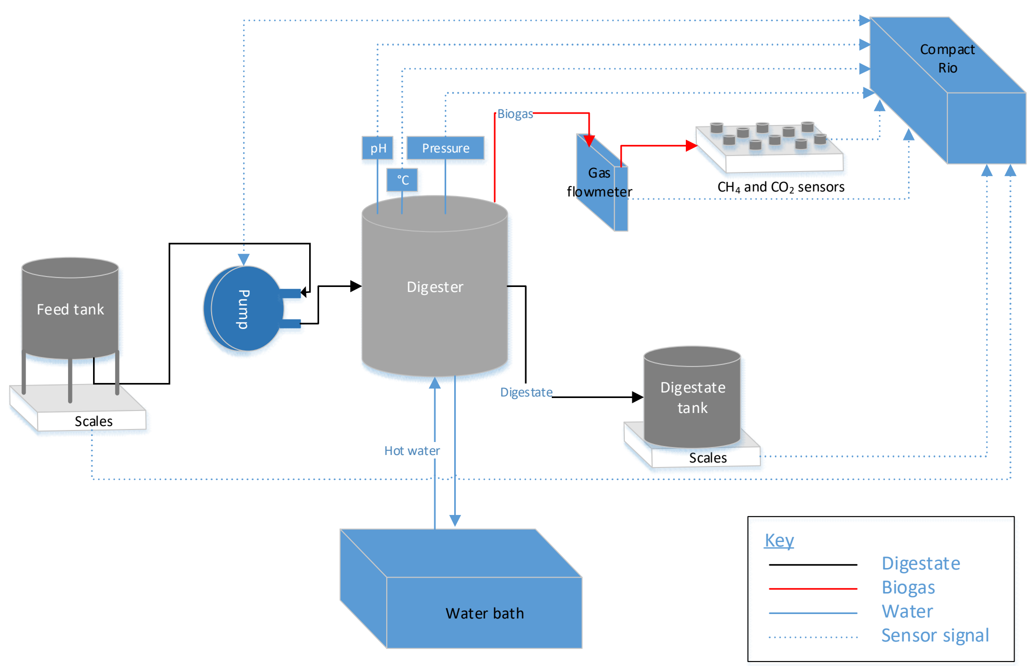


Figure 4-1: General layout of the automatically controlled laboratory-scale AD plant.

The plant was constructed on a stainless steel bench, reinforced with extra steel struts to ensure that it could hold the weight of the equipment and its contents. The main equipment was placed on the bench top, and the water bath and control system on the lower shelf. The control system was positioned so that it would not be vulnerable to liquid leaks from the bench top. The equipment was positioned to minimize pipe run lengths for feed, digestate, water and biogas and also minimize electrical cable run lengths, to reduce uncertainties and error, and the possibility of blockages (figure 4-2).

Figure 4-2: The experimental equipment set up in the laboratory.

4.1.2 Equipment

The equipment was sized by creating a mass balance of the system (Appendix A), which calculated the throughput of feedstock in VS per day and litres per day, expected total solids content of the digesters and biogas output.

Table 4-1: Plant equipment.

Equipment	Make and model	Details
Control system	CompactRIO running LabVIEW National Instruments, USA	24V DC. An 8-card control system containing the following input cards: - NI 9205 ± 10V voltage input module - NI 9238 ± 500mV voltage input module - NI 9203 ± 20A current input module - NI 9403 digital I/O module - NI 9870 RS232 input module - NI 9216 temperature input module - NI 9260 0-30V voltage output module
Feed tank Digestate tank 1 Digestate tank 2	Hanningfield stainless steel drums with sealable lids	5-litre sealable drums, modified with feed outlet/inlet and gas outlet/inlet points.
Water bath	Stabletemp WB80 water bath Cole-Parmer, UK	240V AC, 18 L capacity.
Water bath pumps	Xylem LVM Centrifugal Pump	24V DC, 18 L min^{-1}
Feed tank stirrer motor	Crouzet geared motor	12V DC, 20 r.p.m.
Feed pump	Verderflex Dura 10 peristaltic pump Verder, UK	With inverter. 5 to 70 r.p.m., 4 to 97 L hr^{-1} flow.
Digester 1 Digester 2	CSTR-10S Bioprocess control, Sweden	10-litre stainless steel digester with built-in stirrer, feed tube and connections for water heating. Lid modified to add fitting for temperature and pressure sensors (both ¼" BSP). Insulation added on both digesters to regulate temperature.

Some of the equipment was modified, as follows:

- A motor/stirrer was added to the feed tank lid, to stir the feed.
- A 6mm hose tail fitting was added to the feed tank and digestate tank lids to connect the tubes for biogas collection.
- A ½" BSP outlet/drain point was added to the feed tank to allow the feed to flow to the feed pumps.
- A ½" BSP hose tail fitting was added to the digestate tank lids to enable the digestate to flow into the digestate tanks, with an air gap separation.
- A 13mm hole was made in the feed tank and digestate tank lids, which would be loosely sealed with a bung, to form an emergency pressure release point.
- Two ¼" BSP connection points were added to each of the digester lids, for the temperature and pressure sensors.

Figure 4-3: The modified feed tank.

Figure 4-4: The modified digester.

Figure 4-5: The modified digestate tank.

4.1.3 Instrumentation

The system design included five methane sensors, five carbon dioxide sensors, two pressure transducers, two temperature sensors, two pH sensors, three sets of weighing scales and two gas flowmeters (figure 4-6).

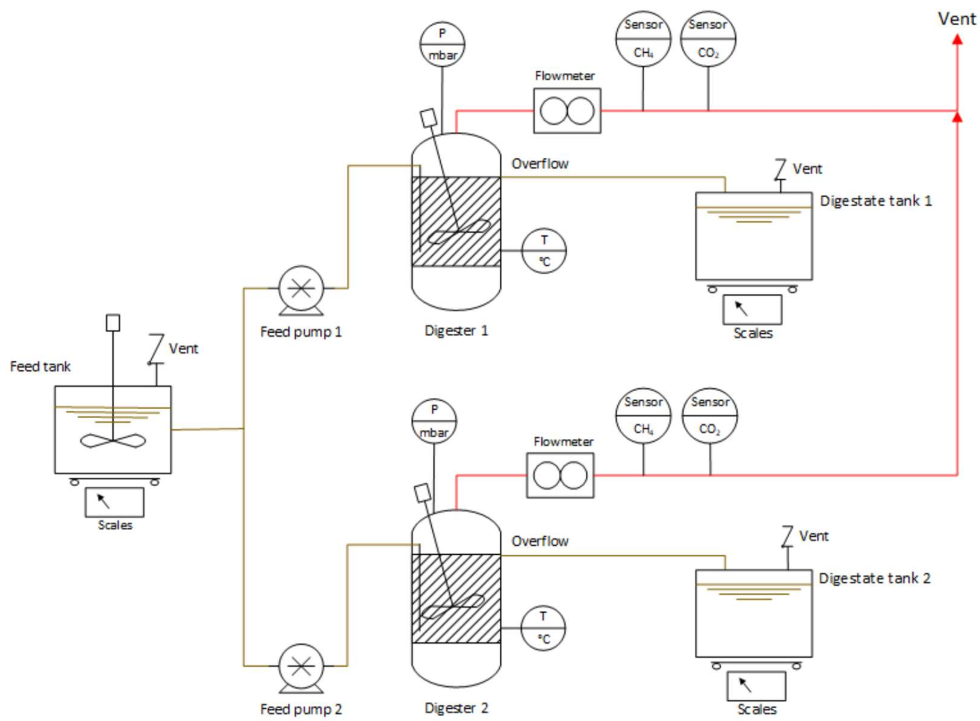


Figure 4-6: Piping and instrumentation diagram of the automated lab-scale AD plant.

The instrumentation on the plant (table 4-2) was chosen for robustness and accuracy, due to the long duration of the experiment and the chemically and biologically hostile environment.

Table 4-2: Instrumentation for the experimental rig.

Instrument	**Location**	**Make and model and range**	**Signal, range and accuracy**
Weighing scales	Feed tank Digestate tanks 1 and 2	CKE 16K0.1 Kern, Germany	RS232 connection 16 kg ± 0.1g
pH sensor	Digesters 1 and 2	PHE-4830 pH sensor (Omega, USA) with IXIAN pH Transmitter (Atlas Scientific, USA)	0 to 14 ± 0.02 pH range, temperature 0-100°C 4-20mA signal
Methane sensor	Sensors box – detecting methane levels from digesters 1 and 2 and digestate tanks 1 and 2	P/HCP/NC/5/V/P, 5-pin platinum IR hydrocarbon sensor Dynament, UK	0-100% methane, ±0.1% 0.4V to 2.4V linear signal
Carbon dioxide sensor	Sensors box – detecting methane levels from digesters 1 and 2 and digestate tanks 1 and 2	P/HCO2/NC/5/VP, 5-pin premier high-range carbon dioxide sensor Dynament, UK	0-100% carbon dioxide, ±0.1% 0.4V to 2.4V linear signal
Pressure transducer	Digesters 1 and 2	PXM309-0.07GI Omega, UK	4-20 mA 0-70 mbar ±0.25%
Temperature probe	Digesters 1 and 2	Industrial thermocouple, enclosed in a stainless steel probe Omega, UK	-100°C to 400°C
Gas flowmeter	After digesters 1 and 2	µflow ultra-low flow flowmeter Bioprocess control, Sweden	Approx. 10mL resolution 4-20 mA with 1% repeatability 20 to 4000 mL/h

All instrumentation was connected to the control system, a CompactRIO PLC controller (National Instruments, USA). The system was controlled through a selection of input cards, specific to the signal type (table 4-1) and operated using a LabVIEW control system. The power to the experimental rig was provided at different voltages via a network (figure 4-7).

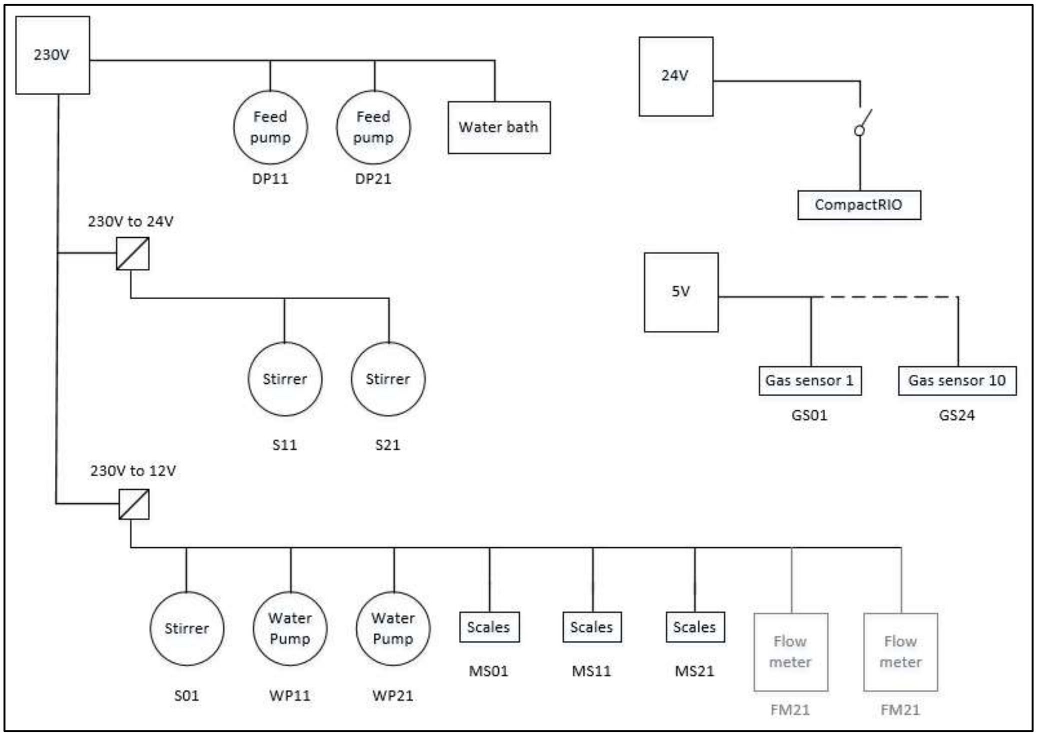


Figure 4-7: Electrical configuration of the experimental rig.

Sensors box

The gas sensors were housed in a purpose-built box to ensure optimal performance and minimise failure rate of the gas sensors (figure 4-8, figure 4-9).

Figure 4-8: A gas sensor, gas sensor with connector, gas sensor housing and gas sensor housing containing a gas sensor.

Figure 4-9: The gas sensors box.

The sensors box was constructed so that the sensors were pointing downwards with the gas entering from underneath, to minimise the possibility of water ingress by water condensed from the biogas.

4.1.4 Readings and data logging

The system was set up to take measurements of each of the inline sensors (weighing scales, gas flow meters, pH sensors, temperature sensors, pressure transducers) every 2.5 seconds and logged in the log file.

4.2 Equipment commissioning

4.2.1 Initial commissioning – system setup and verification

The commissioning period for the plant lasted for approximately 3 months. The equipment was initially dry tested, then tested with water and pressure tested to ensure that they were gas-tight. The digesters were then emptied and filled with digestate from a local food waste AD plant. The digestate was sieved and thoroughly mixed before adding to the digesters.

After the digesters had been filled, the feeding and heating systems were started. The temperature of the digesters was set to 37°C. The feed was started at an OLR of 0.4 gVS L^{-1} day^{-1} and ramped up at a rate of 0.2 gVS L^{-1} day^{-1} to 2 gVS L^{-1} day^{-1}. Verification of the system readings (temperature, biogas flow, pressure, gas composition, scales reading) was performed by comparison with known values.

Following the initial commissioning phase, further tests were performed to determine the optimal setup of the equipment.

4.2.2 Optimal feed volatile solids

The feedstock, ground dog biscuits, was reconstituted using deionised water and mixed thoroughly to produce a suspension. The solids content of the suspension affected the thickness and settling properties of the feed. To determine the optimal solids content of the feed, a number of feedstocks were made up at a range of 10 to 22% VS (figure 4-10). The samples are labelled with their %VS, from 10 to 22, left to right.

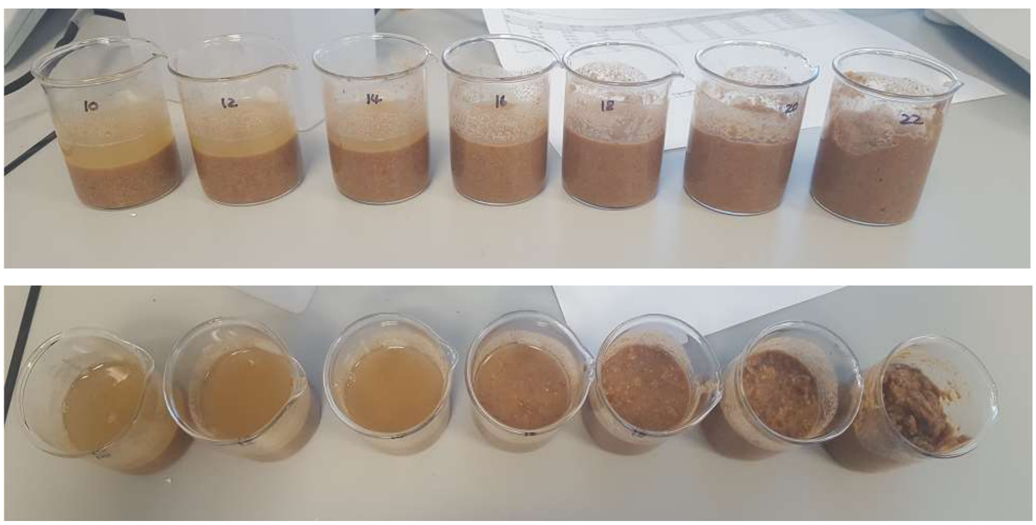


Figure 4-10: Side view and top view of feedstock at 10 to 22 % VS.

The consistency of the feedstock at 18, 20 and 22% VS was very thick, and would not be easy to pump as it would not flow easily. The samples at 10, 12 and 14% feedstock had a relatively large ‘watery’ fraction when settled and therefore would separate more easily. It was therefore concluded that the sample at 16% VS was best suited as a feedstock.

4.2.3 Settling test

During experiments, the 16% VS feedstock was made up in batches of 1000 to 2000 mL, with the feed tank being refilled every 2-6 days. There would be several hours between feed events, in which time settling would be expected if the mixing system was not sufficiently thorough. To test the mixing systems, a period of 'settling time' would therefore need to be left between feeding events. To determine how long the feedstock took to settle, the 'settling time' (the time to separate into liquid and solid fractions) was tested (figure 4-11). Rather than a 16% VS feedstock, a 14% VS feedstock was used for this test as it would produce a more distinct separation on settling.

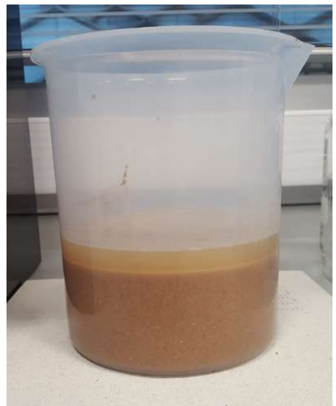


Figure 4-11: Settled feedstock at 14% VS, after 5 minutes of settling time.

Three different container types were tested, containing the same volume of made up feedstock at 14% VS. The feedstock was made up by mixing 250g of crushed dry dog biscuits with 1250 mL deionised water, stirred thoroughly, left for 10 minutes to allow the dry feed to absorb the water, and then stirred again. After stirring, a timer was immediately started and the depth of the top layer was noted each minute until it reached a steady state. The settling time for each container was then noted (table 4-3).

Table 4-3: Settling times for different containers.

Container type	Surface area (cm^2)	Settling time (min)
Measuring cylinder	63.6	14
Large plastic beaker	196.0	9
Rectangular tub	563.8	2

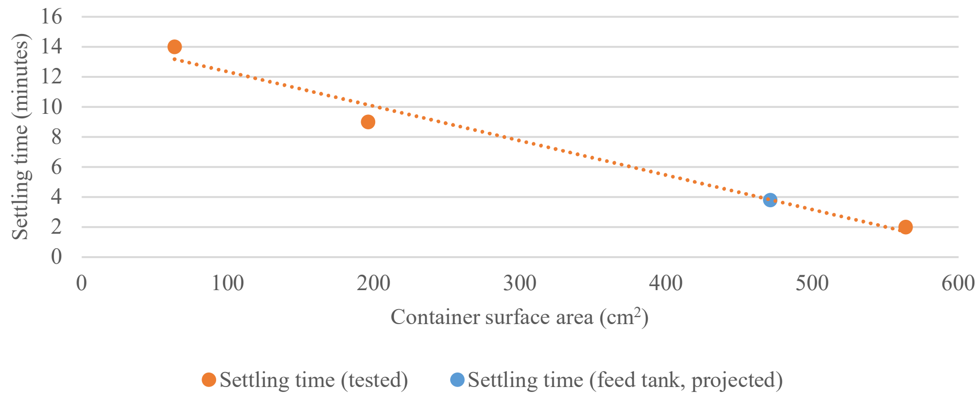


Figure 4-12: Settling time for different types of container holding 14% VS feedstock.

The settling time decreased with surface area in a roughly linear pattern (figure 4-12). From this, a settling time for the feed tank of 3.8 minutes was estimated. This time was used as a 'pause' period between mixing events in the mixing test (section 4.2.4).

4.2.4 Testing the mixing system

To ensure the VS and TS of the feed remained constant throughout the experiment, it was important that the feed tank was well mixed. It was observed that the original mixing system (figure 4-13) allowed the feed to separate in the feed tank, therefore variations of the mixing system were tested to ascertain which gave the best performance.

Figure 4-13: Original mixing system (mixing system 1).

To test the feed consistency at different feed events, a 'dummy run' of 16 sequential feed events was performed, with 4 minutes of 'settling time' between each feed, to simulate all the feeding events from a single batch of feedstock. Samples from each feed event were collected, and the total solids content (TS) of each sample was tested. Two alternative mixing systems were tested, named mixing systems 2 and 3 (figure 4-14, figure 4-15).

Figure 4-14: Mixing system 2.

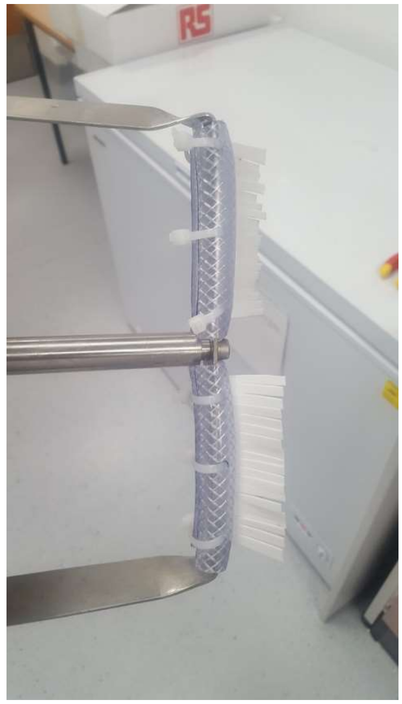


Figure 4-15: Mixing system 3.

The results of the mixing tests showed the variation in VS between samples (table 4-4).

Table 4-4: Total solids for each feed event for a single batch of feed.

Feed event	Mixing system 2 Feedstock TS (%)	Mixing system 3 Feedstock TS (%)
1	16.9%	17.2%
2	15.5%	16.1%
3	15.0%	16.1%
4	14.9%	16.2%
5	14.3%	16.3%
6	14.9%	16.4%
7	14.9%	16.7%
8	14.9%	16.0%
9	14.9%	15.6%
10	13.3%	15.5%
11	14.0%	15.4%
12	14.1%	15.2%
13	14.3%	15.0%
14	13.8%	14.9%
15	15.7%	14.5%
16	-	14.6%

The standard deviations of the TS during the experiments were 0.88% for mixing system 2, 0.77% for mixing system 3, which gives an indication of the uniformity of TS over the course of a batch of feeding (figure 4-16).

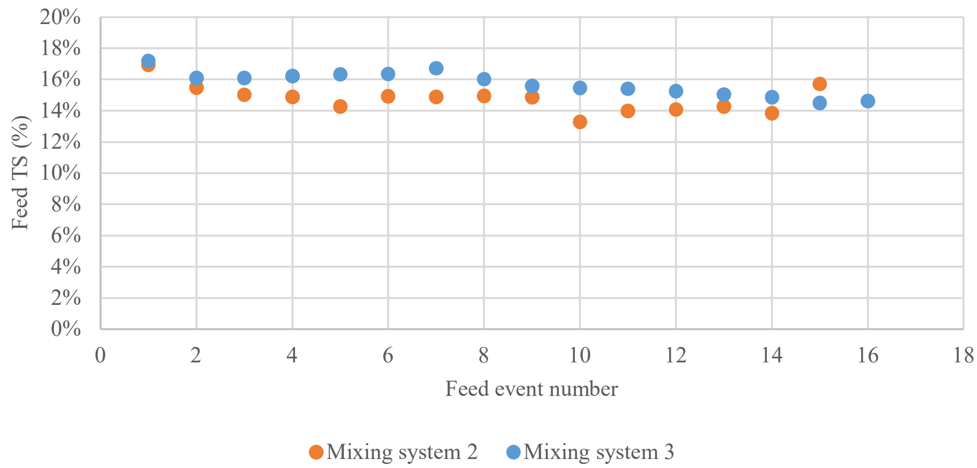


Figure 4-16: Total solids content (%) at feed events 1 to 16 for mixing systems 2 and 3.

From this data, it was concluded that mixing system 3 was suitable for providing a uniform feed. Although the total solids showed a reduction over the course of the test, the reduction was far more gradual than for mixing system 2. It was also noted that further improvement would require changes to the motor and paddle equipment, which was not feasible within the experimental time frame.

4.2.5 Data collection frequency

The system was set up to take measurements from each of the inline sensors (weighing scales, gas flow meters, pH sensors, temperature sensors, pressure transducers) every 2.5 seconds. For data analysis, the readings were to be averaged over a set time period. A variety of averaging periods were tested to determine the best averaging frequency (figure 4-17 to figure 4-21). Biogas production (flow, in mL hr^{-1}) was used as the example measurement to compare, as it was the most variable reading in the system.

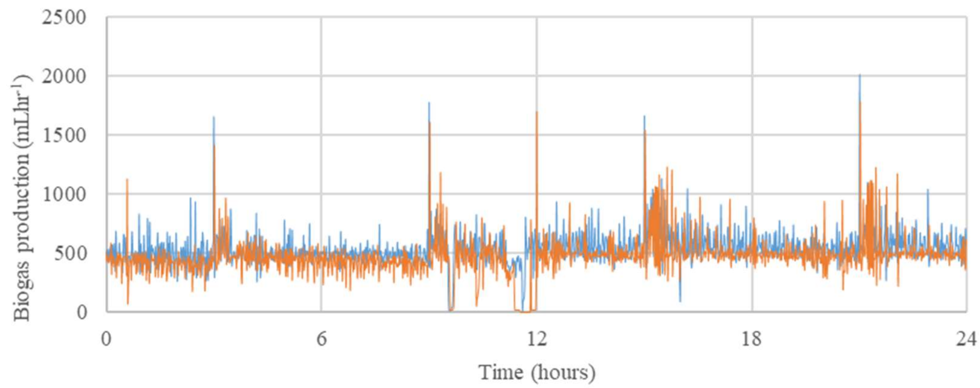


Figure 4-17: Biogas production in digesters 1 and 2 averaged at 1-minute intervals.

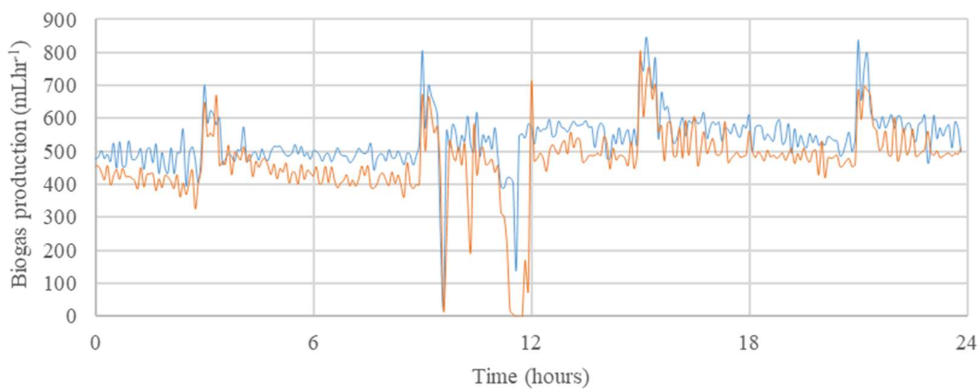


Figure 4-18: Biogas production in digesters 1 and 2 averaged at 5-minute intervals.

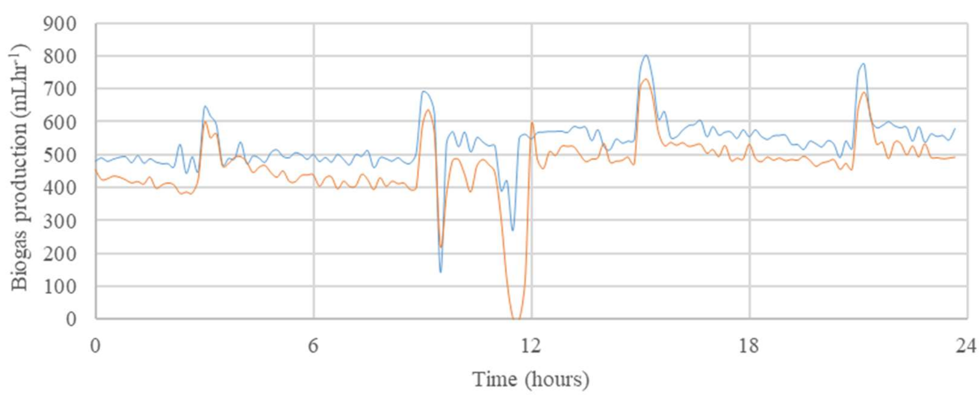


Figure 4-19: Biogas production in digesters 1 and 2 averaged at 10-minute intervals.

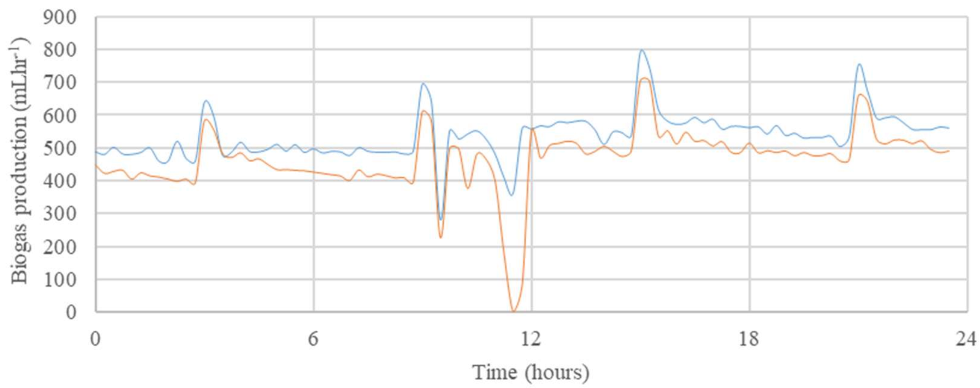


Figure 4-20: Biogas production in digesters 1 and 2 averaged at 15-minute intervals.

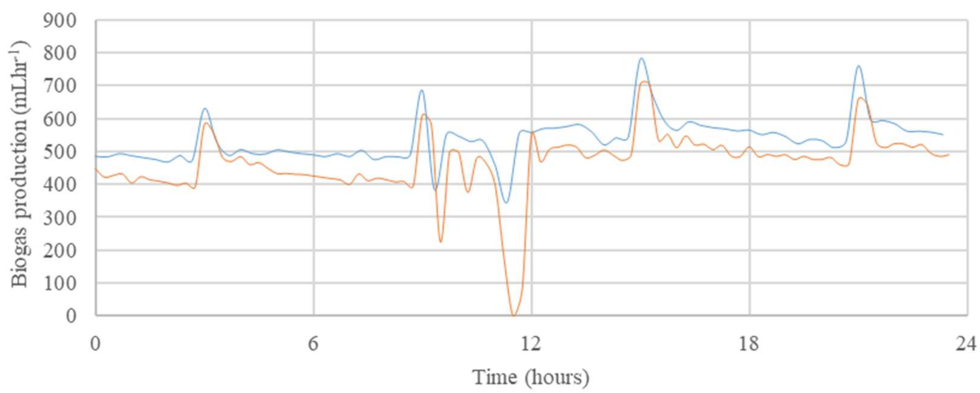


Figure 4-21: Biogas production in digesters 1 and 2, averaged at 20-minute intervals.

When the biogas flow data is averaged at a frequency of 1 minute or 5 minutes, the signal produced was too noisy to see a distinct pattern. At an averaging frequency of 20 minutes, some of the fine detail of the signal is lost. The best averaging frequencies were 10 and 15 minutes, which showed good detail without excessive noise. For the experimental work, a logging frequency of 15 minutes was chosen as it would limit the number of readings being stored and processing memory required therefore would reduce the likelihood of running out of disk storage space and reduce processing time.

4.2.6 Alkalinity during commissioning

During the commissioning period, the partial alkalinity (PA) and total alkalinity (TA) was tested every weekday (figure 4-22) and the alkalinity ratio (PA/TA) was calculated (figure 4-23).

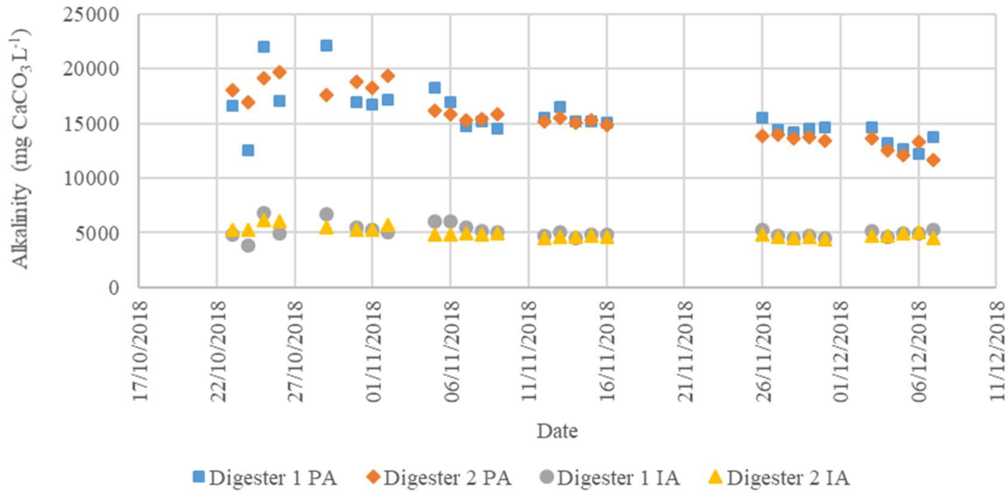


Figure 4-22: Digesters 1 and 2 partial and intermediate alkalinity ratio during commissioning.

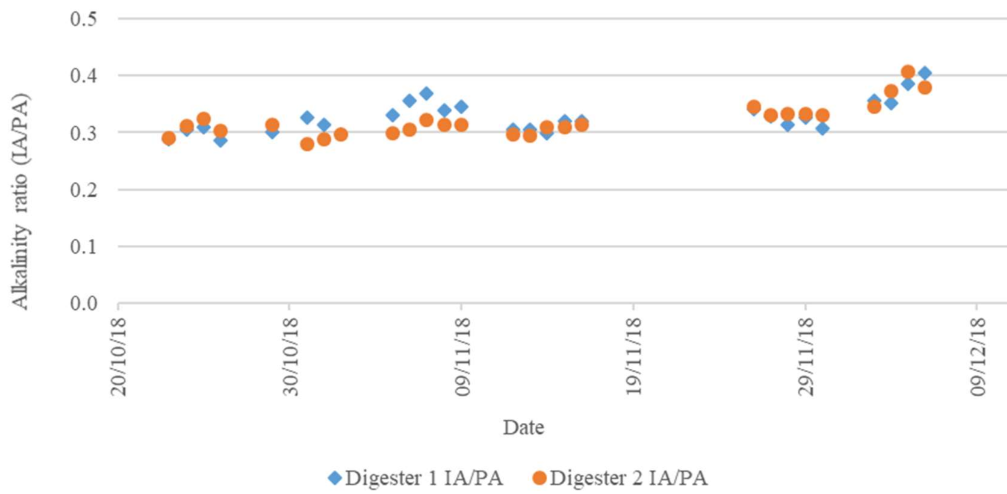


Figure 4-23: Alkalinity ratio (IA/PA) for digesters 1 and 2 during commissioning.

The PA and IA were both erratic at the start of the commissioning period for approximately two weeks, and then settled to a steady level in both digesters. During this time, the alkalinity ratio was relatively stable and within the acceptable range (around 0.3). After a further four weeks, the alkalinity ratio started to rise and at the end of the commissioning period was at a level of 0.4 in both digesters, which suggests that the digesters were becoming unstable. This is accompanied by a drop in the partial alkalinity, which indicates the amount of bicarbonate ions in the digester had dropped (Ripley, Boyle and Converse, 1986).

The cause of the instability could have been that the feeding was continued when the digester temperature was low (section 4.2.7), which would have led to a reduced rate of digestion, then a build-up in feed and subsequent overloading and instability.

4.2.7 Temperature during commissioning

At the start of commissioning, the pressure and gas production in the experimental rig were found to be very sensitive to changes in temperature – a drop in temperature causing a drop in pressure and gas production, with a very slow recovery. For this reason, the rig was altered to keep the temperature steady to within ±0.5°C and the sampling system was also altered to minimise any pressure drop whilst sampling.

Towards the end of the commissioning, the heating system developed a fault and the temperature of the digesters dropped from mesophilic (38°C) to room temperature several times but the feeding cycle was not stopped (figure 4-24).

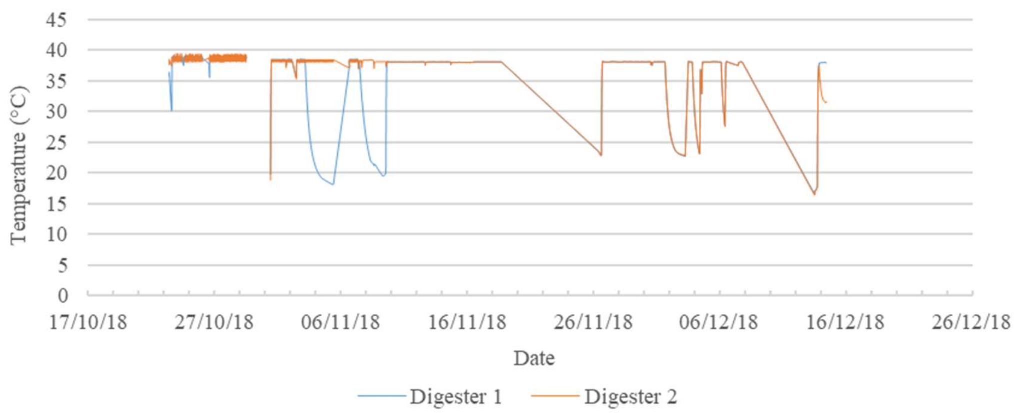


Figure 4-24: Temperature of digesters 1 and 2 during commissioning.

When plotted with the alkalinity ratio during this time, it was evident that disturbances in the alkalinity ratio occurred at roughly the same time as the temperature dropped (figure 4-25).

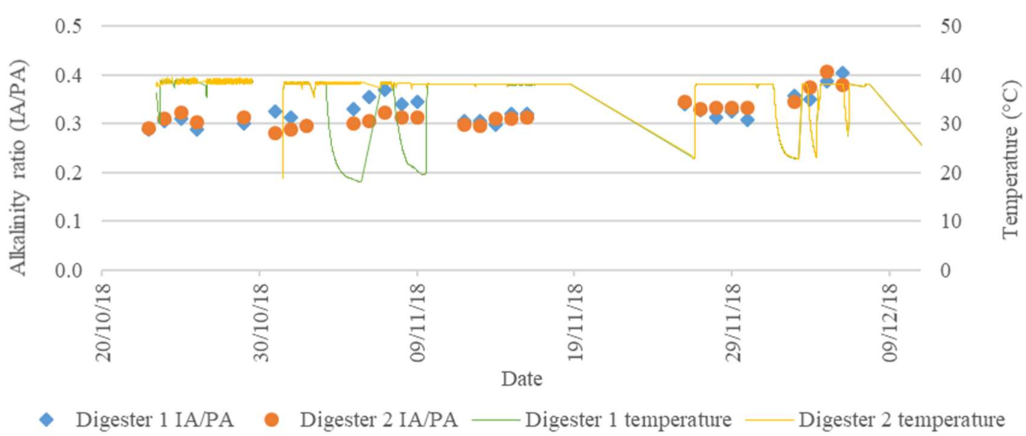


Figure 4-25: Digester IA/PA and temperature during commissioning.

In particular, there were two temperature drops for digester 1 and not digester 2 on the 2nd and 7th November and a corresponding rise in alkalinity ratio in digester 1 but not digester 2. There were no other disturbances that could explain the rise in alkalinity ratio: the feed type and OLR were the same, the mixing regime was unchanged and no inhibitors were introduced. It is known that digesters operating on food waste can experience inhibition due to a lack of trace elements and a build-up of ammonia, but that this normally happens after a longer period, approximately a year (Walker *et al.*, 2017).

A constant temperature in the digester was therefore understood to be of key importance, and feeding should not continue if the digester temperature dropped below 30°C (the lowest point of the optimal operating range for mesophiles (Gerardi, 2003e)). A safeguard was added to the control system to prevent feeding if the temperature dropped below 30 °C.

5 Flexible feeding of an anaerobic digester

5.1 Introduction

The experimental section of this thesis presents an investigation into the effect on an anaerobic digester of a feed regime that varies widely in its loading rate. As discussed in the literature review, the response of digesters to a variable feed has been investigated using different feedstocks such as maize, rye, sugar beet, grass silage and carrots (Hahn *et al.*, 2014b; Laperrière *et al.*, 2017; Mauky *et al.*, 2015). Variable feeding has been shown to increase microbial community diversity and resistance to toxicity and shock loads (De Vrieze, Verstraete and Boon, 2013) and that a high degree of flexibility is achievable – from 25% to 400% of the digester's normal gas production at organic loading rates of 1.5 to 3.5 gVS L^{-1} day^{-1} (Mauky *et al.*, 2015; Laperrière *et al.*, 2017).

The purpose of this experimental work was to add to previous research by examining more closely the effect on a digester of a fluctuating feed load in both the short term and the long term. This was done through measurements of the volatile fatty acids, biogas production, alkalinity, total and volatile solids, biogas composition and biological methane potential (BMP). The experiment used food waste, as it is a feedstock of growing interest and availability in the UK. This is evidenced by the introduction of the landfill tax in the UK in 1996, with a higher rate for non-inert waste, which has steadily increased and is currently £91.35 per tonne of non-inert waste (GOV.UK, 2019a). Additionally, the recent publication of the circular economy package by the EU (Moore, 2018) has prompted the publication of the new UK government waste and resources policy for England (DEFRA, 2018), which has set out a time scale to make separate food waste collections mandatory throughout the UK by 2023, which will increase food waste availability for further use. This could lead to more installations of smaller, local food waste digesters.

The work used an experimental digester running on a variable feed rate, and a control digester running simultaneously with the same feedstock but a constant feed rate, to obtain comparison data. The methane production capability of the digestates from both digesters to was assessed in a BMP (biological methane potential) test at the beginning and end of the experimental period; this has not been reported in previous work. This could in turn add to the current understanding of the mechanism of biogas production and its relation to feed rate in anaerobic digesters.

5.2 Experimental design and methods

5.2.1 Aim

The overall aim was to ascertain how the biogas production and stability of an anaerobic digester are affected by the application of an increasing and decreasing loading rate. This was to be achieved in four stages:

- Feed two identical lab-scale digesters at a steady rate until the digesters are 'stable', as determined by laboratory testing of the gas production rate and stability indicators such as alkalinity and volatile fatty acid (VFA) concentrations.

- When stability is established in both digesters, test the flexibility in one of the digesters by altering the feed pattern to a variable feed loading rate.

- Collect data to show the differences in behaviour of the two digesters when they are subjected to different feeding patterns.

- After several weeks following this pattern, test the response of the digesters to the same feeding regime. To do this, return both digesters to a stable feed pattern and test them for any changes in parameters (for example, biogas production, methane % in biogas, VFA content) compared to the start of the experiment or differences in behaviour between the two digesters.

5.2.2 Experimental apparatus

The apparatus used and its construction and commissioning was described in Chapter 4. The experimental setup (figure 5-1) contained two identical digester tanks, which were fed via two positive-displacement pumps from a common tank containing the liquid feedstock.

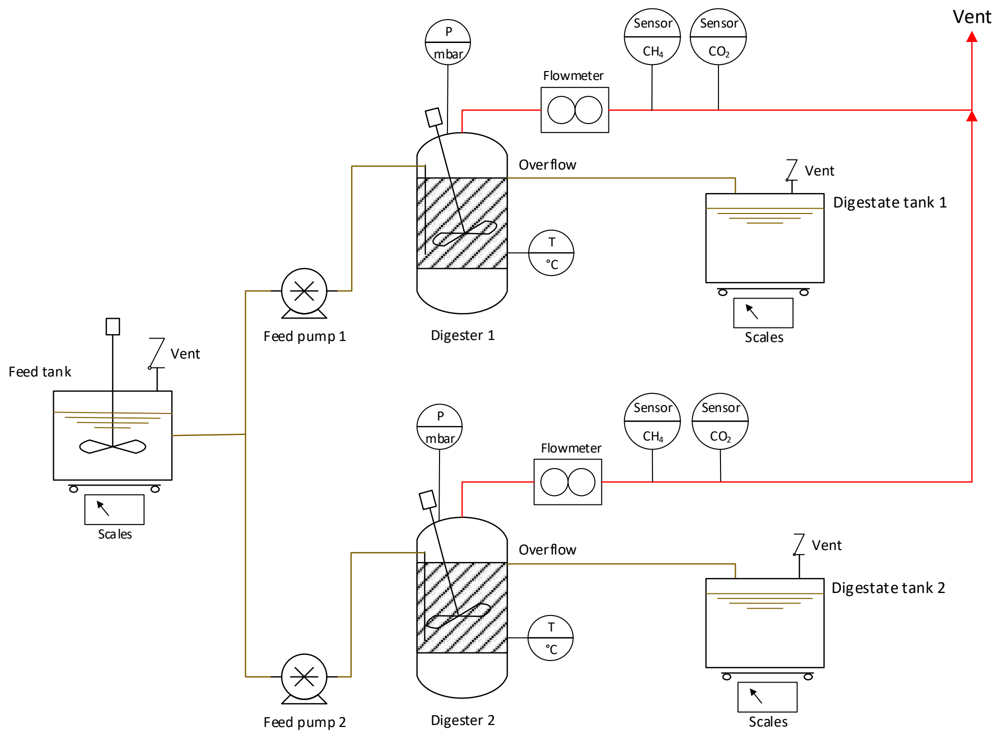


Figure 5-1: Piping and instrumentation diagram (P&ID) of the experimental setup.

The use of a common feed tank ensured that the feedstock delivered to both digesters was of equal composition throughout the experiment.

5.2.3 Feedstock choice, composition and preparation

Food waste was chosen as the experimental feedstock, as it is a common resource in urban areas. Food waste has not been extensively tested as a feedstock in flexible feeding scenarios and so the results of this experiment would contribute to the existing body of knowledge in this area. Additionally, as discussed in the introduction, food waste is expected to become more readily available as a feedstock in the UK, because of the introduction of new waste and resources legislation (section 1.6).

To enable a consistent feed composition (volatile and total solids, nutrient content) throughout the testing period, dry animal feed was used as a representative 'synthetic food waste' (SFW). This feedstock has been used in previous studies (VanderGheynst, Gossett and Walker, 1997; Lemus *et al.*, 2004) and has the advantages over real food waste that it is more stable during storage and has a reliably consistent composition, therefore the experiment will contain less inherent margin for inconsistency. It is also easier to handle and pump, which was necessary for an automated system in order to ensure smooth running of the system and prevent blockages, which could also introduce error.

The dry animal feed (SFW) was sourced from Wagg Foods Ltd, UK. When a new bag of dry feed was used, it was treated as a new feedstock and the feedstock tests were repeated (Table 5-3). Two different bags of the 'Complete dog food mix with Chicken and Vegetables' were used during the experimental period, the first ('A') from day 1 to day 114 (part way through phase 4), and the second ('B') from day 115 onwards. The ingredients and composition lists provided by the manufacturer on the different bags of SFW were slightly different (table 5-1, table 5-2).

Table 5-1: Contents listed in order of total % by weight of SFW mixes 'A' and 'B', as given by the manufacturer (Wagg Foods Ltd).

SFW mix 'A'	SFW mix 'B'
Wheat	Cereals
Meat Meal (min 10% beef in red kibble)	Meat and Animal derivatives (24.5% including 4% in chicken disc)
Wheatfeed	Oils and fats
Maize	Derivatives of vegetable origin
Poultry Fat	Vegetables (4% pea in pea disc)
Digest	Yeasts (MOS 0.1%)
Linseed	Citrus extract (0.05%)
Beet Pulp	Yucca extract (0.015%)
Rice	
Peas (min 4% in pea kibble)	
Lucerne	
Minerals	
Yeast (0.08%)	
Citrus Extract (0.04%)	
Yucca Extract (0.01%)	

Table 5-2: Content of SFW mix in order of weight given by the manufacturer (Wagg Foods Ltd).

Constituent	% by weight, SFW mix 'A'	% by weight, SFW mix 'B'
Protein	21%	20%
Fat Content	8%	8%
Crude Fibre	3%	3.5%
Crude Ash	8.5%	8.5%
Omega 6	1.4%	1.5%
Omega 3	0.3%	0.3%

The two batches of complete dog food mix were very similar as quoted on the packaging, but both were tested further to confirm that they were sufficiently similar to be treated as the same feedstock. Analysis of the feedstock composition, with a comparison to food waste, is provided in table 5-3.

Comparison with food waste

40 kg of food waste was collected from a university canteen, then separated into categories, and each category was weighed (figure 5-2).

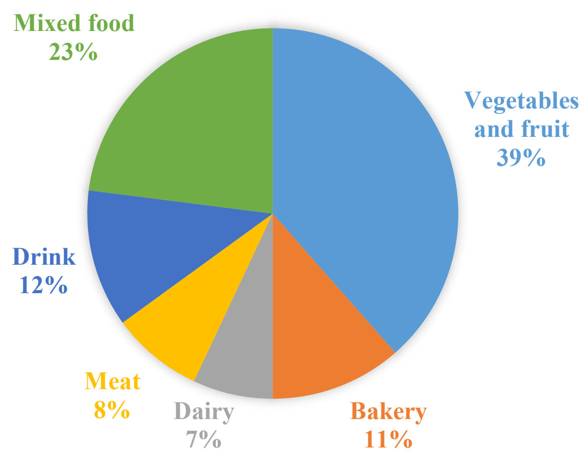


Figure 5-2: Percentage composition of food waste sample (by weight).

The food waste sample was then homogenised to a thick liquid within 6 hours using a food processor and a mincer (aperture size 6mm), with the final particle size of <0.5mm, and immediately stored at -18°C. The characteristics of the food waste were compared against the SFW (Table 5-3). For volatile and total solids, the sample sizes were 5 g, for BMP, approximately 11 g. For the CHNS test, approximately 5 g of the sample was dried, then finely ground using a pestle and mortar, and 5mg of this used for each sample.

Table 5-3: Compositional analysis of the two dry animal feeds (SFW) and food waste.

	Food waste	**SFW A**	**SFW A difference from food waste**	**SFW B**	**SFW B difference from food waste**	**Typical value (Banks *et al.*, 2018)**
Carbon (% TS)	50.0±1.3	44.9±0.7	-10.2%	45.0±2.5	-10%	52
Hydrogen (%TS)	7.1±0.3	6.3±0.1	-11.3%	6.2±0.1	-12.6%	6.9
Nitrogen (%TS)	3.8±0.1	3.8±0.1	0%	3.5±0.1	-7.9%	3.4
Sulphur (%TS)	0.2±0.3	0.5±0.4	+150%	0.2±0.3	0%	0.3
Oxygen (%TS)	39.0±1.6	44.5±0.4	+14.1%	45.0±2.3	+15.4%	38
Volatile solids (% WW)	14.7±0.1	83.5±0.6	n/a	83.9±0.9	n/a	22
Total solids (% WW)	15.3±0.1	92.9±0.4	n/a	94.7±1.1	n/a	24
Calorific value (kJ gTS^{-1})	19.88±1.10	17.97±0.12	-9.6%	17.27±0.10	-13.1%	22
Biological methane potential (BMP) (mLCH_4 gVS^{-1})	471.2±19.7	374.0±7.5	-20.4%	-	-	450

The nitrogen content of the SFW, indicating the relative protein content, is the same or slightly lower than the food waste. The hydrogen and carbon of the SFW, indicating the fats and carbohydrates, is 10-12% lower than the food waste, and the oxygen content is 14-15% higher. The sulphur content is roughly the same in food waste and SFW but the uncertainty is very high (over 100% of the average sulphur content, which means that the sulphur content could be double or zero of the average), so this result is not statistically significant. The biological methane potential (BMP) of the SFW is 20.4% lower than that of the food waste. These results collectively show that the SFW is similar to food waste, but generally of a lower nutritional quality. A review of published literature stated that the range of BMP of food waste has been reported between 160 and 530 mL CH_4 gVS^{-1} (Xu *et al.*, 2018). Both samples of SFW are within this range and can be considered valid as an example food waste.

The impact on the experiment that lower carbon, nitrogen, hydrogen and oxygen of the SFW would have might be a lower production of biogas than would be expected with food waste or possible failure from lack of nutrients or micronutrients.

To prepare for feeding into the digester, the dry SFW was ground using a food processor into small (~1mm diameter) particles, in order to ensure it was fully homogenised when taking small sample amounts. After grinding, the SFW was stored at -18 °C to prevent degradation. The SFW was defrosted and rehydrated as required with deionised water to a volatile solids content of 16%, to make it easy to pump and of a similar dry matter content to standard food waste (Zhang *et al.*, 2014; Xu *et al.*, 2018).

After reconstitution with water, a trace element solution (table 5-4) was added to the SFW, to avoid process inhibition during the experiment due to a lack of trace elements (Banks *et al.*, 2012; Zhang *et al.*, 2014). The concentration of trace element solution in the feed solution was derived from accepted practice by the inoculum provider (section 5.2.4) and from recommended concentrations in literature (Banks *et al.*, 2012; Facchin *et al.*, 2013).

Table 5-4:Trace elements concentration in solution and feed.

Trace element	Concentration in feed (mg $kgTS^{-1}$)
Manganese	12.9
Nickel	5.1
Cobalt	1.9
Iron	2.7
Zinc	1.9
Copper	1.3
Molybdenum	1.7
Tungsten	0.2
Selenium	0.2

The feedstock was added to the feed tank and used over 2 to 6 days. It was noted that there was some degradation of the feedstock during the time that it was situated in the feed tank, indicated by an acidic odour (indicating the presence of VFA) and some growth of mould. This degradation has been found to not reduce the biomethane yield in food waste, even though more VFA was produced (Aichinger *et al.*, 2015) and would therefore not significantly affect the loading rate. However, the change in VFA concentration in the feed was considered in the analysis of results.

5.2.4 Inoculum

The inoculum for the two digesters in the main experiment was sourced from an established large local food waste anaerobic digestion plant. Following extraction from the digester, the inoculum was delivered to the laboratory the following day, where it was immediately sieved using a 1mm sieve to remove any large lumps of solid matter. The inoculum was thoroughly mixed by inversion, and total solids, volatile solids and BMP tests were performed (sections 3.1.2, 3.1.3). The remaining inoculum was again thoroughly mixed by inversion and then used to fill the two digesters (9 litres each).

5.2.5 Feed pattern during the whole experiment

The study was separated into five phases (table 5-5).

Table 5-5: Experimental phases duration and description.

D1 denotes digester 1, and D2 denotes Digester 2.

Phase	Time (days)	Week number(s)	Day number(s)	Phase description
1. Ramp-up	10	1-2	1 to 9	Feed rate starting at 0.4 gVS L^{-1} day^{-1} ramping up in 0.2 step increments each day to 2 gVS L^{-1} day^{-1}.
2. Stabilisation	45	3-8	10 to 54	Constant feed rate at 2 gVS L^{-1} day^{-1} for both digesters until readings are stable.
3. Overload test	7	9	55 to 61	Feed is constant for both digesters except for an 'overload' on the second and fifth days of the week.
4. Variable feed (D1) Stable feed (D2)	74	10-20	62 to 135	Main test period, during which digester 1 is fed in a pattern of variable feed rates and digester 2 is fed at constant 2 gVS L^{-1} day^{-1}.
5. Stable feed	11	20-21	136 to 146	Both digesters fed at 2 gVS L^{-1} day^{-1}.

Phases 1 and 2: Ramp up and stabilisation

During phases 1 and 2, the digesters were both fed at the same rate, increasing from 0.4 gVS L^{-1} day^{-1} to 2 gVS L^{-1} day^{-1} from days 1 to 10 and then continuously at 2 gVS L^{-1} day^{-1} for 45 days. The purpose of these phases was to allow the digesters to acclimatise to the feed and to reach a point where the digesters were performing similarly with respect to methane yield, producing approximately the same amount of biogas per day at the same methane content.

Phase 3: Overload test

Both digesters were subject to the same loading pattern in this phase. This test lasted one week and was an average organic loading rate of 2 gVS L^{-1} day^{-1} spread evenly over the week except for two feeding 'spikes' on days 1 and 4. The OLR at these 'spikes' was 4 gVS L^{-1} day^{-1} and 6 gVS L^{-1} day^{-1}, which represented a medium load and a high load, according to the normal OLR range, 3.2-7.2 gVS L^{-1} day^{-1} (Gerardi, 2003d). The 'spikes' were spaced 3 days apart so that the biogas production had enough time to return to a normal level between them. This was based on the findings from a previous study that noted that the extra biogas production due to an overload lasted approximately 72 hours (Laperrière *et al.*, 2017). However, the previous study did not compare two digesters.

Phase 4: Variable feed period

The review of published literature on the subject of micro-scale food waste digestion found that at this scale, the feedstock supply was likely to be variable over the course of a week but steady over a year (Papargyropoulou *et al.*, 2016; Edjabou *et al.*, 2015).

An example of a real-life variable feed pattern, the output of food waste from a hotel restaurant (table 5-6, figure 5-3) (Papargyropoulou *et al.*, 2016), was used to design the variable feed pattern for digester 1.

Table 5-6: Food waste and customer number patterns from a restaurant (Papargyropoulou et al., 2016) and the derived experimental OLR design.

Day	Food waste (kg)	Number of customers	Experimental OLR
Monday	160.6	161	0.5
Tuesday	217.9	148	3.5
Wednesday	224.9	295	6
Thursday	162	243	0.6
Friday	118.5	101	0.1
Saturday	179.3	168	3
Sunday	149	89	0.3

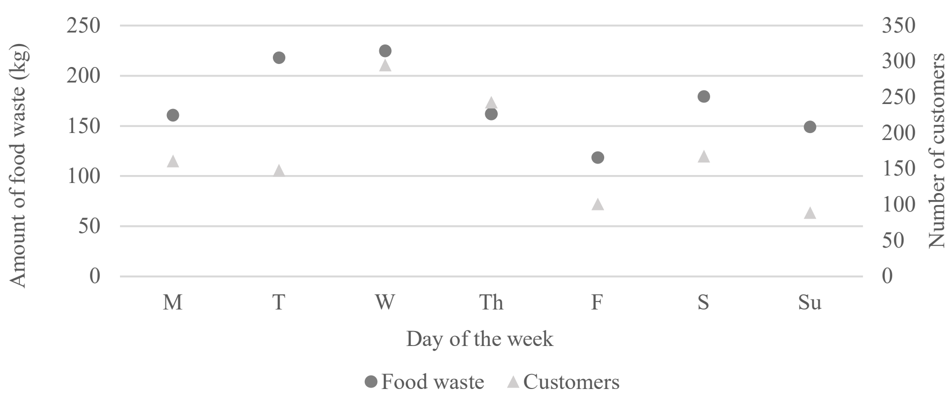


Figure 5-3: Restaurant food waste and customer numbers over the course of a week (Papargyropoulou et al., 2016).

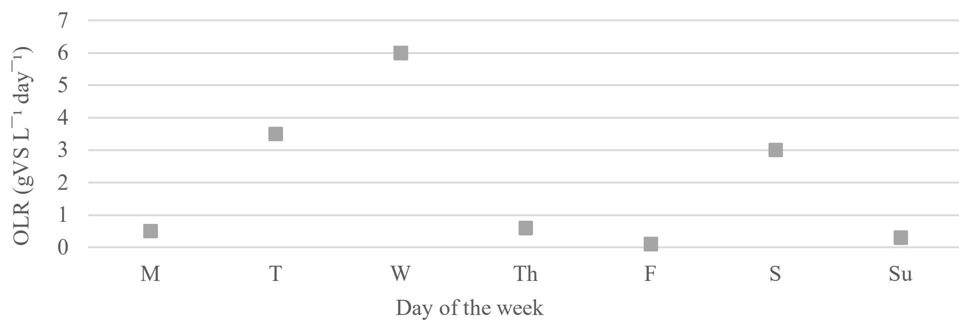


Figure 5-4: Experimental OLR design for one week of the variable feed period (phase 4).

The experimental OLR (figure 5-4) was derived from the food waste amounts and customer numbers over the course of a week (figure 5-3) and then modified so it would reach the maximum OLR recommended for a continuously-stirred tank reactor (6 gVS L^{-1} day^{-1}) (Gerardi, 2003d).

The loading pattern was calculated so that both digesters would be fed at the same average OLR (2 gVS L^{-1} day^{-1}) despite being subject to different feed patterns.

Phase 5: Stable OLR for both digesters

During phase 5 both digesters were fed at a constant OLR of 2 gVS L^{-1} day^{-1}.

Planned versus experimental feed

The planned feed pattern for the whole study is provided (figure 5-5, figure 5-7). The actual feed amounts during the experiment, measured from the difference in weight of the feed tank before and after feeding, were higher than the planned feed amounts (figure 5-6, figure 5-8). This was because there was a delay in the response to the control system by the feed pump. The feed amount was calculated to be 28.5% higher than programmed on average. The measured feed weight has been used in any calculations (as opposed to the planned feed weight), thus allowing for this ‘overfeed’.

After the ramp-up period (phase 1), the feed to digester 2 stayed approximately constant, except for the spikes on days 56 and 59 (phase 3). On days 70 to 71 (phase 4) there was an accidental overfeed in both digesters (due to an issue with the automatic feeding programme) and consequently the feed was stopped for 30 hours in both digesters to prevent digester failure. During phase 4, digester 1 was supplied with a variable feed rate, repeating in a weekly pattern, while digester 2 was fed at a steady rate. In phase 5, both digesters were fed at the same steady rate.

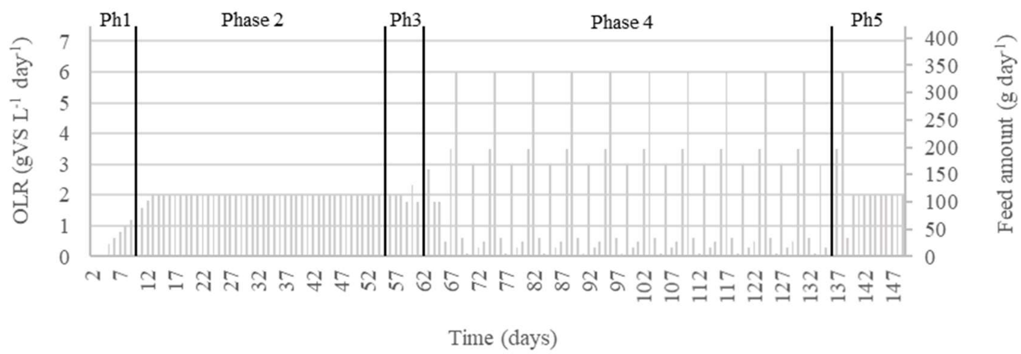


Figure 5-5: Digester 1 planned organic loading rate during the experimental period.

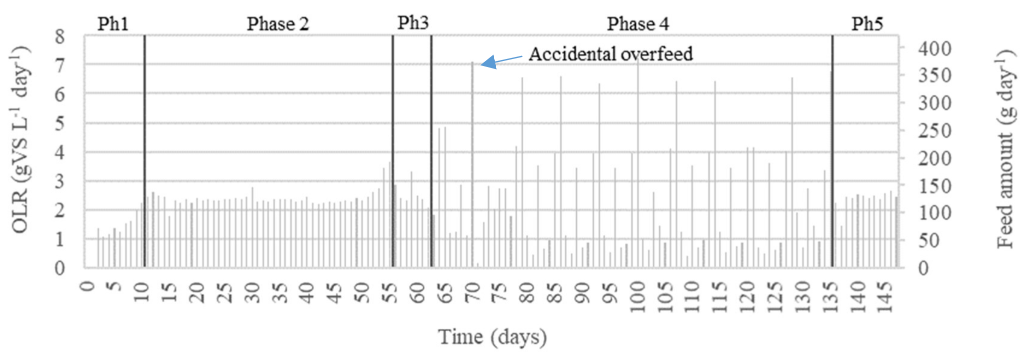


Figure 5-6: Digester 1 actual organic loading rate and feed amounts during the experimental period.

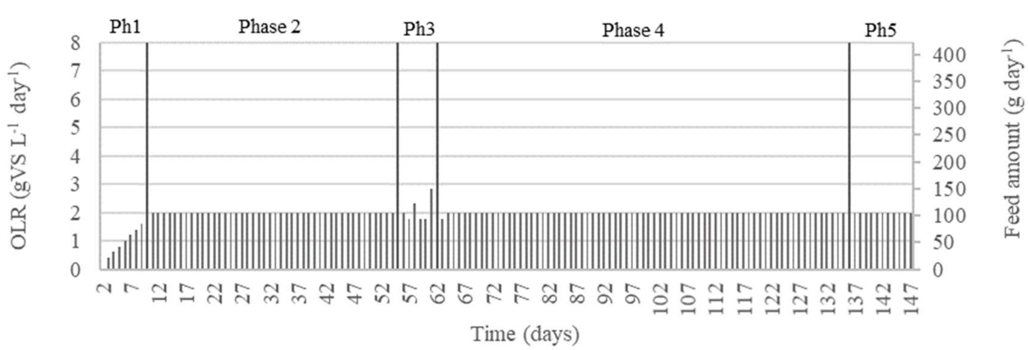


Figure 5-7: Digester 2 planned organic loading rate during the experimental period.

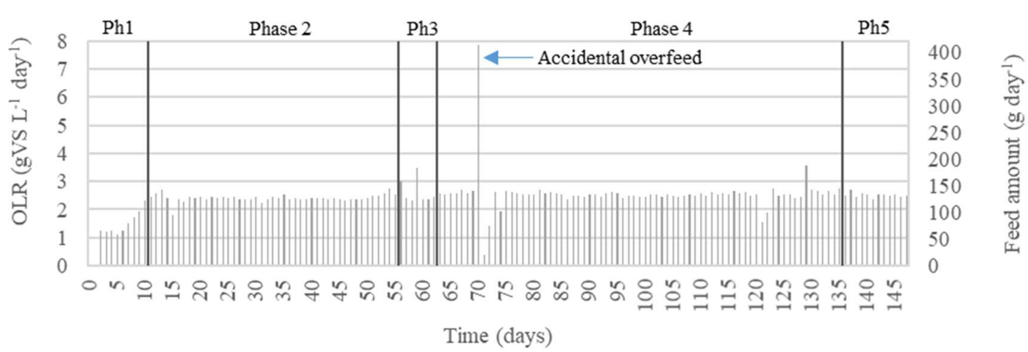


Figure 5-8: Digester 2 actual organic loading rate and feed amounts during the experimental period.

5.3 Results and discussion

5.3.1 Digester acclimatisation and VFA levels during phases 1 and 2

The acclimatisation of the digesters was shown by the acetic acid and propionic acid measurements during phases 1 and 2 (figure 5-9, figure 5-10).

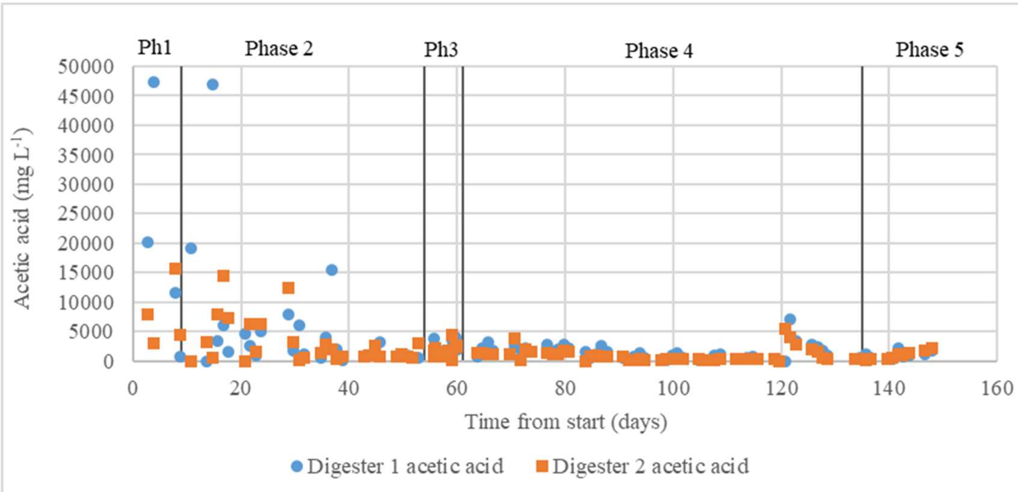


Figure 5-9: Acetic acid in digesters 1 and 2 during the experimental period.

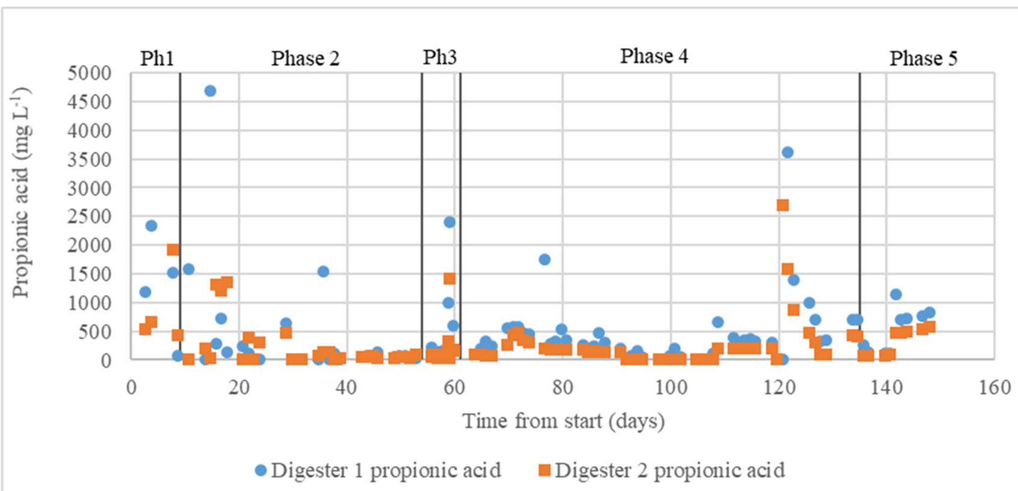


Figure 5-10: Propionic acid in digesters 1 and 2 during the experimental period.

In both digesters, the acetic acid and propionic acid levels are high and erratic for approximately the first 30 days, then reduce to much lower levels, stabilising at about 40 days, with some brief disturbances. Propionic acid is often cited as an indicator of instability in the digestion process (Gerardi, 2003c), and high levels (over 1100 mg L^{-1}) indicate that the later anaerobic digestion processes (acetogenesis, methanogenesis) are not balanced with the earlier stages (hydrolysis, acidogenesis)(Nielsen, Uellendahl and Ahring, 2007). Propionic acid is formed in the acidogenesis stage from simple substrate precursors (amino acids, sugars and fatty acids) and is consumed by the most slow-growing, sensitive VFA-degrading microorganisms in the anaerobic digestion process (Nielsen, Uellendahl and Ahring, 2007). For this reason, propionate is the slowest VFA to return to a low concentration following a disturbance.

During the experiment, regular measurements were taken of acetic, propionic, isobutyric, butyric, isovaleric, isocaproic, hexanoic and n-heptanoic acids. Isobutyric and butyric acids have been cited in previous research as good indicators of process instability (Ahring, Sandberg and Angelidaki, 1995), but the measurements in this experiment were erratic and do not show a discernible pattern, at times giving zero readings. Figure 5-11, for example, shows the butyric and isobutyric acid measurements for digester 2, which should have been consistently low from the start of phase 4 as the organic loading rate was stable throughout and the digester showed no signs of stress (its alkalinity ratio remained stable and within the normal range of 0.3 to 0.5 – see figure 5-15). However, the measurements were not stable and there was a peak of both isobutyric and butyric acids at around day 120 that is unexplained.

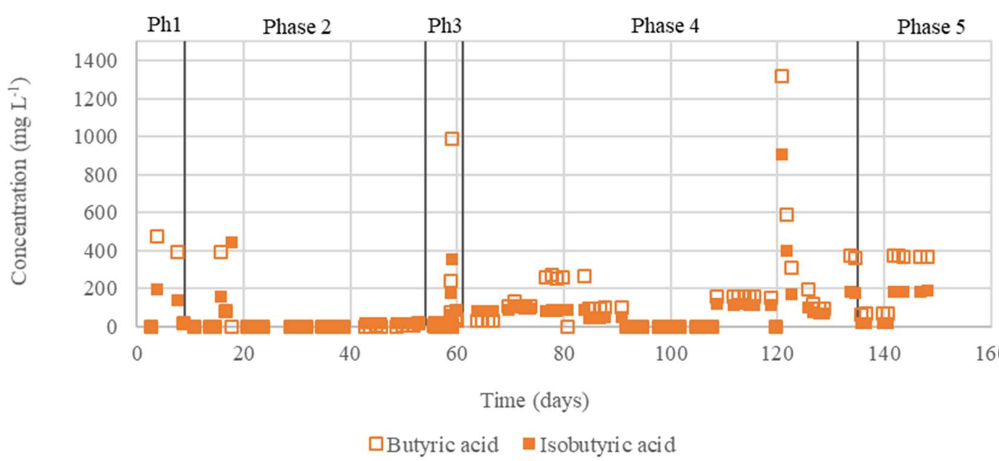


Figure 5-11: Digester 2 butyric and isobutyric acid measurements during the experimental period.

A study of stability under perturbed conditions in a digester fed with pig manure (Sun *et al.*, 2019) found that the ratio of propionic acid to acetic acid (P:A) was a good indicator of stability. This ratio was calculated using the results from this experimental study (Figure 5-12).

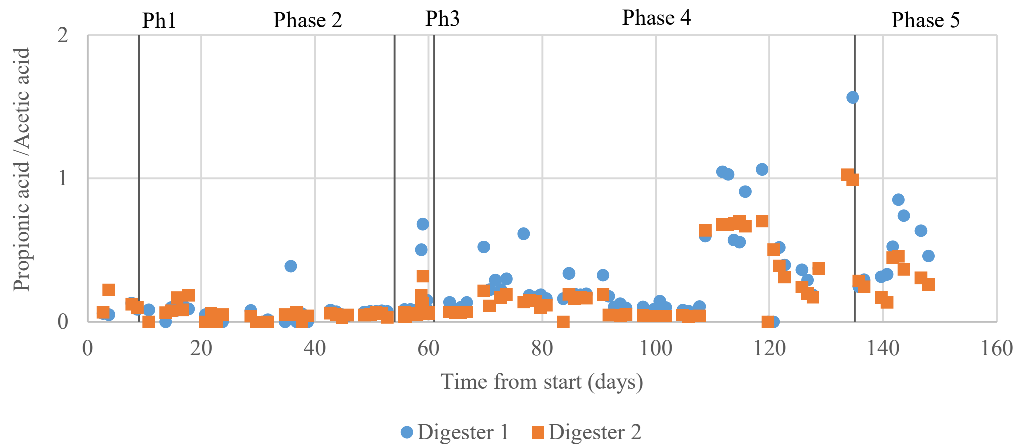


Figure 5-12: Propionic acid to acetic acid ratio for digesters 1 and 2 during the experimental period.

The P:A ratio does not appear to be a good indicator of stability in this case – showing lower stability in phases 4 and 5 for both digesters, which would be expected to be more stable than phases 1 and 2. Stability would be particularly expected in digester 2, which was fed at the same rate throughout the experiment, and should therefore have been acclimatised by phase 5. No instability was shown in other results (Figure 5-15, Figure 5-17). However, the same source also states that this ratio is more sensitive to disturbances than other indicators, and it is possible that the digesters were both becoming unstable towards the end of the experimental period. The raised levels of isobutyric acid and butyric acid in phase 5 (Figure 5-11) also show this. A longer experimental period with further disturbance testing would be required to confirm the usefulness of this indicator.

5.3.2 Digester acclimatisation shown by alkalinity during the experimental period

The acclimatisation was also shown by the intermediate and partial alkalinity (figure 5-13, figure 5-14). In this case, the measurements stabilised after around 60 days, which was longer than the VFA values took to stabilise (around 40 days). The partial alkalinity (PA) is a titration to pH 5.7 and measures the level of bicarbonate buffer in the digester. The intermediate alkalinity (IA) is a further titration to pH 4.3 and measures the level of VFAs in the digester. The alkalinity ratio (IA/PA, figure 5-15) combines these values to give an early indication of instability (Ripley, Boyle and Converse, 1986) measuring the ability of the digester to buffer against changes in pH. It is therefore logical that the alkalinity will reach a stable level after the VFA levels have decreased because the bicarbonate in the solution will first react with the excess acids, then start to build up a buffer. Values of over 0.5 indicate that a digester is 'unstable', particularly if the ratio has recently increased quickly from significantly lower values.

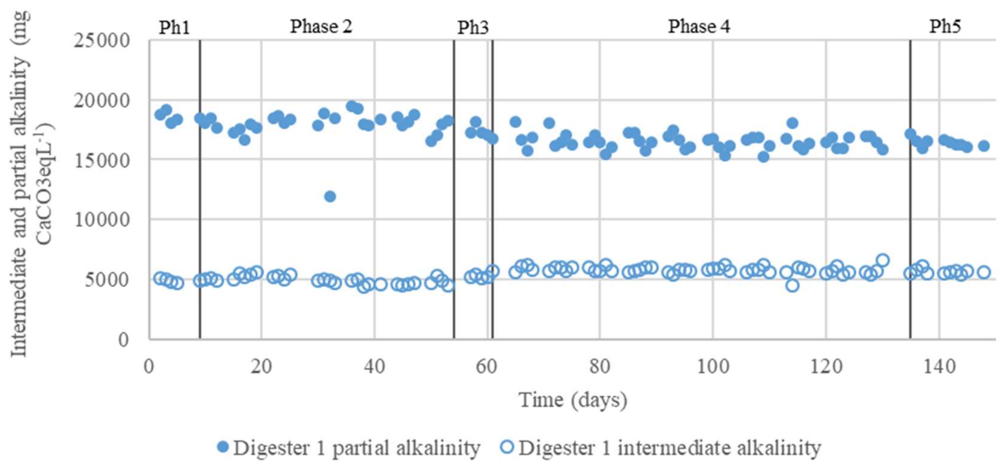


Figure 5-13: Digester 1 partial and intermediate alkalinity for during the experimental period.

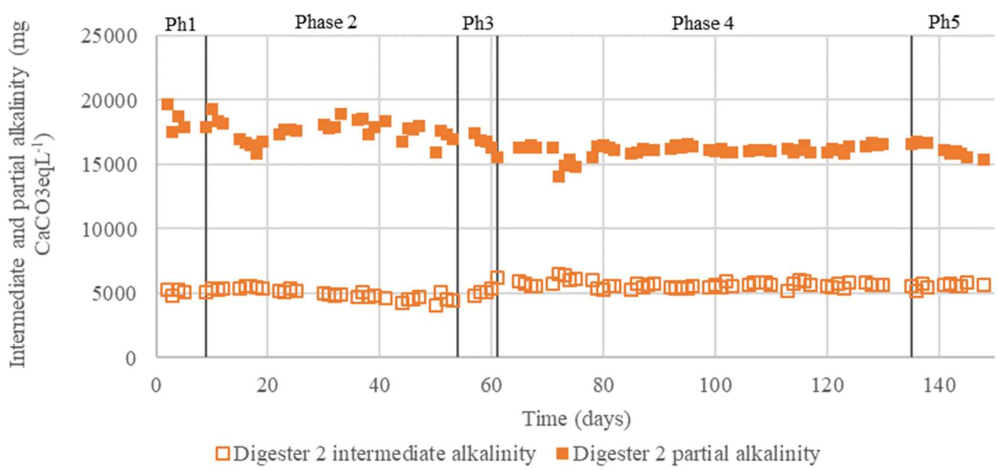


Figure 5-14: Digester 2 intermediate and partial alkalinity during the experimental period.

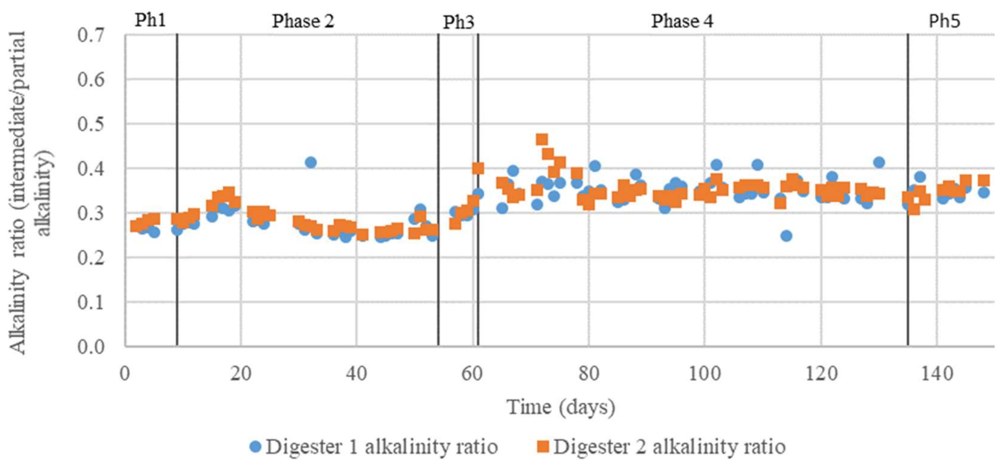


Figure 5-15: Digesters 1 and 2 alkalinity ratio during the experimental period.

In digester 2 (figure 5-16), the PA and IA both reach a constant level at around day 65 (phase 4). In digester 1 (figure 5-13), the IA stays at a roughly constant level from about day 65, whereas the PA varies with the fluctuations in feed rate. This fluctuation with feed rate indicates that the PA in this system is a better indication than IA of system instability, and that the bicarbonate values are varying as the loading rate goes up and down. The VFA levels, shown by the IA, are being kept relatively stable. As can be seen in phase 5 for digester 1 (day 134 onwards), the alkalinity ratio (figure 5-15) and the PA (figure 5-13) both stabilise quickly under constant feed conditions.

The alkalinity ratio has been shown to be a useful indicator of stability in other studies, more sensitive than pH or biogas methane composition (Sun *et al.*, 2019; Martín-González, Font and Vicent, 2013). As it is also a relatively quick and simple test, it is a useful element of digester monitoring and control.

5.3.3 Digester stability shown by biogas methane concentration

Another indicator of stability is the methane concentration of the biogas (figure 5-16, figure 5-17).

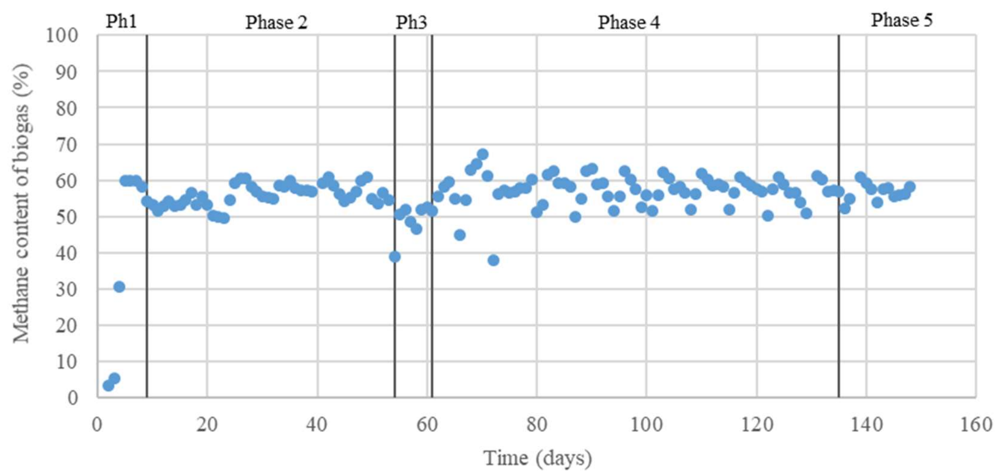


Figure 5-16: Digester 1 1-day average biogas methane concentration during the experimental period.

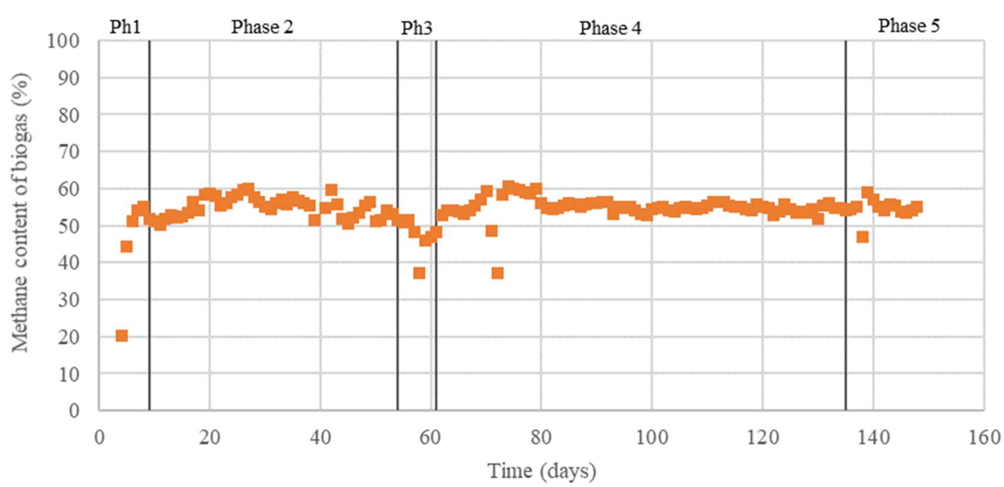


Figure 5-17: Digester 2 1-day average biogas methane concentration during the experimental period.

In digester 2, the biogas methane concentration indicators stabilised at around the start of phase 4 (day 64), with an increase caused by the overfeed that occurred overnight between days 70 and 71 (see section 5.2.5).

5.3.4 Digester 1 biogas quality during phase 4

During phase 4, digester 1 was subjected to a variable loading rate, which resulted in a fluctuating biogas methane concentration. The amount of fluctuation from the average during this phase (57.0%) is shown (figure 5-18), with the same period in digester 2 (average methane concentration 54.9 %) for comparison (figure 5-19).

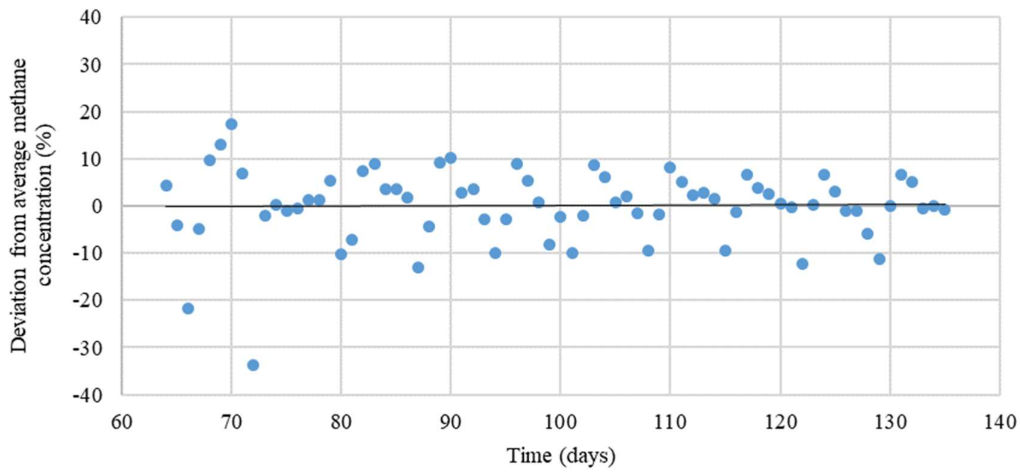


Figure 5-18: Digester 1 deviation of 1-day average from average methane concentration during phase 4.

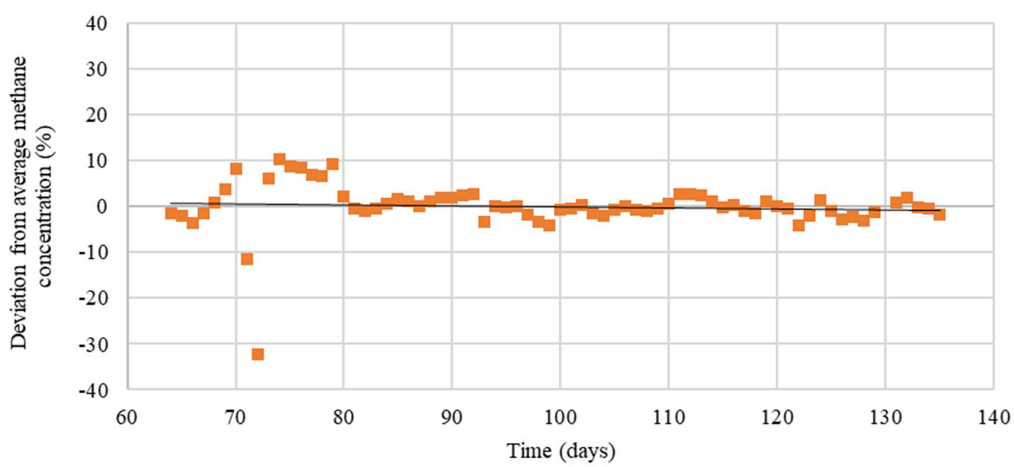


Figure 5-19: Digester 2 deviation of 1-day average from average methane concentration during phase 4.

Digester 1 exhibited a high degree of deviation from its average biogas methane concentration at the beginning of phase 4 (days 64-75), but this reduces and stabilises to a 'predictable' pattern at about day 103, 39 days after the start of the changing feed pattern. In the acetic and propionic acid levels for the digesters during phases 1 and 2 (figure 5-9, figure 5-10), the stabilisation took about 39 days. The similarity in these periods suggests that this may be a 'standard' stabilisation period for this digester when a change is introduced.

The methane concentration in digester 2 remains steady throughout phase 4, with a maximum variation of ±4.2% and standard deviation of 2.87 (figure 5-19). The overfeed on day 70-71 produces a large drop in gas quality, showing instability and imminent failure, followed by a period of high methane concentration when the feed is stopped (until day 72) and afterwards. The methane concentration returns to the average level 10 days after the disruption.

The methane concentration for digester 1 rose and dropped with the loading rate (OLR) in phase 4, with the rise and fall in methane concentration gradually decreasing in magnitude towards the end of phase 4 (figure 5-16). A plot of the biogas methane concentration against the loading rate for digester 1 across all phases (figure 5-20) shows that there is a roughly linear relationship between these two indicators, with the methane concentration decreasing as the loading rate increases. This is a mechanism that has been observed in other studies (López-Escobar *et al.*, 2014).

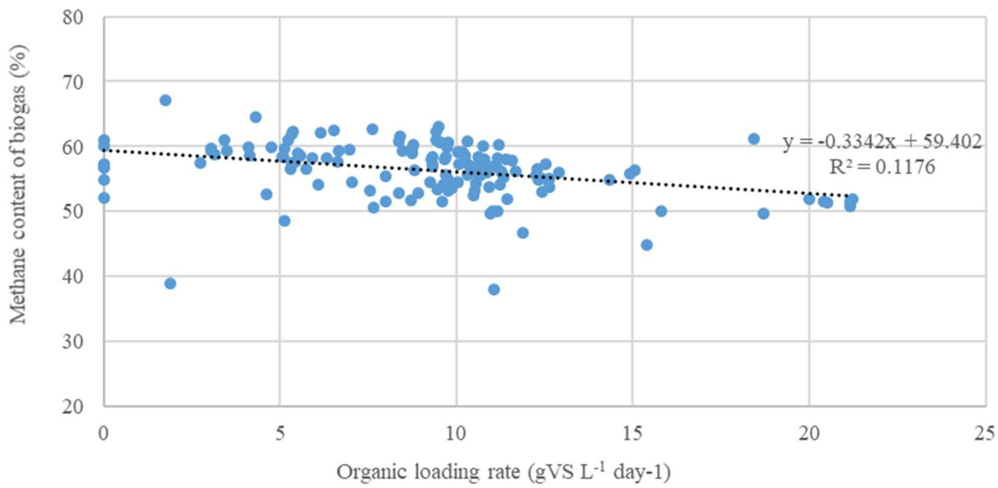


Figure 5-20: Digester 1 1-day average biogas methane concentration against organic loading rate during the whole experimental period.

The r-squared value of the linear regression is very low (0.0834), therefore the variance from the line of best fit is high, showing that there is only a loose connection between these two indicators. This is likely to be due to the complicated nature of the dynamics of an anaerobic digestion system – there are many different biochemical reactions taking place, and conversions from one substance to another can take place via several different pathways and may also be affected by feedback inhibition loops. As the digester remained stable throughout the experimental period, this graph does not show the effect of the digester under stress or failing conditions, which would show less linearity.

The methane concentration has been evaluated as a stability indicator in other studies and found to be relatively insensitive compared to the VFAs and alkalinity (Boe *et al.*, 2010; Sun *et al.*, 2019). In this experiment, it is a good indicator of a change in feed rate but not necessarily instability, as this would have been reflected in other indicators. It was also an indicator of acclimatisation to the variable feed rate, this being a short-term indicator rather than an indicator of a long-term instability.

5.3.5 Solids monitoring and mass balance

The total and volatile solids (TS and VS) and ash content for both digesters were monitored weekly throughout the experimental period. The TS and VS showed a steady rise in both digesters, with a slower rise in ash content (figure 5-21, figure 5-22).

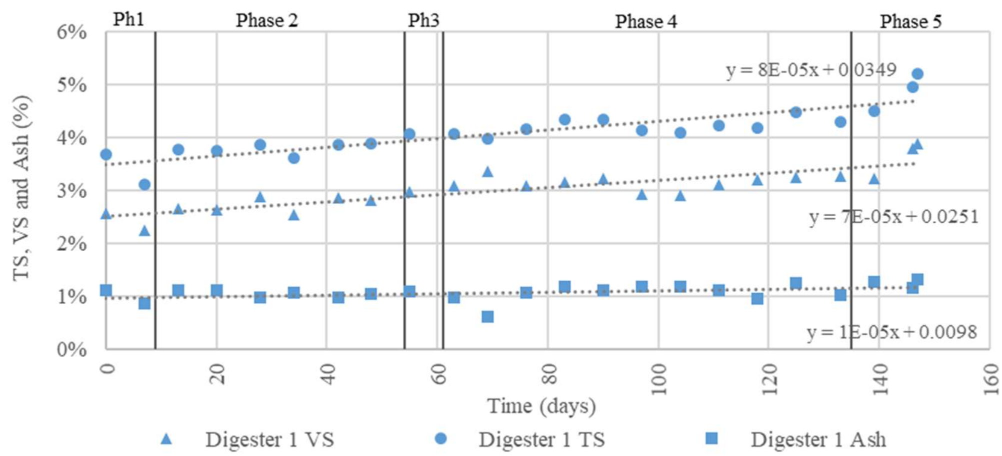


Figure 5-21: Digester 1 total and volatile solids and ash content throughout the experimental period.

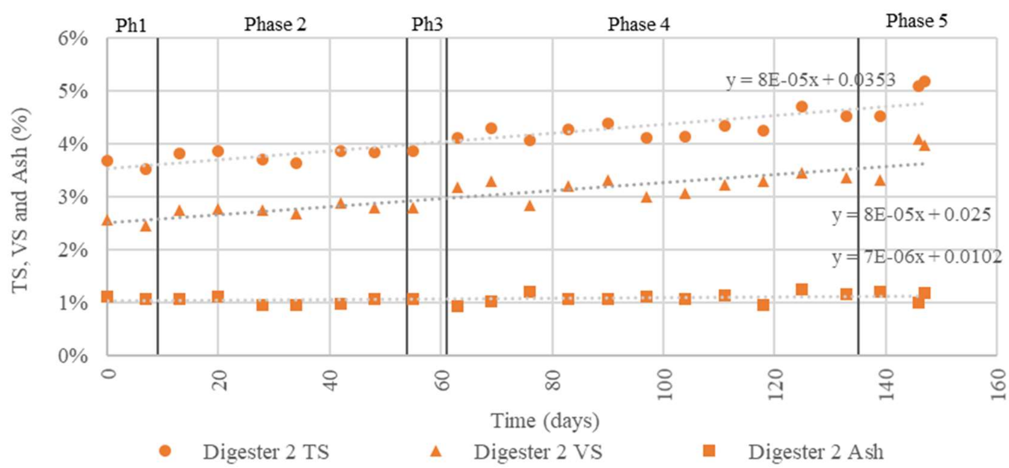


Figure 5-22: Digester 2 total and volatile solids and ash content throughout the experimental period.

At the beginning of the experimental period, the TS and VS of the inoculum (see section 5.2.4) were measured. When supplied with a different feed input, the TS and VS will gradually change to reflect the new conditions – that is, the composition and loading rate of the new feedstock. However, the TS and VS will also increase if there is a growth in microorganisms.

To determine what the expected TS and VS of the digesters would be after they had adjusted to the feed and stabilised, a mass balance was constructed (Appendix A). The TS and VS used were determined experimentally, as were the average OLR and methane % of biogas. The

BMP value used was the average of the BMP measurements taken at the start and end of the experimental period (see section 5.3.7).

Table 5-7: Inputs and predicted outputs for the mass balance of the laboratory digesters.

Item	**Unit**	**Digester 1**	**Digester 2**
Inputs			
Organic loading rate (average over experimental period)	gVS L^{-1} day^{-1}	2.402	2.426
BMP	$mLCH_4$ gVS^{-1}	374.1	366.2
Methane content of biogas (average over experimental period)	%	56.0%	54.0%
Predicted outputs			
TS	%	8.9	8.2
VS	%	7.3	6.4
Ash content	%	2.3	2.3

The mass balance showed that when the systems had stabilised, the predicted TS and VS for digester 1 would be 8.9% and 7.3% respectively, and for digester 2 would be 8.2% and 6.4% respectively. The predicted ash content for both digesters was 2.3% (table 5-7).

	Total solids (TS)	**Volatile solids (VS)**	**Ash content**
Digester 1			
Predicted	8.9	7.3	2.3
Final measured value	5.2	3.9	1.3
Difference	58%	53%	57%
Digester 2			
Predicted	8.2	6.4	2.3
Final measured value	5.2	4.0	1.2
Difference	63%	63%	52%

The difference in the predicted TS and VS was due to the different inputs. At the end of the experimental period, the TS in digester 1 had reached 5.2% and the VS had reached 3.9%, with both still increasing (figure 5-21). The TS and VS for digester 1 were both below that predicted in the mass balance, which indicates that they were still adjusting to the feed input. This was supported by the fact that the TS and VS in the graph had not started to level off. The same is the case for digester 2 (figure 5-22). Comparing the predicted versus final measured TS, VS and ash content in both digesters, the measured TS and VS in digester 2 were closer to the predicted value than that of digester 1. This suggests that relatively speaking, the VS was increasing in digester 2 more quickly than in digester 1. There could be two explanations for this; digester 2 had greater microbial growth or had a build-up of undigested VS.

The difference in VS was investigated further in the feed response analysis, BMP tests and the analysis of predicted versus actual methane production (sections 5.3.6, 5.3.7 and 5.3.10).

5.3.6 Feed response

The biogas production immediately after a feed event during the different phases was investigated (figure 5-23 to figure 5-30). The days studied were days 4 and 11 (phase 1), days 20 and 41 (phase 2), days 83 and 125 (phase 4) and day 139 (phase 5). These days in particular were selected as they seemed typical of that period and were roughly at the start or end of the phases. In each of the figures, a feeding event is shown by vertical black lines – the feed events were every six hours (i.e. four times a day).

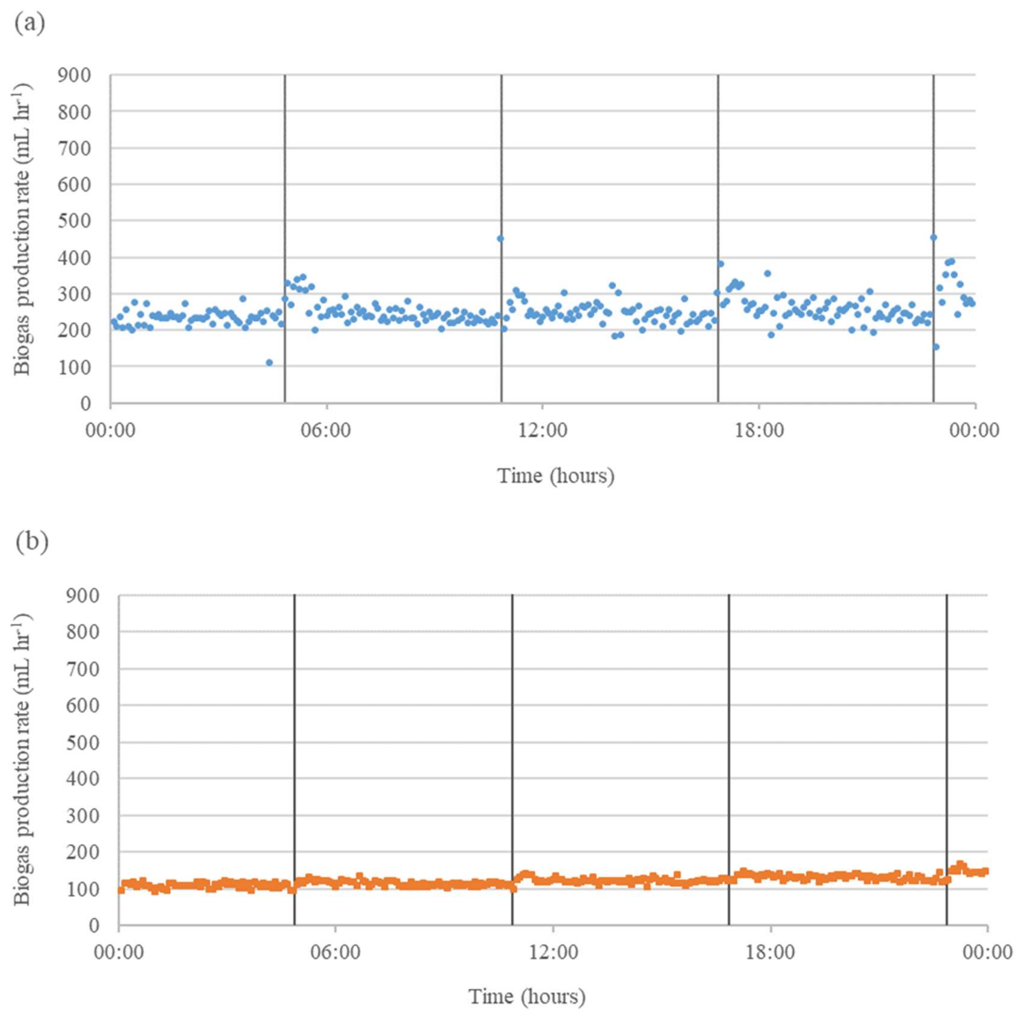


Figure 5-23: Biogas production on day 4 (phase 1) for (a) digester 1 and (b) digester 2. Feeding events are indicated by vertical black lines.

At the start of phase 1, when the feed rate was 1.4 gVS L^{-1} day^{-1}, the biogas production rate is steady in both digesters with almost no biogas production 'spike' (that is, a sharp peak) at feed events (figure 5-23). Digester 1 is producing approximately twice as much biogas as digester 2, and its biogas production rate is more erratic.

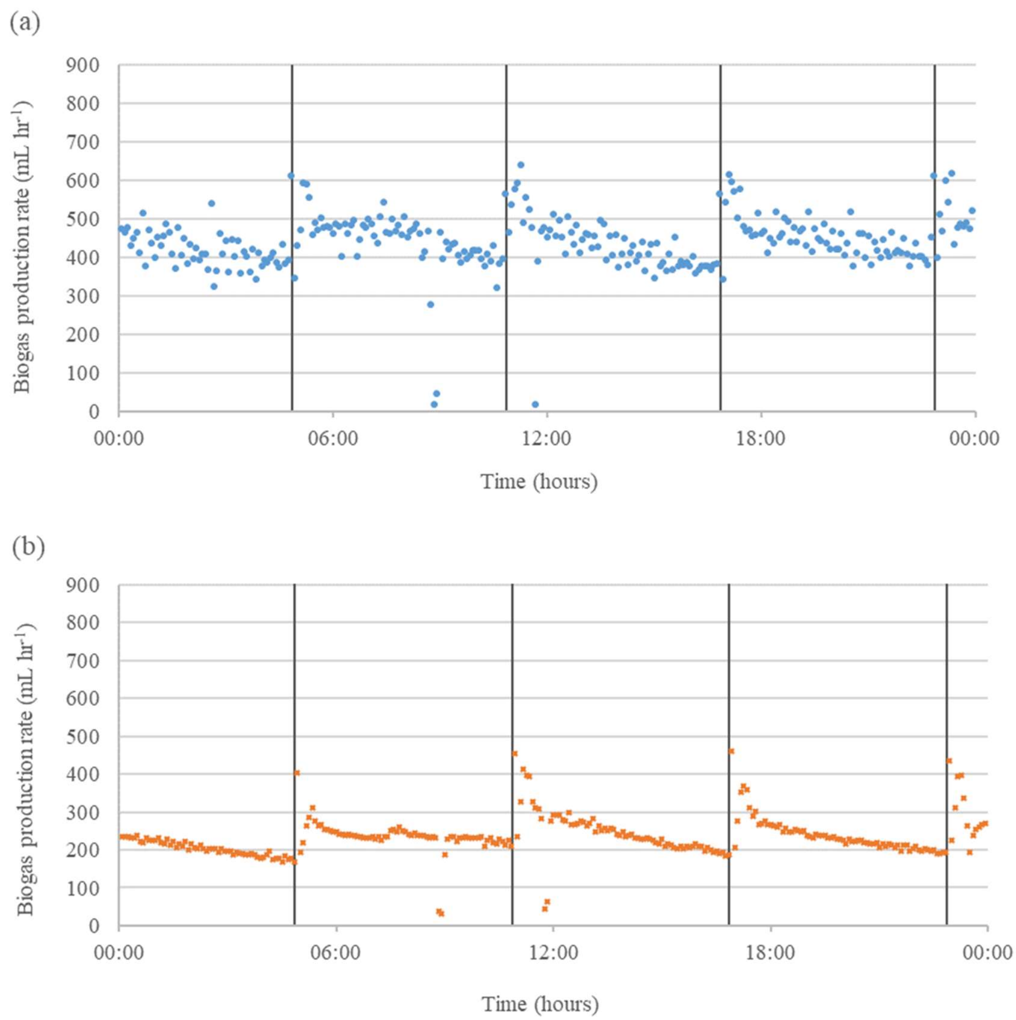


Figure 5-24: Biogas production on day 11 (Phase 1) for (a) digester 1 and (b) digester 2. Feeding events are indicated by vertical black lines.

At 11 days (at the end of phase 1, the ramp-up period), the digesters were being fed at a rate of 2.3 gVS L^{-1} day^{-1} (figure 5-24). The biogas production in both digesters showed a pattern of increase immediately after feeding (within 5 minutes) followed by a gradual decline until the next feed event, with the average biogas production steadily increasing. The slowly increasing feed rate over a day shows that there is a delay in response to the increased loading rate.

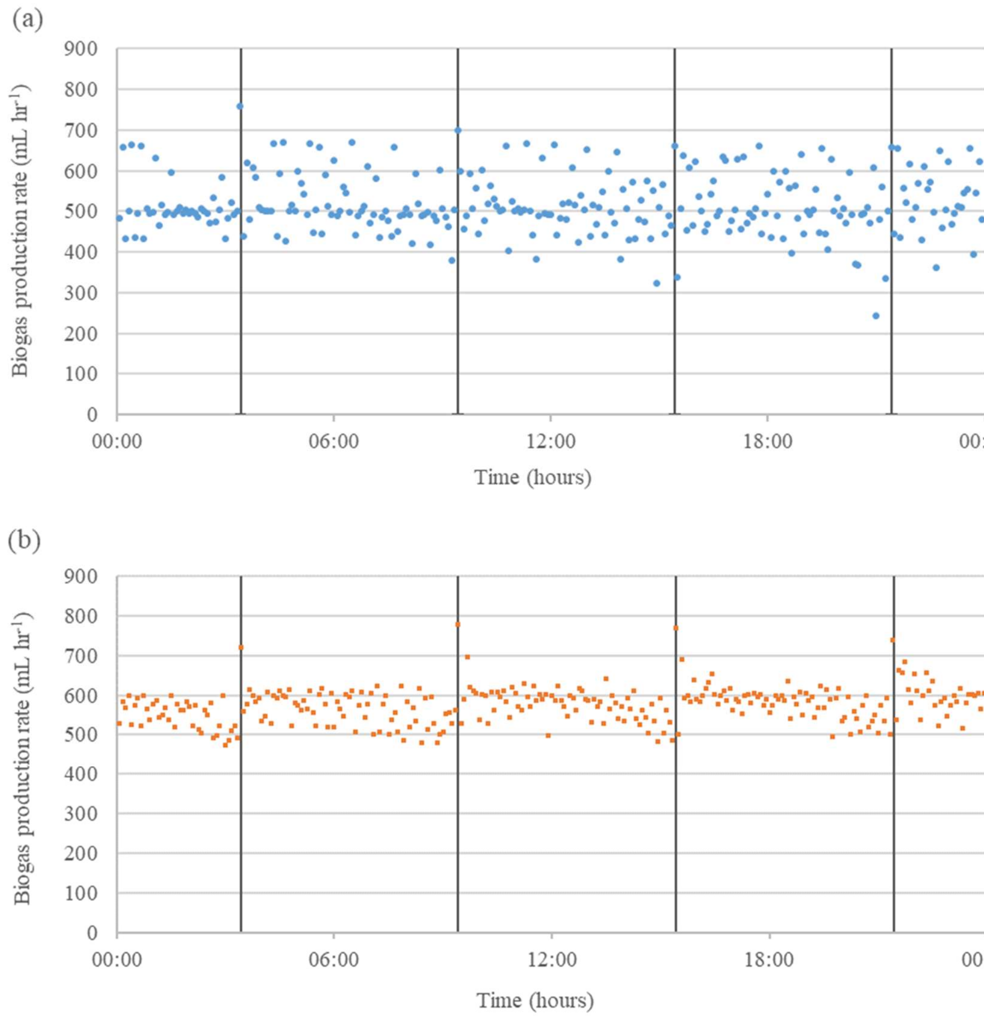


Figure 5-25: Biogas production on day 20 (phase 2) for (a) digester 1 and (b) digester 2. Feeding events are indicated by vertical black lines.

On day 18 (at the start of phase 2) the feed rate was 2.3 gVS L^{-1} day^{-1} (figure 5-25). The gas production was slightly greater and more stable in digester 2. The biogas production in both digesters was variable with no discernible pattern.

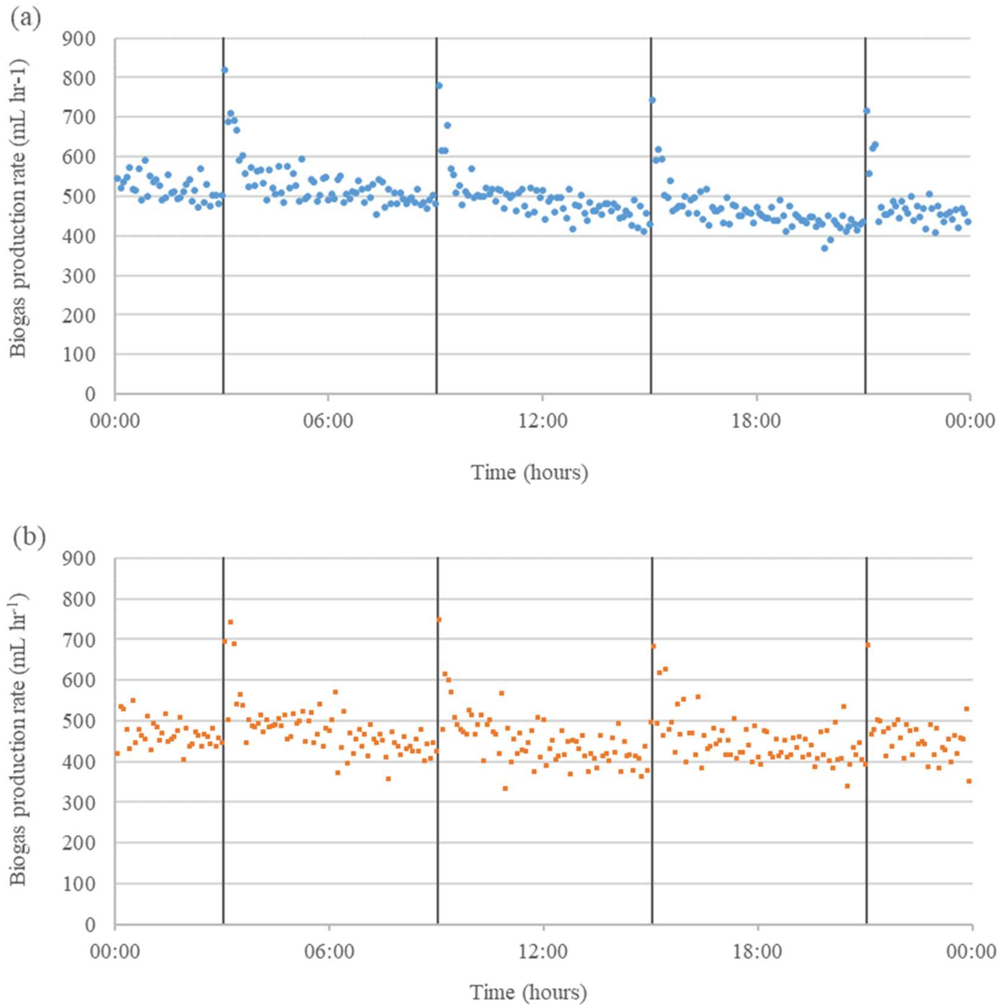
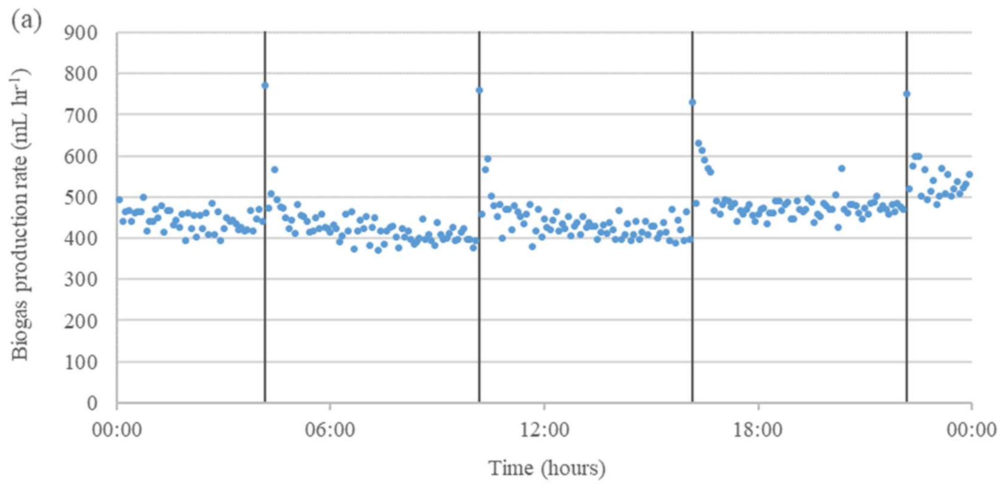


Figure 5-26: Biogas production on day 41 (phase 2) for (a) digester 1 and (b) digester 2. Feeding events are indicated by vertical black lines.

On day 41, towards the end of phase 2 (figure 5-26), at a feed rate of 2.3 gVS L^{-1} day^{-1}, both digesters showed a spike of biogas production of roughly equal magnitude at each feeding event, followed by a decreasing biogas production rate which stabilised at about 1.5 hours after feeding, indicating a degree of acclimatisation to the loading rate when compared to day 18 which showed no pattern of 'settling'.



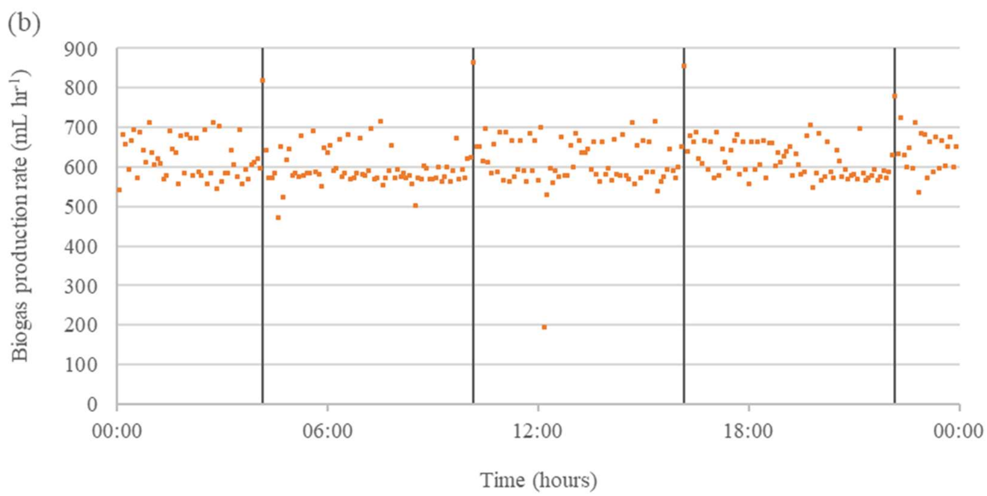


Figure 5-27: Biogas production on day 83 (phase 4) for (a) digester 1 and (b) digester 2. Feeding events are indicated by vertical black lines.

On day 83 (phase 4, figure 5-27), the feed rates were 3.5 and 2.6 gVS L^{-1} day^{-1} for digesters 1 and 2 respectively. The spike of biogas production at each feeding event was larger in digester 1 than digester 2. The biogas production was lower in digester 1 because it had been fed at 0.5 gVS L^{-1} day^{-1} on day 82. The digesters showed responses to the addition of feedstock on two different timescales: a short-term response (within minutes) to individual feed events, and a longer term response (over a number of days) to the amount of feed input over the previous period.

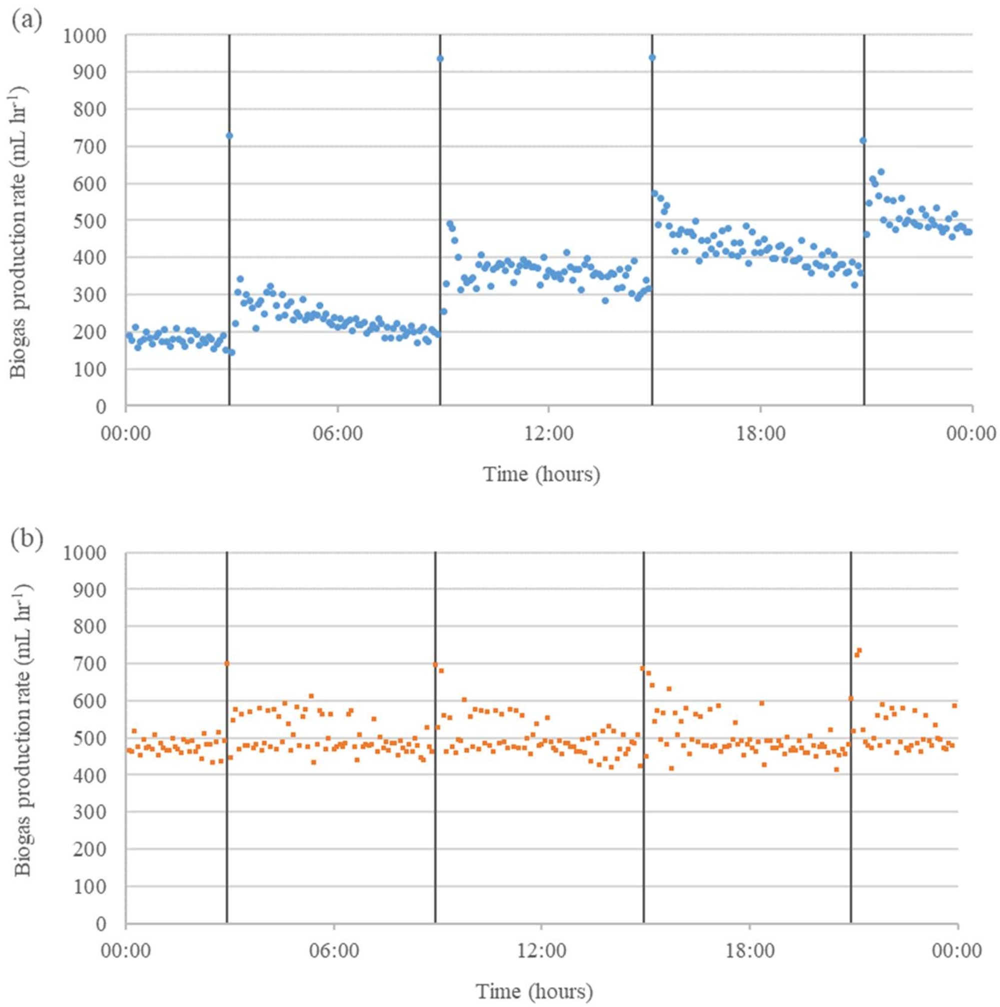


Figure 5-28: Biogas production on day 125 (phase 4) for (a) digester 1 and (b) digester 2. Feeding events are indicated by vertical black lines.

The increased response to feed events by digester 1 continued to become more pronounced throughout phase 4 (figure 5-28). On day 125 (phase 4, figure 5-28), the digesters were being fed at 3.5 and 2.6 gVS L^{-1} day^{-1} for digesters 1 and 2 respectively. The previous day, the feed rates had been 0.5 and 2.8 gVS L^{-1} day^{-1} for digesters 1 and 2 respectively. Digester 1 again showed a much larger spike in biogas production after a feed event than digester 2.

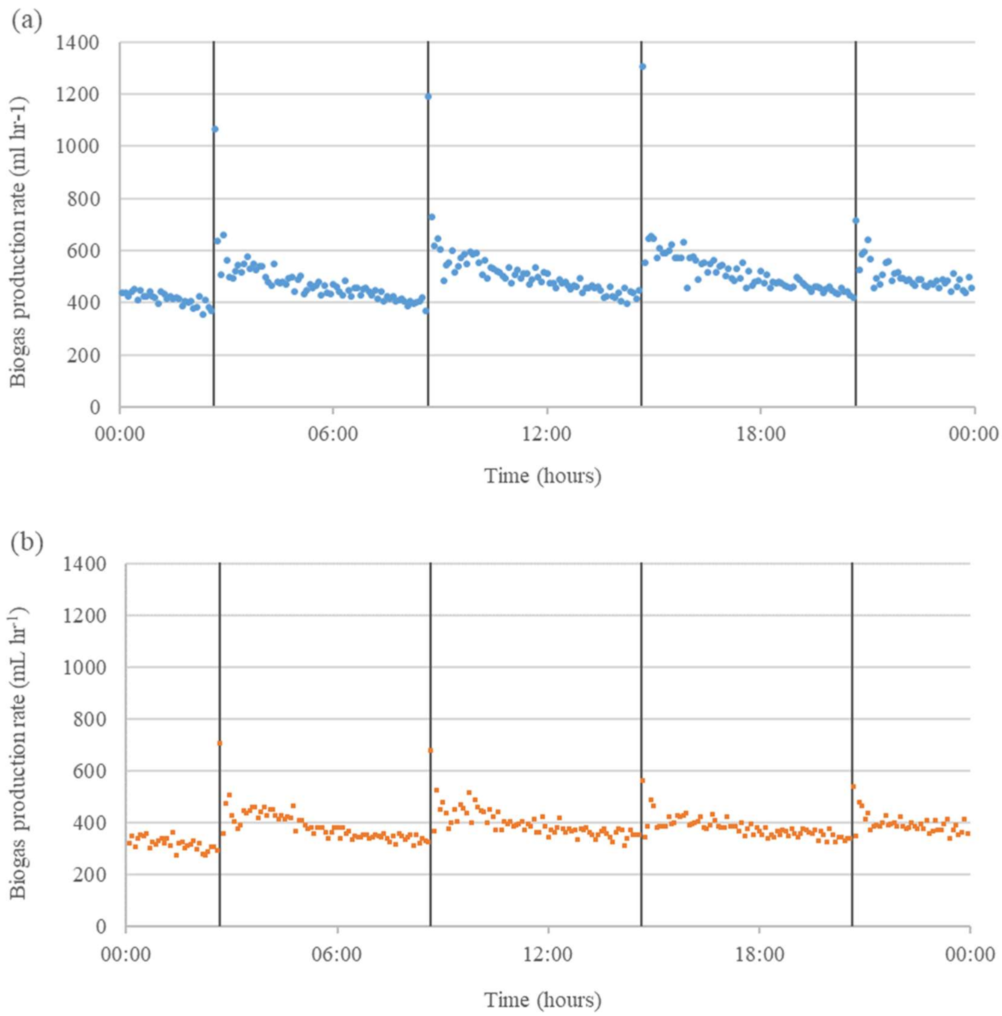


Figure 5-29: Biogas production on day 139 (phase 5) for (a) digester 1 and (b) digester 2. Feeding events are indicated by vertical black lines.

At the start of phase 5 (day 139, figure 5-29), both digesters were subject to the same constant loading rate (2.6 gVS L^{-1} day^{-1}). Both showed a very similar biogas production pattern except that digester 1 continued to show a much larger spike after a feeding event than digester 2. Digester 1 was also producing more biogas than digester 2 (493.1 ml hr^{-1} compared to 382.4 ml hr^{-1} one-day average).

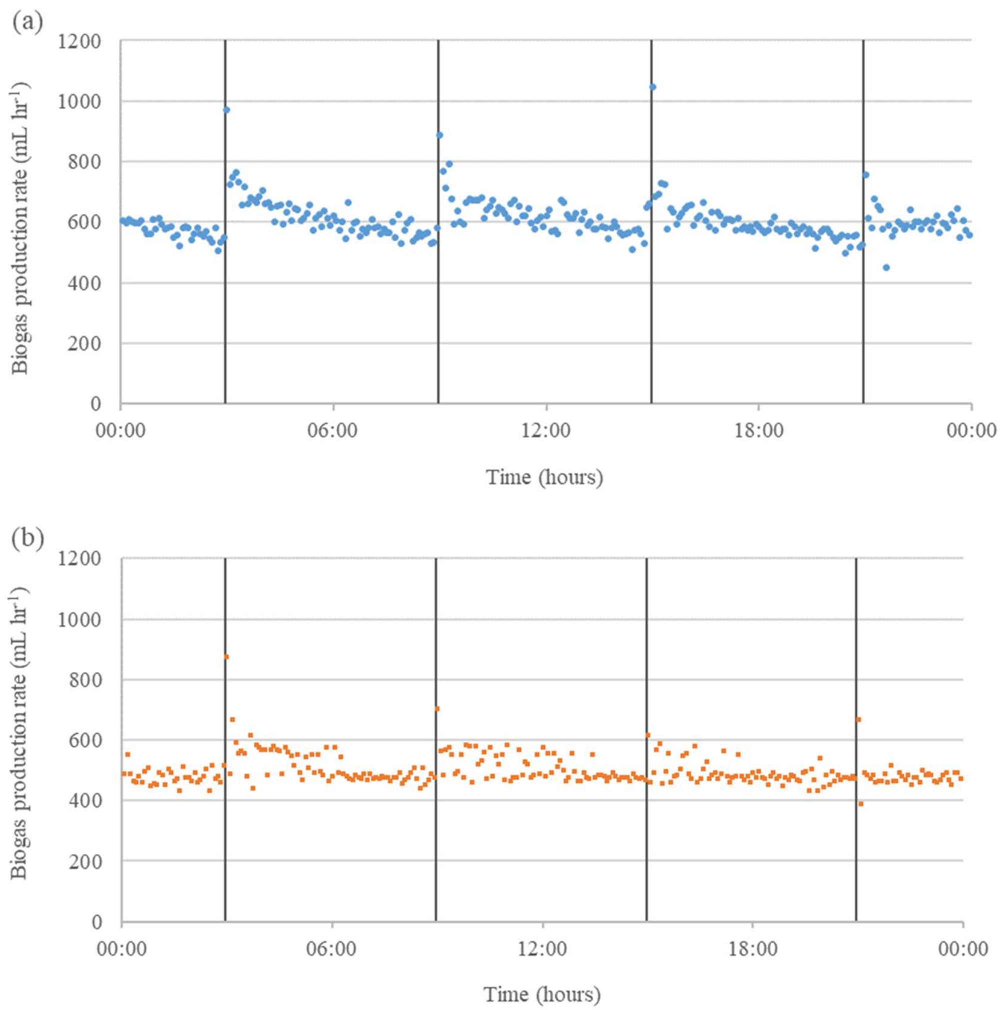


Figure 5-30: Biogas production on day 146 (phase 5) for (a) digester 1 and (b) digester 2. Feeding events are indicated by vertical black lines.

After a week in which the digesters were both fed at an OLR of 2.6 gVS L^{-1} day^{-1} (phase 5, figure 5-30), digester 2 (b) continued to produce gas in a steady pattern, with a moderate increase in biogas production rate at each feeding event. There was a higher spike after feeding events for digester 1, and a higher rate of biogas production (601.6 mL hr^{-1} compared to 498.3 mL hr^{-1}).

These biogas production patterns show that both digesters responded in two ways to the changing feed: a spike after each feed event, and a more gradual increase or decrease in response to the loading rate. This can be explained by the composition of the feedstock, which is made of a mixture of complex molecules (for example, proteins and carbohydrates) and smaller molecules (for example, volatile fatty acids). The spikes after feeding in biogas production rate are caused by the digestion of the smaller molecules, whereas the larger molecules take longer to break down and so will have a longer-term effect on the biogas production rate.

During phase 4, digester 1 was subjected to a variable loading rate and consistently had a larger spike of biogas production rate after feed events compared to digester 2, with the effect becoming more pronounced the longer the variable loading pattern was imposed. This could be explained by an increase in microbial population. At periods of high loading, the microbial population in digester 1 will have increased in size in response to the extra feed, resulting in a larger microbial population. This would explain why digester 1 produced a larger response to feed events; there were more microorganisms available to convert the VFAs (figure 5-29).

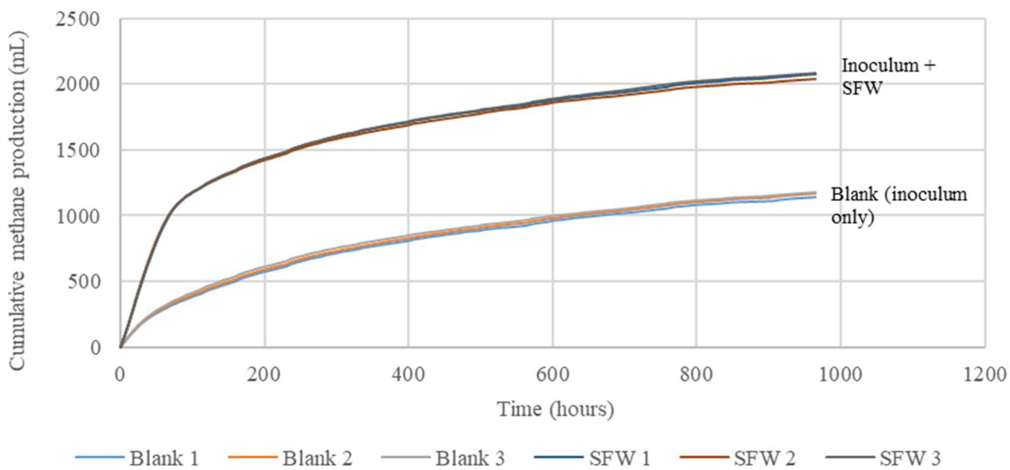
The speed of response to feed events, shown by the time of the peak of biogas production after a feed event, was the same in both digesters throughout the experiment, so the variable loading rate in digester 1 and acclimatisation in both digesters had no effect in this respect. This suggests that the microorganisms breaking down the small molecules do not become more or less efficient in their operation as a result of these effects, and the increased size of 'spike' in digester 1 is purely due to an increased microbial population.

Other studies that reported on response to changes in feed have focused on the increase and decrease in biogas production over the time span of hours rather than minutes after changes in feed (Laperrière *et al.*, 2017; Mauky *et al.*, 2015; Lemmer and Krümpel, 2017), and so cannot be used as a comparison to these results. Studies on variable feeding regimes have reported on the stability of the digesters (Lemmer and Krümpel, 2017; Mauky *et al.*, 2017) but not other effects such as the methane concentration or immediate feed response. The effect on the microbial population has been reported upon (De Vrieze, Verstraete and Boon, 2013; Bonk *et al.*, 2018; Mulat *et al.*, 2016b), finding that the methanogen genus *Methanosarcina* became dominant in digesters that were intermittently fed, which has a higher substrate uptake rate than other methanogens (Bonk *et al.*, 2018) and could explain the greater feed response in digester 1. Further experimentation could include analysis of the microbial population to explore this.

5.3.7 Biological methane potential

Biological methane potential (BMP) tests were performed twice – once using the digestate from a local food waste digestion plant as an inoculum, and once using the digestate from digesters 1 and 2 at the end of the experimental period as an inoculum. The purpose of the test was to determine whether the variable feed pattern had brought about any change in the BMP.

The BMP at the start of the experimental period was performed using cellulose and synthetic food waste (SFW) as a substrate, with blank tests (inoculum only) as a control (figure 5-31).



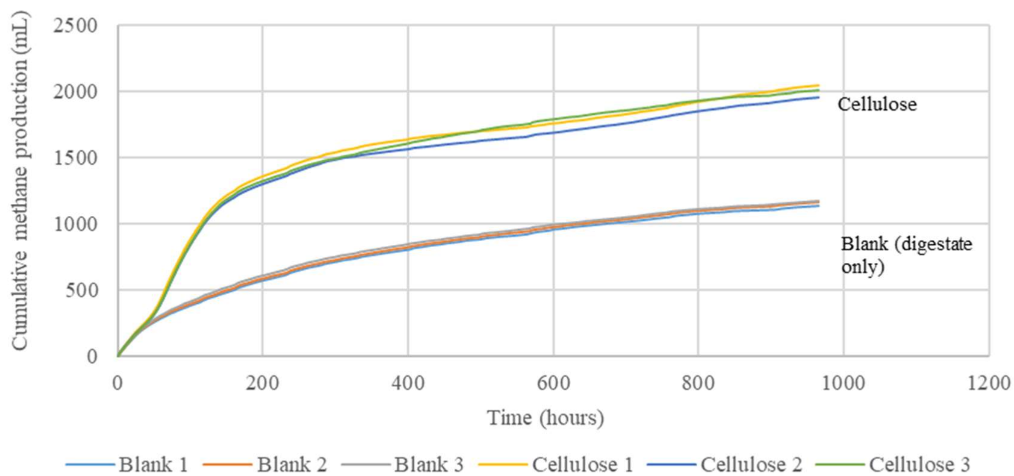


Figure 5-31: Cumulative measured methane production over time for biological methane potential tests of synthetic food waste and cellulose, conducted at the start of the experimental period.

The BMP test was run for 965 hours (40.2 days). All three samples of SFW gave similar curves for the methane production over time– a linear period initially, flattening out after about 90 hours. The curves for cellulose were an 'S' shape– a slow rate at the beginning, followed by a linear period, then flattening out after about 120 hours. The uncertainty for cellulose (±6.3% of the BMP) is greater than the uncertainty for the SFW (±1.7%). All samples were tested under the same conditions (using the same inoculum, at the same temperature, with the same ratio of inoculum VS to sample VS), so this difference in error between substrates is likely to be due simply to variations in microbial activity for individual test jars.

The average methane production in the inoculum was subtracted from the average methane production for each set of samples to give BMP measurements with error values (figure 5-32).

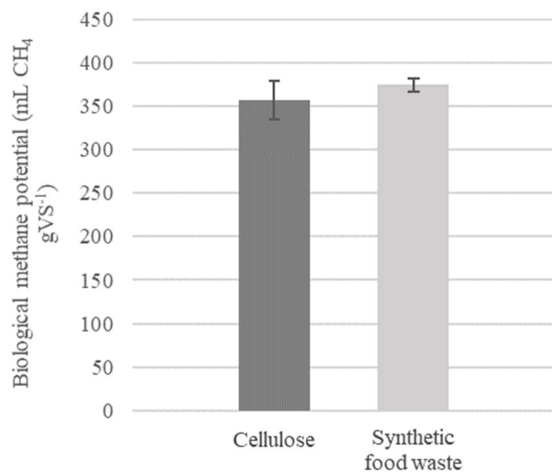


Figure 5-32: BMP for cellulose and synthetic food waste using the starting inoculum.

The known BMP for cellulose is 340-370 mL CH_4 gVS $^{-1}$ (Filer, Ding and Chang, 2019; Wang *et al.*, 2014) so the value obtained (357±22 mL CH_4 gVS $^{-1}$) is within the accepted range and confirms that the test equipment was working correctly. There is a range of possible BMP values for cellulose because the measured value depends on the conditions and the resulting amount of degradation that is achieved (Wang *et al.*, 2014). The BMP value of the synthetic food waste was 374± 7 mL CH_4 gVS^{-1}. The BMP for food waste has been reported between 160 and 530 mL CH_4 gVS^{-1} (Xu *et al.*, 2018) so this value is also within the expected range.

A second BMP test was performed at the end of the experimental period (figure 5-33 to figure 5-36), to compare the digestates from digesters 1 and 2 and see what effect the variable feed rate had had, if any. The substrates used were SFW and cellulose.

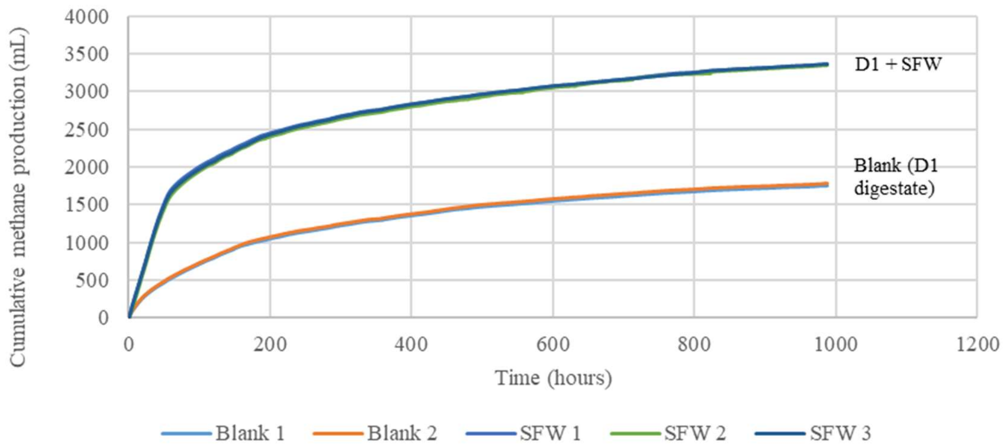


Figure 5-33: Cumulative measured methane production for SFW samples using digestate 1, in a BMP test conducted at the end of the experimental period.

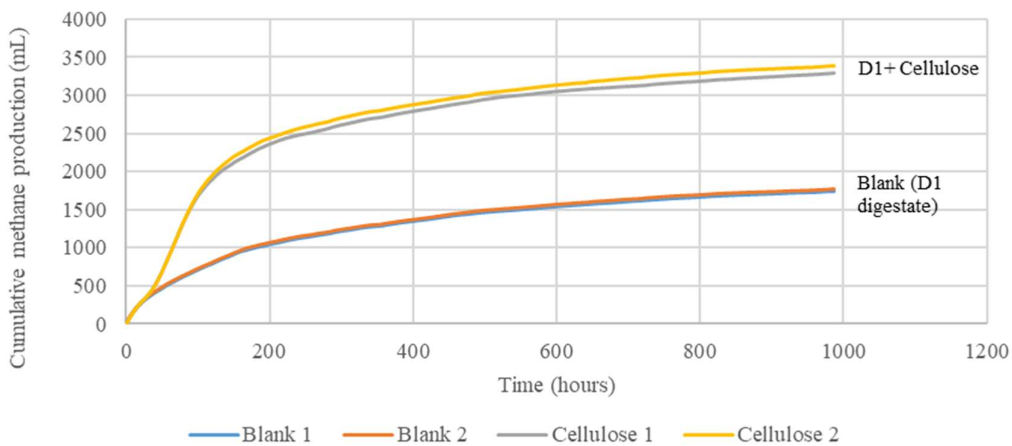


Figure 5-34: Cumulative measured methane production for cellulose samples using digestate 1, in a BMP test conducted at the end of the experimental period.

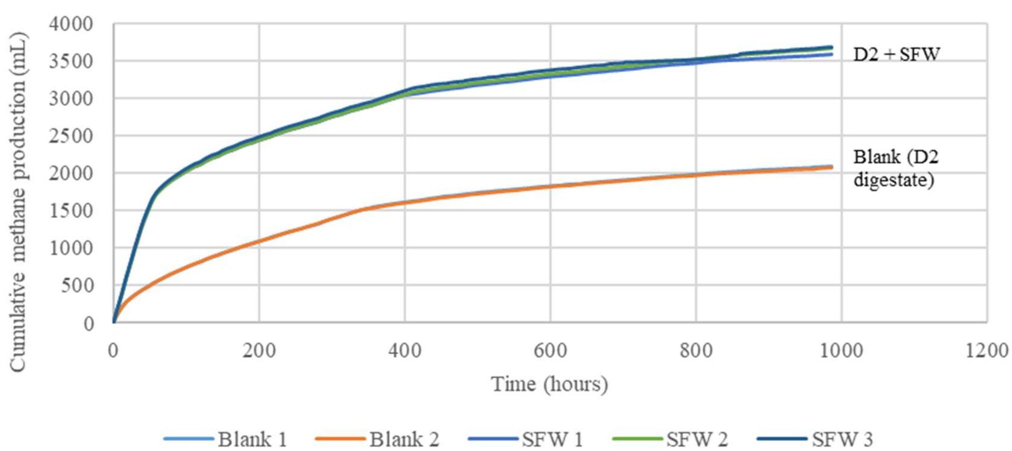


Figure 5-35: Cumulative measured methane production for SFW samples using digestate 2, in a BMP test conducted at the end of the experimental period.

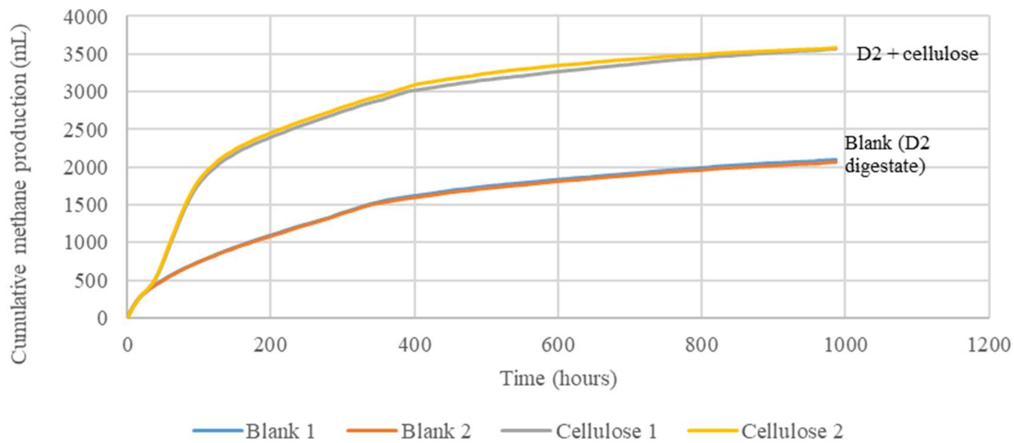


Figure 5-36: Cumulative measured methane production for samples using digestate 2, in a BMP test conducted at the end of the experimental period.

In the second BMP test, the graphs produced by the different substrates were similarly shaped to the graphs in the previous BMP test which supports the validity of both tests.

Figure 5-37 shows a comparison between the BMP results for the first BMP test (that is, with the inoculum) and the second BMP test (with digestates from digesters 1 and 2) with cellulose and SFW. The tests showed a higher BMP using the digestate from digester 1 ('digestate 1'), compared to the digestate from digester 2 ('digestate 2'), for both cellulose and SFW (7.2% and 6% higher respectively). A comparison of the first and second BMP tests shows the effect of the variable feeding rate on methane production.

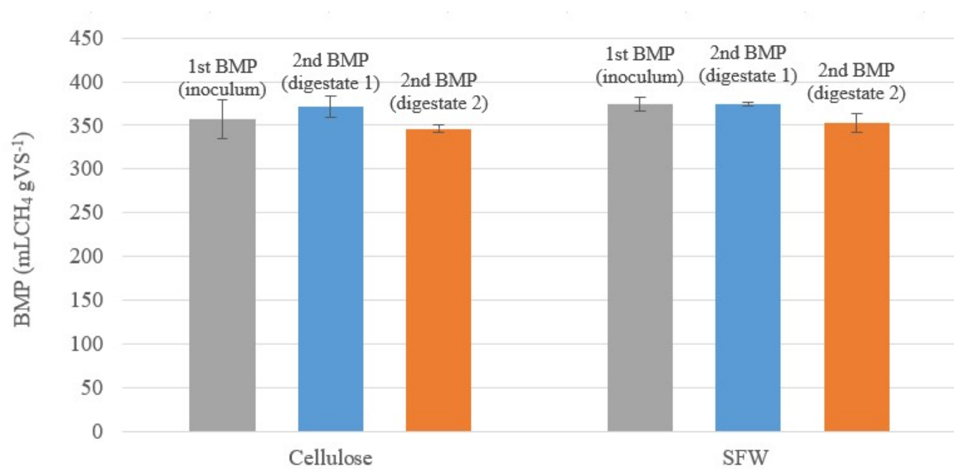


Figure 5-37: BMP tests using cellulose and synthetic food waste as a substrate, with digestate 1 and digestate 2 as inoculums.

Comparing the BMP of cellulose between the original inoculum and the post-experimental digestates, digestate 1 produced a higher BMP than the inoculum in the first BMP test, whereas the BMP with digestate 2 was lower in the second test compared to the first. With SFW as a substrate, digestate 1 produced roughly the same BMP as the inoculum, whereas digestate 2 produced a lower BMP than the inoculum.

The higher BMP in digester 1 suggests that the varied feeding regime has created more favourable conditions for digestion of both the cellulose and the SFW, compared to digester 2.

Previous studies have compared the use of acclimatised and unacclimatised inocula in a BMP test and have found differing results. Some studies found that acclimatisation led to a better breakdown for the test sample (Steinmetz *et al.*, 2016; De Vrieze *et al.*, 2015). However, an earlier study (Elbeshbishy, Nakhla and Hafez, 2012) found that using an acclimatised inoculum with food waste resulted in a lower biological methane potential measurement. The effect on the inoculum brought about by the different feed regimes in this experimental work is not possible to state decisively as there were differences in the performance of the digesters from the start of the experiment (see section 5.3.9). Further experimentation, with a longer experimental period and larger variations in feed rates, might provide clearer and more informative results.

5.3.8 Solids accumulation shown by the BMP test

The specific methane production over time (the methane production taking into account the initial VS in the test jar) for the blank digestate for each digester was compared (figure 5-38).

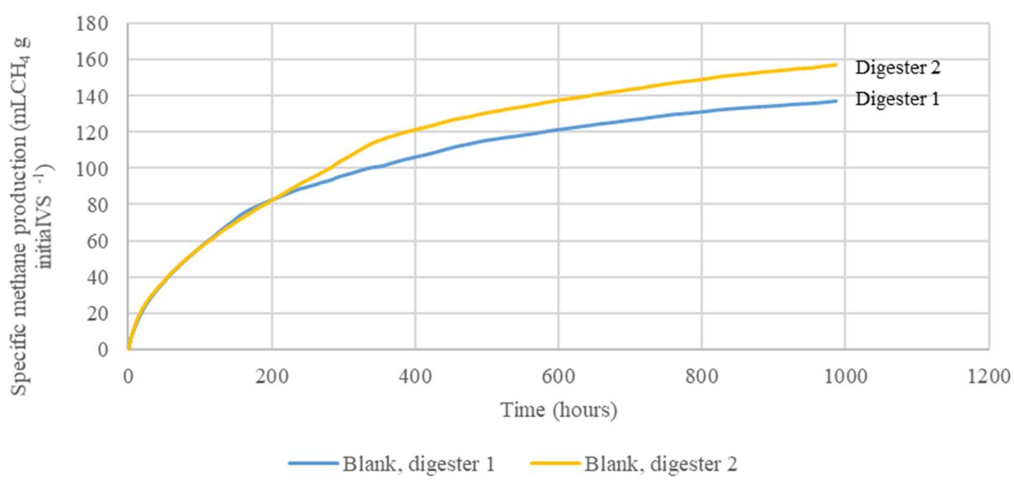


Figure 5-38: Specific methane production over time for the blank digestates during the BMP test conducted at the end of the experimental period.

The specific methane production was 14% higher in digestate 2 than in digestate 1, with no feed added (i.e., a blank sample). The VS in the digestate is composed of a mixture of microorganisms and undigested feed. To determine which of these was causing the difference in methane production, the kinetics were studied.

If the extra gas was due to a larger or more effective population of microorganisms in digester 2, the initial methane production rate would be higher in digestate 2 than digestate 1, resulting in a steeper initial gradient for digestate 2. The shapes of the graphs are very similar at the beginning, indicating that the size of the microorganism population was similar, so it is likely that the difference in methane production is due to undigested feed in digestate 2. The methane production rate starts to slow at around 160 hours in digestate 1, showing that the feed source for the microorganisms is becoming scarce. However, in digestate 2, the same rate of methane production continues for longer, suggesting that there was more undigested feed in digestate 2 at the start of the BMP test. This implies that during the experiment, digester 1 was working more efficiently that digester 2, as there was less undigested feed in the digestate. Other studies have not reported on the volatile solids destruction rate in digesters under variable feed, and so this area would benefit from further research.

5.3.9 Digester 1 and 2 performance in each phase

The performance in both digesters in terms of the specific methane production (mLCH_4 gVS^{-1} added), was calculated from the methane production and feed added (figure 5-39). The specific methane production was calculated by dividing the total methane production during the phase by the total feed added (in gVS) during the phase.

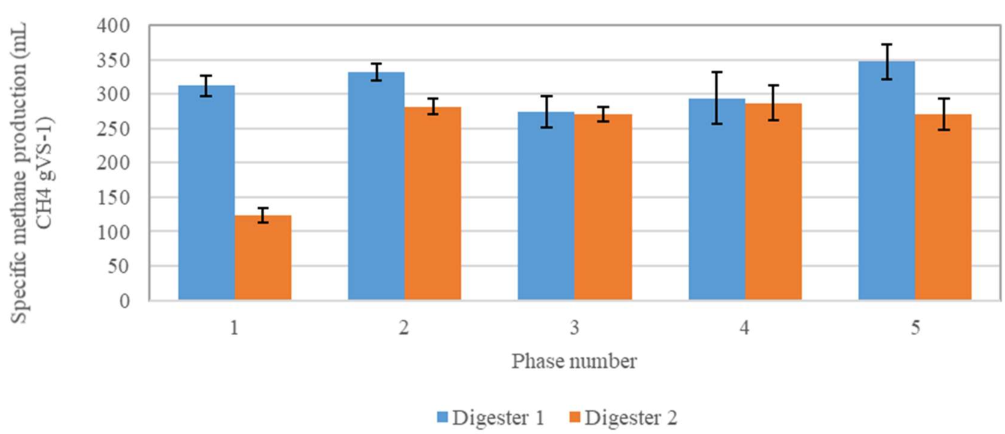


Figure 5-39: Specific methane production for digesters 1 and 2 in each phase of the experimental period.

Digester 2 has a much lower specific methane production than digester 1 in phase 1. This was also reflected in the low methane production (figure 5-42) and low biogas quality (figure 5-44) in digester 2 during this phase. The digesters were under the same conditions at this point, and had not been running for a long period, so this difference could be explained by differences within the digester microbial populations. The heterogeneity of anaerobic digester reactors has been observed in other studies and was shown to produce unpredictable and unaccountable differences in performance in digesters under the same conditions (Lv *et al.*, 2014b). This difference in performance continues into phase 2, where the specific methane production for digester 2 was lower than that of digester 1, despite being under the same conditions. However, by the end of phase 2, digesters 1 and 2 were producing the same amount of biogas (figure 5-26) with the same methane content (figure 5-16, figure 5-17).

The specific methane production was roughly the same for both digesters in phases 3 and 4, but in phase 5, the specific methane production in digester 1 increased by 18% compared to phase 4, whereas in digester 2 the specific methane production dropped by 6%.

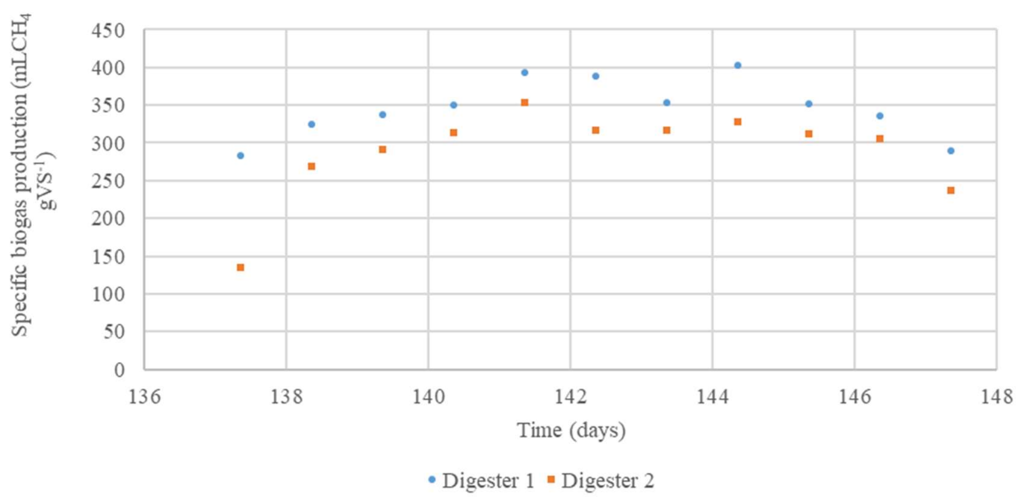


Figure 5-40: 1-day average specific methane potential for digesters 1 and 2 during phase 5 (days 136 to 147).

The 1-day average specific methane production of digester 1 in phase 5 was consistently higher than that of digester 2 (figure 5-40) and over the whole of phase 5 was 28% greater (347 mLCH4 gVS^{-1} compared to 271 mLCH4 gVS^{-1}). This suggests that the digester was producing more methane for each gram of VS as a result of the varied feed pattern. This was supported by the outcome of the BMP test (section 5.3.7).

5.3.10 Predicted versus actual methane production

In the mass balance (Appendix A), the predicted methane production for each digester was calculated, both over the entire experimental period and over each phase, based on the amount of VS fed to the digester, the average methane concentration, the biogas production and the biological methane potential (BMP) for the feedstock which was tested at the start and end of the experimental period. To obtain the predicted methane production for each phase, the actual feed loading rate (in gVS per day) for that phase was put into the mass balance, so that both the predicted and actual methane were derived from the same feed amount, ensuring a fair comparison. The predicted and actual methane production for each phase were calculated and compared for both digesters (figure 5-41, figure 5-42). A VS destruction rate of 85% was initially used for all scenarios as this was a mid-range value from published literature for food waste (Banks, 2009; Paritosh *et al.*, 2017).

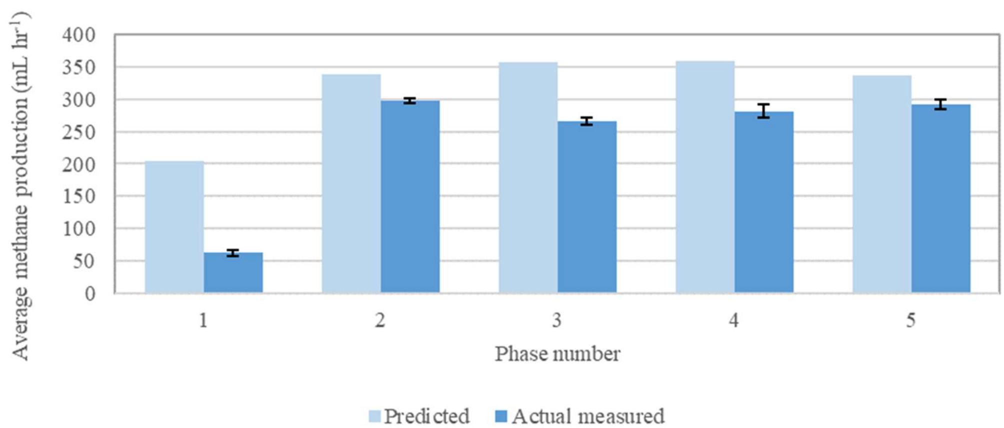


Figure 5-41: Digester 1 predicted versus actual average methane production in each phase, using a VS destruction rate of 85%.

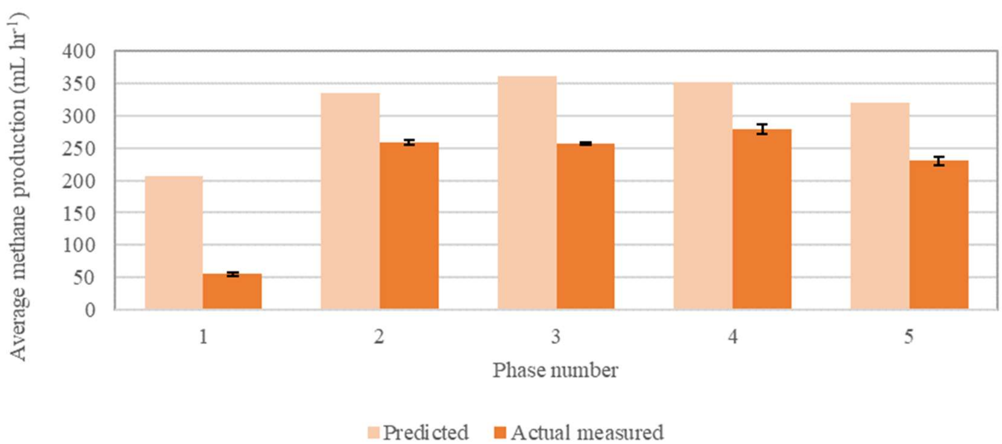


Figure 5-42: Digester 2 predicted versus actual average methane production in each phase, using a VS destruction rate of 85%.

The predicted versus actual methane production in the digesters can be used as an indicator of the %VS destruction. A higher %VS destruction will give a higher methane production, and vice versa (table 5-8).

Table 5-8: Actual methane production as a percentage of predicted methane production for both digesters during each phase, using a VS destruction rate of 85%.

	Digester 1 actual methane production as a % of predicted	**Digester 2 actual methane production as a % of predicted**
Phase 1	36%	31%
Phase 2	103%	91%
Phase 3	88%	84%
Phase 4	92%	93%
Phase 5	102%	84%

In phase 1 for both digesters, the measured methane production is significantly below that predicted by the mass balance. In this phase, the digesters were acclimatising to the new feed and so would not be performing optimally, resulting in a lower %VS destruction and lower than predicted methane. However, in phase 4, where both digesters were acclimatised and stable (sections 5.3.1 and 5.3.2), the biogas production was also lower than expected. This suggests that the %VS destruction used in the mass balance was not correct.

The mass balance was used to calculate the actual %VS destruction for each phase for both digesters, by altering the %VS destruction until the predicted and actual methane production were equal (table 5-9).

Table 5-9: VS destruction in each phase for digesters 1 and 2.

	Digester 1	**Digester 2**
Phase 1	30%	27%
Phase 2	88%	77%
Phase 3	75%	71%
Phase 4	79%	79%
Phase 5	87%	72%

From these results it can be seen that digester 1 outperformed digester 2 in terms of VS destruction in all phases except phase 4. As they were under the same conditions, the VS destruction should have been the same in both digesters for phases 1, 2 and 3. This could be the result of the heterogeneity of anaerobic digester reactors, which has been observed in other studies and was shown to produce unpredictable and unaccountable differences in performance in digesters under the same conditions (Lv *et al.*, 2014b). In phase 4, the %VS destruction for digester 1 was lower than in phases 2 and 5, which could be the result of the variable loading rate.

5.3.11 Methane concentration in the biogas

The quality of the biogas from each digester for each phase (figure 5-44) shows that the methane concentration was consistently higher in digester 1.

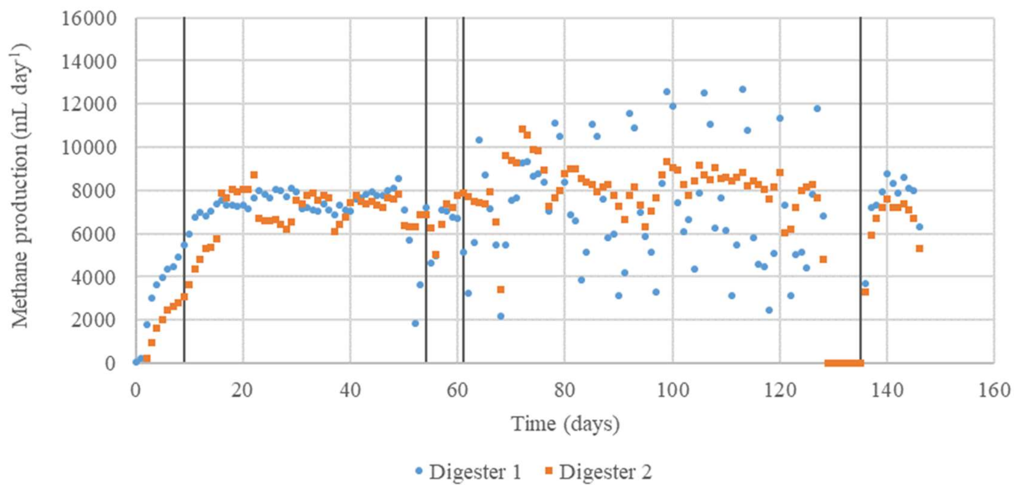


Figure 5-43: 1-day average methane production for digesters 1 and 2 during the whole experimental period.

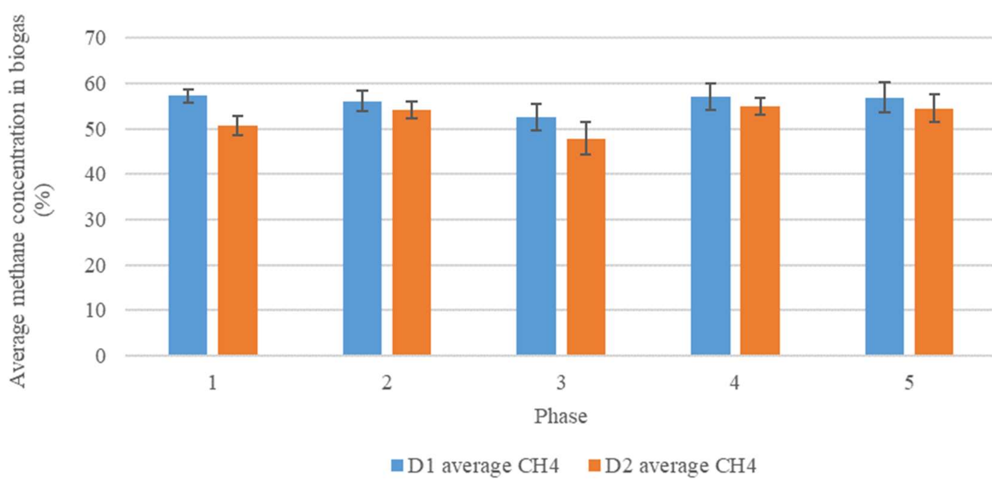


Figure 5-44: Average methane concentration in biogas (%) for digesters 1 and 2 for each phase.

In stable conditions in a digester, the methane concentration is affected by the feedstock composition and how well the different stages of the anaerobic digestion process are functioning (Wellinger, Murphy and Baxter, 2013). As the feedstock was the same for both digesters, but the methane concentration was higher in digester 1, this suggests that the digestion processes were working better in digester 1. However, considering that this was the case throughout the experimental period, it is not possible to conclude how the fluctuating loading rate in digester 1 affected this process from these results. Other studies on variable loading have not

5.3.12 Total ammoniacal nitrogen (TAN)

The TAN concentration in the digesters was measured weekly throughout the experimental period (figure 5-45).

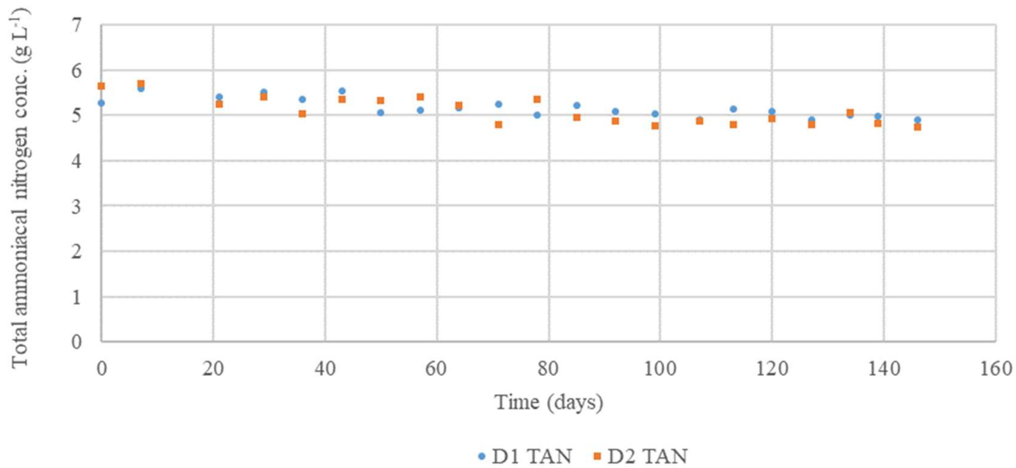


Figure 5-45: TAN measurements for digesters 1 and 2 during the experimental period.

The TAN concentration that is inhibitory to an anaerobic digester depends on the rate of throughput (high-rate digesters have a lower tolerance to ammonia inhibition) and the feedstock, and ranges from 2.5 to 11 g L^{-1} (Yenigün and Demirel, 2013). The TAN concentration in digesters 1 and 2 was 5.38 and 5.64 g L^{-1} respectively at the start of the experimental period and 4.89 and 4.75 g L^{-1} respectively at the end, showing a general downward trend and no signs of build-up or inhibition.

The ammonia inhibition in other studies of anaerobic digestion of food waste (Banks *et al.*, 2012; Walker *et al.*, 2017) became evident after about a year of operation and was resolved by the addition of a trace elements solution. As this experiment was running for under 6 months and a trace elements solution was added to the feed, it was expected that no ammonia inhibition would be observed.

5.3.13 Operational issues

The overload test (phase 3) produced very little response in the two digesters (for example, in the IA and PA, figure 5-13, figure 5-14) and as a result this test was not repeated as had been planned. The results from this phase have not been presented in detail as there were no significant results.

The accidental overfeed on day 70-71 was effective in showing the response of the digesters to overload (figure 5-14, figure 5-15, figure 5-17), however this was not repeated as it would potentially have had an effect on the BMP test performed at the end of the testing period.

5.4 Conclusions

The most notable differences between digester 1 and digester 2 were in the biological methane potential, the build-up of volatile solids, and the response after feeding.

5.4.1 Response to feeding

The digesters showed a short-term (over a few minutes) and a long-term (over several days) response to feeding. In the short-term, the digesters both showed a sharp increase in biogas production after feed events. In digester 1, this biogas 'spike' was more pronounced than in digester 2, but only in phases 4 and 5, during and after the period of variable loading rate. This suggests that the variable loading rate led to a larger response to feed events, and this could have been as a result of the cultivation of a larger microbial population in digester 1. Despite the different magnitude, the speed of short-term response to feed events was the same in both digesters throughout the experimental period. It can therefore be concluded that a variable loading rate has no effect on the speed of response, at least when this particular feedstock is used.

5.4.2 BMP test

The BMP test showed that at the end of the experimental period, the digestate from digester 1 had a higher biological methane potential than the digestate from digester 2. In phase 5, digester 1 produced more biogas, with a higher methane content, and this could have been the result of the variable loading in phase 4. A build-up of VS was shown by the blank BMP test for digester 2 when compared to digester 1.

5.4.3 Volatile solids

The VS and TS measurements at the end of the experimental period were lower than predicted for both digesters, and were still increasing, showing that the TS and VS for both digesters had not stabilised. However, there was a faster rate of increase towards the predicted VS in both digesters compared to the rate of increase towards the predicted ash content (from the mass balance), particularly in digester 2. This could be explained by the growth of the microbial population or undigested feed in the digestate. The specific methane production for blank digestate in the second BMP test (figure 5-38) showed that there was a build-up of VS from undigested feed in digester 2, but that this effect was not as great in digester 1. This could be because digester 1 was more efficiently converting VS to biogas due to the varied loading rate in phase 4. This is supported by the response to feed events in digester 1, which suggested that there was a larger microbial population in digester 1 than digester 2 as a result of the variable loading rate pattern.

The % VS reduction was variable in different phases for both digesters, with the best % VS destruction achieved consistently by digester 1. The VS destruction achieved was lower than the average for food waste, but within the known range (Banks, 2009; Paritosh *et al.*, 2017). A build-up of VS in digester 2 compared to digester 1 was shown by the blank BMP test. This may have been due digester 2 containing a lower microbial population than digester 1. However, as digester 2 consistently performed less well than digester 1 (lower methane production, lower methane concentration in biogas), it is not possible to determine this without further research.

5.4.4 Effect of loading rate fluctuations

The loading rate fluctuations in digester 1 during phase 4 (after the change in feed regime) produced a pattern of methane concentration fluctuations in the biogas that became more predictable and less pronounced towards the end of this phase. This showed an acclimatisation to the changed regime and took roughly 46 days (section 4.3.5).

The long-term effect of the difference in feed regime in the two digesters is shown by the partial alkalinity, methane concentration and the BMP. In phase 5, when both digesters had been returned to the same loading rate, digester 1 showed a higher concentration of methane in the biogas and a higher methane production. Digester 1 also showed a higher BMP for both cellulose and SFW compared to digester 2.

5.4.5 Relationship between loading rate and methane concentration

Over the duration of the experiment, the methane concentration varied in a roughly linear way with loading rate; however, there was not a strong relationship between these factors, possibly because of the complex nature of the anaerobic digestion process.

5.4.6 Acclimatisation and stability

The digesters both took roughly 39 days to acclimatise, which was best indicated by the propionic and acetic acid levels and supported by the stabilisation of the biogas methane concentration in roughly the same period of time. The partial and intermediate alkalinity and the alkalinity ratio showed that the digesters became acclimatised after approximately 60 days.

This may show that the digesters had a standard ‘stabilisation period’ and this concept may be worth pursuing in further research, to see what conditions might increase or reduce this period.

The indicators of stability (alkalinity ratio, partial and intermediate alkalinity, biogas methane content, VFAs) have been discussed by previous publications (Wu *et al.*, 2019; Boe *et al.*, 2010). From these references and from the work in this chapter it is reasonable to conclude that the best form of stability monitoring for an anaerobic digestion system is a combination of indicators rather than a single indicator. For example, the biogas composition would show short-term organic overloads, the propionic to acetic acid ratio would give an early warning indicator of an imbalance, and a drop in the partial alkalinity would show a depletion of the buffering capacity within the digester. Limitations

During the experimental work, the performance of the two digesters was different when they were under the same conditions. Digester 1 reached the state of steady biogas production after approximately 11 days, whereas digester 2 reached this state after 30 days. Additionally, digester 2 consistently produced less biogas than digester 1 in all phases. This was not obvious until a trend had formed over the course of several weeks and calculations had been done on the readings to draw an average. Ideally when this trend was noted the experiment would have been re-started with a different inoculum, or re-mixed and allowed to stabilise for a longer time. However, this was not possible as the experimental work had already been severely delayed, and a re-start would have not left time to gather sufficient experimental data. Therefore, the decision was made to continue, and consider this difference during analysis.

Problems were noted in the GC analysis, as the GC column had been in use for some time, and tests for accuracy produced variable results from the same sample. Steps were taken to improve the quality of the results such as shortening the column, washing the column with a solvent, running blanks and running repeats of standards. Ideally, the column would have been replaced and access by other users would have been restricted during the experimental process, but this was not feasible with the available resources.

5.4.7 Further research

Further research in this area could usefully study whether certain feedstocks are easier to acclimatise to than others, could quantify what the acclimatisation time is for each, and whether it can be shortened by changing conditions. It may also be possible to characterise an individual digester in terms of its stabilisation period and what factors might affect this.

To support the hypothesis that the variable loading rate caused an increase in the microbial population rather than a change in the types of microbial species present, genetics techniques such as 16s rRNA sequencing could be used to compare the microbial populations (in terms of size and variety) in the two digesters. The test could for example be used to find out whether certain species were particularly prevalent in one of the digesters.

6 Case study: Assessment of Micro-Scale Anaerobic Digestion for Management of Urban Organic Waste

6.1 Introduction

This chapter presents a case study of an anaerobic digestion system that was built as a demonstration of micro-scale anaerobic digestion in an urban environment. The plant was built in 2013 and operated until 2019, processing urban food waste and generating biogas for use in a community café. The system was monitored for a period of 319 days during 2014. The resulting case study provided real data of the experimental issues studied in Chapter 5, namely, the changing type and amount of feedstock available and the effect on the stability and performance of the plant. The results showed that the plant was capable of stable operation despite large fluctuations in the rate and type of feed.

The case study included an energy balance and economic analysis of the system, which were used to inform the work on the techno-economics of micro-scale anaerobic digestion that is presented in chapter 7.

6.2 Publication

The work in this chapter is an abridged version of a jointly authored publication. The details of the publication are as follows:

WALKER, M., THEAKER, H., YAMAN, R., POGGIO, D., NIMMO, W., BYWATER, A., BLANCH, G. & POURKASHANIAN, M. 2017. Assessment of micro-scale anaerobic digestion for management of urban organic waste: A case study in London, UK. *Waste Management,* 61, 258-268.

6.3 Author's contribution

The author (Helen Theaker) was given a complete set of data from the monitoring of the site and asked to write an academic publication describing the project. This required reformatting to ensure that the time reference points corresponded between readings from different sources so that they could be analysed together. The author analysed the data to create the featured graphs, interpreted the data and discussed the research in the paper with input from the other authors. The main other contributor to the paper was Dr. Mark Walker, who was the leader of the initial study along with Dr. Davide Poggio. Guy Blanch from GDDB and James Murcott of Methanogen co-managed the design and commissioning of the plant. The data was collected automatically through a monitoring system, designed, and installed jointly by Clive Andrews of Aleka Design Ltd., Dr Mark Walker and Dr Davide Poggio, alongside data collected manually by Rokiah Yaman (site manager) and volunteers on the site. The original project resulted in a report published by WRAP (Yaman *et al.*, 2016). The journal paper was reviewed by Professor Bill Nimmo at the University of Sheffield and Angela Bywater of Southampton University, who also made significant contributions during the setup and monitoring phases. Professor Mohammed Pourkashanian was principal investigator on the original research grant funded by WRAP under the DIAD II (Driving Innovation in Anaerobic Digestion, phase 2) scheme, at the time of the University of Leeds, and later at the University of Sheffield.

The author would like to declare that although this chapter is largely her own work, there is a significant amount of overlap with the jointly published paper noted above.

6.4 Site description

The digester system was designed and installed by a consortium of companies and researchers in 2013, and the monitoring took place from October 2013 to November 2014. The plant was built within the grounds of the Camley Street Natural Park in London, UK and the site was used to convert locally produced organic waste, collected by bicycle, into biogas for heating and electricity. The system (figure 6-1) included a 2 m^3 digester (Methanogen Ltd., UK) and a pre-feed system consisting of a chopper mill, a 0.7 m^3 mixed 'pre-digester' tank on load cells and a feeding pump (Guy Blanch Bio Development Ltd, UK). The digester and ancillaries were housed in a greenhouse, denoted by a dotted line.

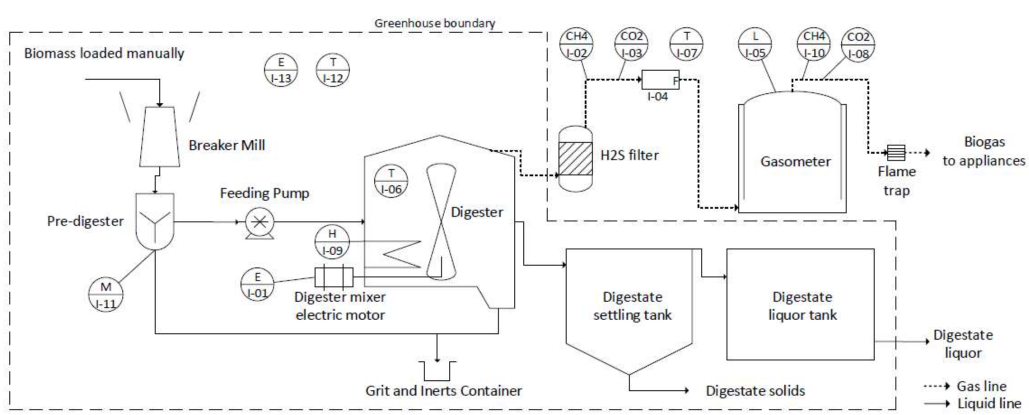


Figure 6-1: Schematic of equipment at micro-AD site.

6.5 System operation

The type and amount of feedstock were very variable, due to a sporadic collection routine. The system was designed with a pre-digester to smooth out these variations. The main feedstocks that were added to the pre-digester tank during the monitoring period can be separated into four phases (table 6-1). The digester feed was nominally 15-20 kg day^{-1}.

Table 6-1: Description of the four phases of feedstock supply to the micro-AD plant.

Phase	Days	Feedstocks
1	1 to 15	Apple pomace, café waste, coffee waste, water
2	16 to 107	Catering waste, coffee, water
3	108 to 294	Catering waste, oats, soaked compost bin liners, water
4	295 to 399	Catering waste, soaked compost bin liners, water

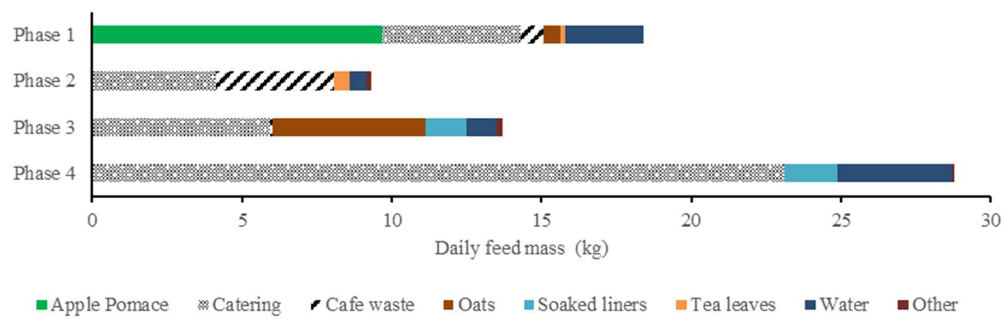


Figure 6-2: Feedstocks added to the pre-digester in each phase.

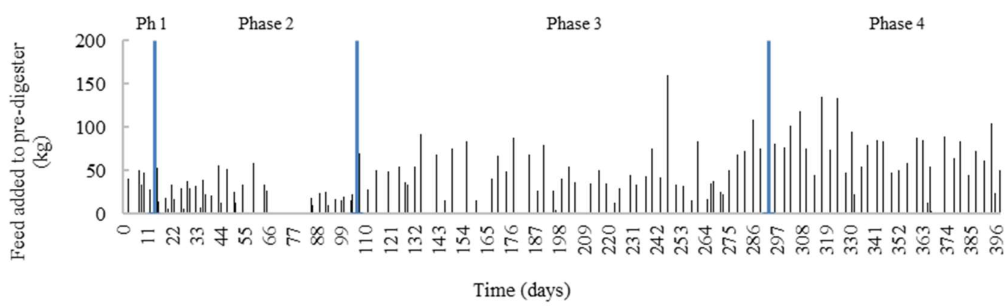


Figure 6-3: Mass of feed added to the pre-digester on each day.

6.5.1 Automatic monitoring using sensors and cloud-based logging software

The system was automatically monitored in real time by a suite of sensors connected to data acquisition hardware. The sensors that were used to record data were as follows: biogas production (Elster BK-G2.5 Diaphragm gas flow meter), methane and carbon dioxide content of the biogas at both the digester outlet and at the system outlet (Dynament NDIR CH_4 sensor, Dynament NDIR CO_2 sensor), temperatures of the digester, greenhouse and outside ambient

(Atlas Scientific ENV-TEMP thermistor), electrical consumption of the site (ISKRAEMECO ME162 electricity meter) and digester (Finder 7E.13 electricity meter), heat consumption of the digester (Superstatic 449 heat meter) and incident solar radiation on the greenhouse (APOGEE CS-300 Pyrometer). In addition, biogas oxygen (ITG-103 electrochemical sensor) and hydrogen sulphide (ITG I-46 electrochemical sensor) composition were measured intermittently.

Calibration of the biogas composition sensors was performed every 2 months using a calibration gas containing 35% carbon dioxide, 1% oxygen, 50ppm hydrogen sulphide and balance methane. All other sensors were pre-calibrated from the factory.

The customised PC data logging software was developed using a commercial program called DAQFactory and data was made available online through the DAQConnect website, for data sharing amongst the project team.

6.5.2 Laboratory-based testing of pre-digester and digestate

Samples from both the pre-digester tank (feedstock) and the digester output (digestate) were taken by the operator, frozen and sent to be analysed. Total and volatile solids were measured as per standard methods (APHA, 1998), pH was measured with a Hach pH meter and probe. VFA were measured using an Agilent 7890A gas chromatograph, with a DB-FFAP column of high polarity designed for the analysis of VFAs, as per the manufacturer's guidelines. Elemental content was determined using an elemental analyser (Flash EA2000, CE Instruments) equipped with a flame photometric detector (Flash EA 1112 FPD, CE Instruments). Alkalinity was measured by titration using endpoints of 5.75 (partial) and 4.3 (total) with intermediate alkalinity being the difference between the partial and total alkalinities. Theoretical COD (Chemical Oxygen Demand) was calculated from the empirical formula obtained from elemental analysis, considering the organic matter to be fully oxidised to CO_2 and water, with N being reduced to ammonia and S oxidised to sulphuric acid (Baker, Milke and Mihelcic, 1999).

6.6 Results and Discussion

6.6.1 Operational key performance indicators

The system treated 4.4 tonnes of waste material over a period of 319 days meaning that during that time the system was treating a nominal 5.1 tonnes yr^{-1}. Water addition, used to facilitate the maceration of the waste, was nominally 0.6 tonnes yr^{-1}. Over the whole testing period, the average daily feed was 12.6 kg day^{-1} which is equivalent to an OLR of 1.6 kgVS m^{-3} day^{-1}. The specific methane yield was 132.4 m^3CH_4 $tonneVS^{-1}$. This is fairly low compared to a standard EU food waste value of 450 m^3CH_4 $tonneVS^{-1}$ (table 2-5, table 6-2).

Table 6-2: Key performance statistics for the micro-AD plant from day 80 to day 399

Measurement	Value	Unit
Average daily feed amount	12.6	kg day^{-1}
Average daily VS added	3.11	kg day^{-1}
Average OLR	1.6	kg VS m^{-3} day^{-1}
Average water added	1.7	kg day^{-1}
Average daily biogas production	3.15	m^3 day^{-1}
Specific daily biogas production	1.7	m^3_{biogas} $m^{-3}_{digester}$ day^{-1}
Total mass of food added	4422	kg
Specific biogas yield	227.9	m^3 tonne fresh $matter^{-1}$
Specific methane yield	132.4	m^3 CH_4 tonne VS^{-1}
Average biogas methane content	60.7	%
Average daily methane production	1.91	m^3 day^{-1}
Average HRT	144.8	days
Operational period	319	days
Average digester temperature	33.1	˚C

6.6.2 Analysis of the pre-digester tank

The effect of the pre-digester tank is that waste loading events (waste added to the pre-digester tank) were decoupled from the feeding events (into the digester) by the mixing of the feedstock into the existing contents of the pre-digester tank.

The small size of the installation means that it is possible to have a relatively large pre-digester tank (compared with the main digester). This means that the period of 'feed buffering' is relatively long compared with a conventional large-scale AD plant, where building such a large pre-digester tank would be uneconomical. In this case, the ratio between their volumes was 1:3 (pre-digester: digester). As food waste is known to be a highly variable feedstock (Fisgativa, Tremier and Dabert, 2016), this represents a useful advantage to the micro-scale application.

Due to the variation in availability of feedstocks, the composition in the pre-digester tank varied significantly over the project period, mainly between predominantly food waste, and a mixture of food waste and oats. The composition was reflected in the measured total and volatile solids contents in the pre-digester (figure 6-4). During the period of oats being fed into in the pre-digester tank (phase 3, days 108 to 294) the TS of the pre-digester rose from

22% to 37%, and then fell during phase 4, when predominantly food waste was added to the pre-digester tank. In the same figure, the VFA and pH of the pre-digester dropped.

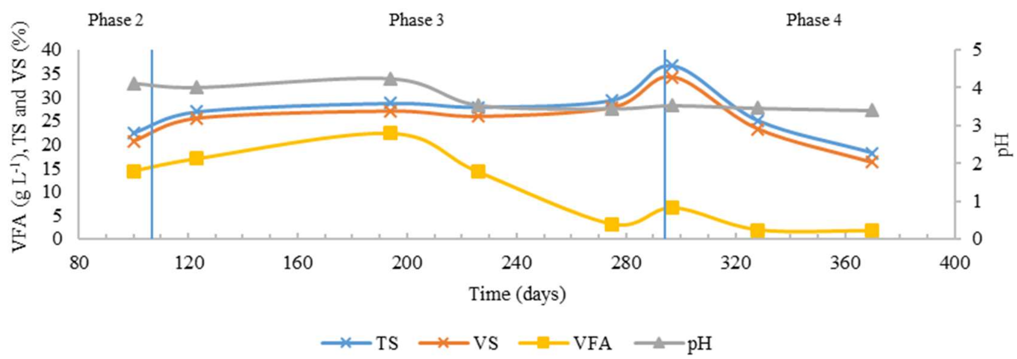


Figure 6-4: Laboratory analysis of the pre-digester tank.

The VFA concentration in the pre-digester tank is an indicator of the amount of hydrolysis and fermentation taking place. This peaked in phase 3 at around 22.4 g L^{-1}. After this point, a reduction in the VFA concentration is observed, likely to be a consequence of the decrease in pH leading to an inhibition of fermentation, analogous to ensiling. The low pH environment in the pre-digester tank is such that the formation of methane by methanogenic organisms can be ruled out since these organisms cannot grow under these conditions (Angelidaki, Ellegaard and Ahring, 2003). The fermentation taking place in the pre-digester tank is not particularly advantageous to the digester except for perhaps a slight increase in the rate of methane production, however it can lead to the generation of a large amount of odour which could be considered a disadvantage.

The average elemental composition of the feedstock was 49.0, 34.8, 6.2 and 2.92 (% by mass of TS) of C, H, O and N respectively, giving a C:N ratio of 14.4:1.

6.6.3 Digester characterisation

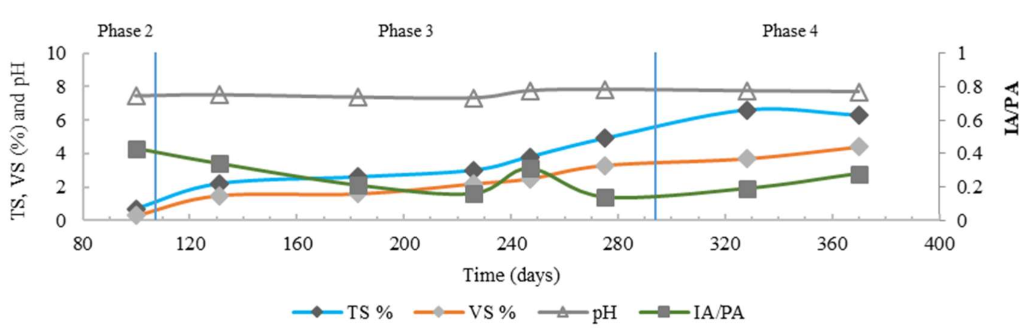


Figure 6-5: Total solids, volatile solids, pH, and alkalinity ratio analysis of the digestate during phases 2 to 4 (days 100 to 400).

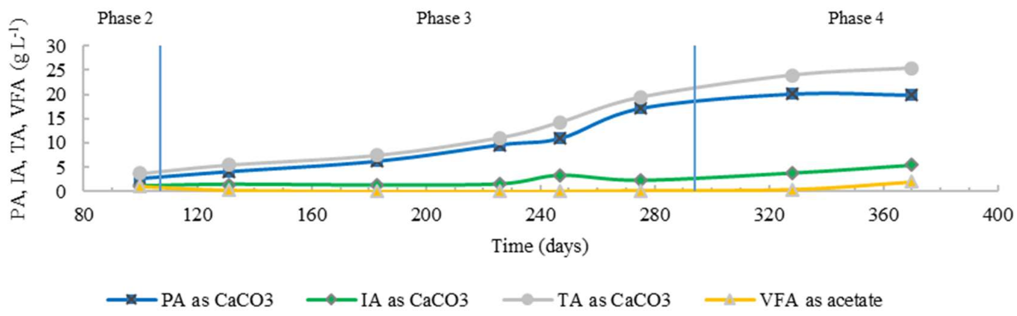


Figure 6-6: Partial, intermediate and total alkalinity and volatile fatty acids analysis of the digestate during phases 2 to 4 (days 100 to 400).

A summary of the laboratory analysis of the digester contents (figure 6-5) shows a general increasing trend in TS and VS as the initial inoculum (diluted digestate and cattle slurry) was replaced with the mixed waste feedstock. The trend appears to have levelled off by day 400, indicating the arrival at a steady state of the system in terms of mass balance, albeit dependent on the input moisture content and added water. The digestion process appears stable throughout the testing period. The process is characterised by stable pH (well within the optimum range for the growth of methanogens (Gujer and Zehnder, 1983)), a gradual increase in partial and total alkalinity (figure 6-6) and generally low (<0.5 g L^{-1}) VFA concentrations after the initial acclimatisation period.

The average temperature of the digester during the testing period was 35.6 °C and stayed within ±2 °C of this, despite large changes in the ambient and greenhouse temperatures, indicating that the mixing and heating systems were successful.

6.6.4 Biogas production

There were variations in biogas production per unit feed over the project period, caused predominantly by variations in the composition and amount of feedstock added to the system. This variation was caused by differing quantities and types of waste added to the pre-digester tank, seasonal variation in the composition and moisture content of the waste, and variations in the amount of water added to facilitate the maceration of the feedstock. While all of these factors could have a large effect on the quantity and quality of biogas produced, the pre-digester tank was designed to smooth out these fluctuations and reduce their impact on the biogas production rate.

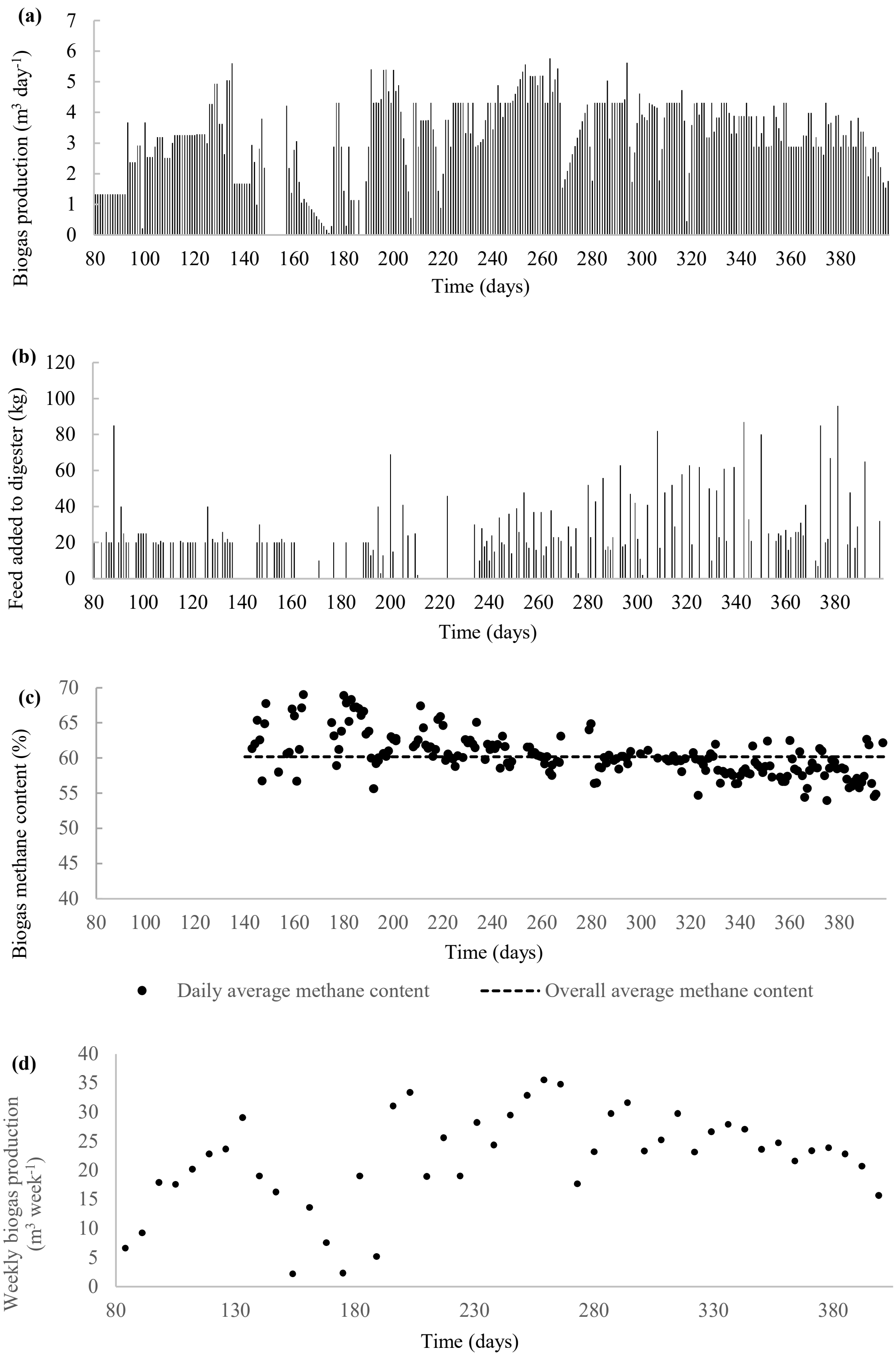


Figure 6-7: (a) Biogas production (b) feed added to the digester (c) biogas methane content and (d) weekly biogas production during the test period.

The biogas production of the system is highly variable on a daily basis (figure 6-7a), whereas the weekly trend shows a gradual increase reaching around 4-5 m^3 day^{-1} up to day 289, after which there was a gradual decrease in the biogas production from the system (figure 6-7d). The methane content of the biogas (figure 6-7c) shows less daily variation but over the course of the project the trend was a gradual decrease from around 65% to around 57%. To understand the reason for these trends, further analysis would be required. It is possible that the change in the feedstock composition led to a reduction in the methane content of the biogas, but it could also be an early sign of process instability (Lv *et al.*, 2014a). This is discussed further in section 6.6.3.

It can be said that the decrease in methane production was not caused by a reduction in the overall feed to the system (which remained fairly constant from around day 235 until the end of the testing period, at around 15-20 kg day^{-1}) but the decrease in VS of the mixed biomass in the pre-digester tank, which decreased from around day 297 onwards (figure 6-4). This would also contribute to the reducing biogas production. The decrease in VS was due to a change in feedstock from waste oats to food waste.

6.6.5 Ammonia inhibition and trace element dosage

The last sample of digestate analysed (on day 370) indicated potential stress, by high VFA and dropping methane concentration in the biogas. For this reason, further samples of the digestate were taken for analysis beyond the official testing period (figure 6-8).

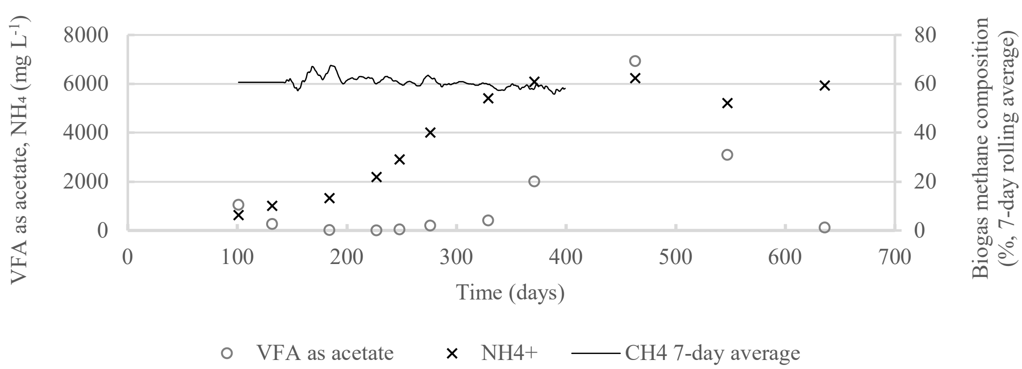


Figure 6-8: Digestate VFA and ammonia concentration, and methane content of the biogas

Testing showed a rise in ammonia concentration and a subsequent rise in VFA concentration and drop in methane content in the biogas. The feedstock being supplied to the digester at this point was mainly food waste, and this feedstock type was fed in from day 294 (the start of phase 4). The IA/PA ratio was also measured in the digestate samples (figure 6-6), and this stayed low throughout the whole monitoring period indicating process stability (Ripley, Boyle and Converse, 1986).

A rise in ammonia concentration has been noted in food waste digesters previously and can be the initial signs of a long term (>1 year) failure of the process, caused by a combination of ammonia inhibition of acetoclastic methanogens with deficiencies in certain trace elements blocking both propionate oxidation and syntrophic hydrogenotrophic methanogenesis (Banks *et al.*, 2012). Acting on the theory that this situation could be resolved by addition of trace nutrients to the system, the required addition of trace elements was calculated (table 6-3).

Table 6-3: Trace element addition for other sites and this site.

Element		**Mo**	**Ni**	**W**	**Se**	**Co**
Suggested addition (Banks *et al.*, 2012)	mg L^{-1} wet	0.2	1	0.2	0.2	1
Recalculated based on TS=23.7% (Banks *et al.*, 2012)	mg kg^{-1} TS	0.8	4.2	0.8	0.8	4.2
Average added (Facchin *et al.*, 2013)	mg kg^{-1} TS	6	10	1	1	10
Values adopted at micro-AD site	mg kg^{-1} TS	4	5	1	1	5
One-off dose to pre-digester	g	1.2	1	0.2	0.2	1
One-off dose to digester	g	0.72	0.6	0.12	0.12	0.6
Dosage every 2 months	g	1.73	1.44	0.29	0.29	1.44
Source compound		$(NH_4)_6Mo_7O_{24}.4H_2O$	$NiCl_2.6H_2O$	$Na_2WO_4.2H_2O$	Na_2SeO_3	$CoCl_2.6H_2O$
Element by weight	%	54	25	56	46	25

A dose of trace elements solution was added to the digester on day 476, followed by doses at two-monthly intervals afterwards. Following the addition, the VFA concentration in the digester dropped to 112 mg L^{-1} on day 636, which is well within the acceptable range (Wang *et al.*, 2009). The ammonia concentration did not drop as a consequence of the trace element addition, but instead the decrease in VFA appeared to indicate the methanogenic microorganisms were better able to metabolise in the presence of ammonia (they were more resistant to the toxic effect) when the correct proportions of trace elements were added, in agreement with previous studies (Banks *et al.*, 2012).

6.6.6 Heat consumption

Temperature data collected by the logging system can be used to analyse the bulk heat transfer characteristics of the micro-AD system (table 6-4).

Table 6-4: Heat consumption and temperature data.

Measurement	Value
Digester temperature (˚C)	32.9
Greenhouse temperature (˚C)	23.7
External temperature (˚C)	15.0
Heat input to digester (W)	79.7
Digester surface area (m^2)	10.2
Incident solar radiation (W m^{-2})	43.3

The temperature of the digester was maintained by the addition of heat via an internal hot water heat exchanger. The heat demand was measured by a heat meter, along with the average temperatures in the system, and had an average value of 80W over the logging period. The average incident solar radiation was measured by a sensor on the roof of the greenhouse. The digester temperature was controlled throughout the project by a thermostatic controller operating the hot water valve to the heat exchanger. The temperature of the digester was approximately constant throughout the project, therefore the heat loss from the digester can be equated to its heat input. The heat loss has conductive, convective and radiative elements although for this analysis they are simply grouped together to give an overall heat loss value and overall heat transfer coefficient.

Using monthly data for temperature and heat use on the heat meter, the heat transfer coefficient (K) can be calculated using the equation $\dot{Q} = K\Delta T$, where $\dot{Q}$ is the heat loss (W), K is the overall heat transfer coefficient (W ˚C^{-1}) and ΔT is the temperature difference (˚C). This equation can be used with the average temperature difference between digester and greenhouse to give the digester overall effective heat transfer coefficient (K_d), and the difference between the digester and ambient temperatures to give the overall effective heat transfer coefficient for both the digester and greenhouse together (K_b).

K_d had an average of 8.7 W ˚C^{-1} (8.0-9.5 with 95% confidence) giving the digester a U-value of approximately 0.85 W m^{-2} ˚C^{-1} (using a surface area of 10.2 m^2, from table 6-4). The heat demand varies in the range 39.1-111.5 W over the logging period, although given the mild winter conditions, this could be expected to increase to around 121 W with an average ambient winter temperature of around 4.4 ˚C and higher in severe winter conditions. K_b was estimated at 4.2 W ˚C^{-1} (3.5-5.0 with 95% confidence).

Using both of these average heat transfer coefficients, an approximation can be made of the energy savings given by housing the digester in the greenhouse.

To assess the heating effect of the greenhouse, the calculations for heat demand above can be repeated, instead using the difference between the digester temperature and the ambient temperature.

The measured heat demand, theoretical (calculated) heat demand, and estimated heat demand without the greenhouse were assessed (figure 6-9).

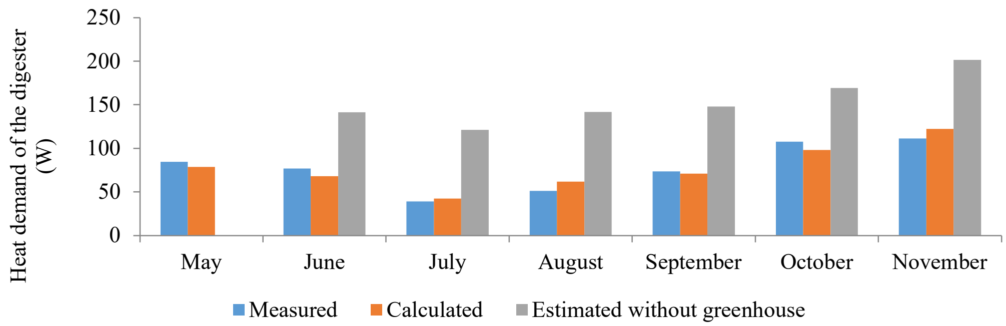


Figure 6-9: Temperature and heat demand of the digester during the testing period. No estimated data is provided without greenhouse in May as the thermistor was not installed at that time.

Based on this analysis, the overall heat savings of putting the digester inside a greenhouse were an average of 49% or 76.6 W.

6.6.7 Electrical consumption

The system had two electrical meters, M1 and M2. M1 measured only the energy consumed by the digester mixing motor. M2 measured the complete consumption of site, including the digester mixing motor, the pre-digester system (macerator, pre-digester tank mixing motor, feeding pump), the logging system (sensors, data acquisition hardware, PC) and in addition a number of other electrical demands not associated with the AD system. These included lighting, charging of power tools, other developmental work occurring at the site and any other plug-in appliances in either the greenhouse or the monitoring room including laptops, PC, phone chargers and kettles. The average electrical consumption of M2 was 150W. The electricity consumed by the digester mixing motor (M1) remained fairly constant throughout the logging period with an average of 54W continuous.

Table 6-5: Electrical consumption data.

Measurement	Value
M1: Electrical demand of digester (W)	53.5
M2: Electrical input to site (W)	149.7

To further break down the electrical use of the site, estimates for the micro-AD system electrical demand have been made based on manual measurements of the separate items in the system (table 6-6). Note that the logging system power consumption has been calculated as the residual power that was measured by M2 and is not accounted for by other components. The other electricity uses mentioned previously, outside the plant, have been assumed to be negligible in order to give the worst case estimated power consumption for the plant only. The actual electricity use of the plant will therefore in reality be slightly lower.

Table 6-6: Estimated electricity demand of AD system based on rate power demand and estimated duty cycle

Component	**Power demand cycle**	**Demand (Wh day^{-1})**	**Equivalent continuous power (W)**
Chopper Mill	1.5kW, 5 min/24 hr	125	5.21
Pre-digester mixing	0.18kW, 10 min/24 hr	30	1.25
Digester feeding pump	72W, 1 min/2 hrs	14	0.60
Extraction (greenhouse)	25W, 18 min/hr	180	7.50
Extraction (monitoring room)	25W, 12 min/3 hrs	40	1.67
Digester mixing (measured)	N/A measured	1284	53.5
Logging system (calculated)	N/A		80
Total (whole site)		**TOTAL**	**149.7**
Total (plant only)		**TOTAL**	**69.7**

6.6.8 Coefficient of performance

Using the data collected by the logging system, an energetic analysis was performed on the micro-AD system (table 6-7). The analysis included the measured energy inputs of heat and electricity as well as the measured outputs of biogas quantity and methane percentage. In order to add relevance to the results, a hypothetical CHP has been included as the biogas appliance with a low electrical efficiency of 25% and heat recovery efficiency of 50%, which are realistic for the scale considered. Using the lower caloric value (LCV) of the methane stated, the methane production has been converted to an average power in watts to give nominal values for net energy output of the CHP and coefficients of performance (COP).

Table 6-7: Energy mass balance for micro-AD site (based on LCV of methane = 40 MJ m^{-3})

Energy output of micro-AD system	
Methane production (m^3 day^{-1})	1.91
Gross energy production in biogas (MJ day^{-1})	76
Gross power output in biogas (W)	884
CHP	
Electrical power output (W)	221
Heat power output (W)	442
Net output power of AD system	
Electricity (whole site) (W)	71
Electricity (plant only) (W)	151
Heat (W)	362
Coefficients of performance (COP)	
Electricity (whole site)	1.48
Electricity (plant only)	3.17
Heat	5.55
Heat (without greenhouse)	2.72

The results show all COPs are greater than 1, indicating a positive energy balance. The plant on its own (without the logging system) has an electrical COP of 3.17 due to its low parasitic electrical requirements. However, when the additional load of the rest of the system is included, this is reduced to 1.48. The relatively high continuous electrical demand of the logging system reduces the electrical COP of the site and it is clear that reduction of this demand would be necessary in further system developments, either through optimisation or through minimisation the system components, to allow continuous logging to be feasible on a micro-AD system.

The high COP on a heat basis (5.55) can be attributed to the performance of the insulation of the digester and the effect of housing the digester in a greenhouse. The solar gain of the greenhouse reduced the heat demand by 49% and therefore an estimate of the coefficient of performance of the digester without the greenhouse can be calculated as 2.72.

In terms of parasitic loads, the plant uses 31.7% of the total electricity production, whereas the whole site uses 67.8% of the total electricity production, and the heat requirement is 18% of the total heat production.

6.6.9 Avoidance of greenhouse gas emissions

Table 6-8 summarises the carbon emissions balance for the plant. An explanation of the carbon emission categories follows:

- The annual methane production of 697 m^3 could result in carbon dioxide reduction of 1411 kg yr^{-1} relative to the same consumption of natural gas based on DEFRA/DECC estimates (DECC, 2016).

- The diversion of 5.3 TPA (tonnes per annum) of organic waste from landfill could result in a carbon reduction of 2724.5 kg yr^{-1} (WRAP, 2011).

- Abated waste transport was calculated by assuming the normal route for food waste would be transport of an average 56 km round-trip in an articulated lorry that could hold 40 tonnes based on UK figures from (WRAP, 2016). This generated a relatively small emissions saving of 13.5 kg yr^{-1}.

- Carbon dioxide emissions savings are also made by using digestate instead of conventional inorganic fertilisers. Of the 4357 kg yr^{-1} added as feed, 1185 kg yr^{-1} was lost as biogas. Taking into account the water added, the digestate production was an estimated 4867 kg yr^{-1} , which would result in a 146 kg yr^{-1} carbon dioxide emissions saving (WRAP, 2012a).

- Using the AD system electrical and heat demand, the consumption of 611 kWh yr^{-1} of electricity and 698 kWh yr^{-1} of heat can be associated with emissions of 250 and 160 kg yr^{-1} (DECC, 2016) of carbon dioxide respectively.

- The net carbon reduction of the AD system was 3885 kg yr^{-1}, 2.93 kg CO_2 kWh^{-1} electricity production or 0.762 kg CO_2 kg^{-1} waste treated.

Table 6-8: Greenhouse gas balance for the plant.

Item	Associated CO_2 emissions	Reference	CO_2 saving kg yr^{-1}
Methane produced, for use in CHP	2.0245 kgCO_2 m^{-3}	(DECC, 2016)	1411.0
Diversion of waste from landfill	500 kgCO_2 $tonne^{-1}$	(WRAP, 2011)	2724.5
Reduction in transport	2.7 kgCO_2 tonne $waste^{-1}$	(GOV.UK, 2015)	13.5
Displacement of artificial fertilisers	30 kgCO_2 tonne digestate	(WRAP, 2012a)	146.0
Use of electricity	0.40957 kgCO_2 kWh^{-1}	(DECC, 2016)	- 250.2

Heating the digester	0.20405 $kgCO_2\ kWh^{-1}$	(DECC, 2016)	- 160.1
NET CARBON EMISSIONS AVOIDANCE ($kgCO_2\ yr^{-1}$)			**3884.7**

6.6.10 Operational observations

Anecdotal evidence given by operators stated that although representing an additional workload, collection of the daily readings enabled the site staff to engage more effectively with the workings of the plant and learn more about the processes involved.

Key lessons learned during the testing period were as follows:

- Space: Due to its location, the site had a very limited space available for the installation and this led to very little room for maintenance and 'housekeeping'. This made the operation of the plant unnecessarily difficult and should be avoided in future.

- Pre-digester: The pre-digester tank provided very useful storage which enabled the operators to add feedstock when it became available, often only twice a week.

- Odour: Odour was a problem with some feedstocks, which was overcome by better sealing of the pre-digester tank

- Noise: Noise is of particular concern in an urban area. The main source of noise pollution was the macerator and this equipment must be carefully chosen to avoid disruption to the surrounding area.

- Biogas use: Biogas was initially used in a biogas hob for making hot drinks but later in the project a custom built automated biogas boiler was installed. There are no type-approved 'off-the-shelf' heating appliances for biogas currently available in the UK. Later in the project a CHP sterling engine was installed.

- Digestate: Although it is a very valuable resource, demand for the digestate was limited and caused process issues throughout the testing period because of the limited number of potential outlets in an urban area. Careful consideration should be put in before a plant is built to identify a reliable outlet for the digestate.

6.6.11 Economic analysis

The economic analysis of the system is split into capital costs, operational costs and revenue. The predicted and actual costs were compared to gauge the accuracy of cost predictions in each area. The capital cost (table 6-9) was higher than predicted, mainly due to the need for an expensive logging system, a bespoke biogas boiler and CHP. Operational costs (table 6-10) were lower than expected but not by a significant amount. Revenue from the plant (table 6-11) was lower than expected, because the plant processed less feedstock than was predicted, which incurred lower gate fees.

Table 6-9: Predicted and actual capital costs.

Capital cost	**Predicted**	**Actual**
Monitoring system	£2,865	£2,865
Pre-feed system	£5,300	£4,950
Digester	£6,150	£6,150
Gas holder	£1,250	£1,250
Ancillaries	£2,320	£2,320
Gas use	£1,350	£9,500
Infrastructure	£1,500	
Commissioning	£1,000	£1,000
TOTAL CAPITAL COST	**£21,735**	**£28,035**

Table 6-10: Predicted and actual operational costs.

Operational costs	**Predicted**	**Actual**
Labour cost for prediction (£ hour^{1})	8	
Wages for operation (£ year^{-1})	1,460	1,248
Parts (£ year^{-1})	405	405
Maintenance (£ year^{-1})	40	40
Total operational costs (£ year^{-1})	1,905	1,693
Electricity cost		
Electricity cost (£ kWh^{-1})	0.10	0.10
Electricity use digester (£ year^{-1})	184	117
Electricity use for feed mill/mixing (£ year^{-1})	17	6
Electricity use for extraction (£ year^{-1})		8
Electricity use for monitoring (£ year^{-1})		91
Total Electricity Use (£ year^{-1})	201	223
TOTAL ANNUAL COSTS	**2,106**	**1,916**

*Table 6-11: Predicted and actual revenue (*based on calorific value of 41.2 MJ L⁻¹).*

Revenue	**Predicted**	**Actual**
Feedstock		
Feedstock (food waste) handled (kg day^{-1})	40	12.8
Feedstock (food waste) handled (kg year^{-1})	14,600	5,317
Methane production		
Cost of heating oil (£ L^{-1})	0.63	0.63
Methane to fuel oil conversion (L)*	1,292	813
Savings in fuel oil (£ year^{-1})	814	513
Digestate		
Standard value (from WRAP) (£ tonne^{-1})	4.46	4.46
Fertiliser savings (£ year^{-1})	65	24
Gate Fees		
Number of caddies collected	1,142	416
Caddy charge (£)	2.75	2.75
Total caddy income (£ year^{-1})	3,142	1,144
Landfill tax savings		
Landfill tax (£ tonne^{-1})	80	80
Diversion from landfill (£ year^{-1})	1,168	425
TOTAL REVENUE (£ year^{-1})	**5,189**	**2,106**

The system was able to cover its operational costs with its revenue generation from waste disposal, energy production and feed-in-tariff payments but required grant funding for its installation. In future systems it is expected that there are significant savings to be made from capital costs by increasing production volume and reducing monitoring requirements.

The system's levelized cost of energy (£ kWh^{-1}), based on these numbers, is variable according to the energy technology used (table 6-12).

Table 6-12: Economic parameters for the Camley Street micro-scale AD plant.

Parameter	**Value**
Gross energy production (kWh yr^{-1})	7744
LCOE (20 years, CHP electricity and heat, £ kWh^{-1})	0.209
LCOE (20 years, CHP electricity only, £ kWh^{-1})	0.313
LCOE (20 years, boiler heat only, £ kWh^{-1})	0.174
Simple payback time (years)	148

The household electricity supply price at the time of writing was £0.156 kWh^{-1} (Statista, 2020) and the household gas supply price was £0.0394 kWh^{-1} (GOV.UK, 2017b). The levelized cost of energy is therefore very high for a boiler-only installation, but only twice as expensive for a CHP-only installation. Considering the added advantages of the production of digestate and the provision of a local waste-processing resource, this might be a viable system for the in-house production of electricity.

6.6.12 Comparison with a large-scale AD plant

Published data (Banks *et al.*, 2011) allows a comparison of some of the performance outputs of micro-AD with large scale AD. The reference paper presents monitoring data of a 900 m^3 commercial anaerobic digestion system fed on food and green waste, with a monitoring period of 426 days. Values, either directly taken from or derived from the data presented in the paper, are shown and compared with equivalent values for the micro-AD site (table 6-13).

Table 6-13: Comparison of key performance indicators of large scale AD and micro-AD plants.

Performance parameter	**Large scale AD (Banks *et al.*, 2011)**	**Micro-AD**
Average specific biogas yield (m^3 $tonne^{-1}$ wet)	156	231
Average methane composition of biogas (%)	62.6	60.6
Average volumetric biogas yield (m^3_{biogas} $m^3_{digester}$ day^{-1})	1.59	1.58
Variation in weekly biogas production (+/- % of average)	32.8	61.6 (manual feed) 38.6 (auto feed)
Average parasitic electrical demand (% of elec. output)	31.4	31.6
Average parasitic heat demand (% of recoverable heat)	30.3	18.0
Digestate nitrogen (kg N $tonne^{-1}$)	5.6	4.7
Digestate phosphorus (kg P $tonne^{-1}$)	0.4	0.2
Digestate potassium (kg K $tonne^{-1}$)	2.3	2.3

Manual feeding was used for the first 192 days. On day 192, an automatic feeding system was installed between the pre-digester and the digester. The variation in weekly biogas flow was greater in the micro-AD system especially during the manual feeding period but was more comparable with the large-scale system once the automatic feeding was implemented.

Results for the volumetric biogas yield and biogas composition are broadly similar for both systems, demonstrating a similar level of performance in terms of biomethane output when compared with the size of the system. The average specific biogas yield from the feedstock was much lower in the large-scale system which could indicate a performance difference. However, in consideration of other available data from the large-scale plant, this can be attributed to a lower biogas potential of the feedstock due to addition of green waste and the feeding of less fresh food waste into the system. In comparison, the micro-AD digester was fed predominantly food waste and oats.

The parasitic requirement of the large-scale system (31.4%) is similar to that of the micro-AD system (31.6%) and the parasitic heat requirement is much greater in the large system, which can be attributed to the pasteurisation heat since no pasteurisation was performed at the micro-AD site.

From the data available, it appears that the performance of the micro-AD is either comparable or slightly better than the large-scale AD system. However, it is likely that the choice of appropriate scale would be made based on factors external to the system (e.g. collections, waste quantities and distribution of production, digestate use) or based on an economic analysis.

6.7 Conclusion

The novelty of this plant lies in its size and location, and from the results obtained and the long-term operation of the plant it can be concluded that it is a viable technology with the potential to help to solve the problem of food waste processing in the urban environment.

The operational performance parameters of the plant were very similar to a large-scale AD plant treating source segregated food waste in terms of main outputs and parasitic energy requirements. The plant processed 5.1 tonnes yr^{-1} of urban organic waste producing an average of 228 m^3 biogas per tonne of waste treated at average 60.6 % methane. The results showed that the plant was capable of stable operation despite large fluctuations in the rate and type of the feed waste biomass.

After initial signs of ammonia inhibition trace elements were supplemented to the system as per literature data and the biological system exhibited symptoms of recovery with a reduction in VFA concentration.

The system achieved a net positive energy balance and potential COP of 3.17 and 5.55 based on electrical and heat energy inputs and outputs respectively. Greenhouse gas emissions analysis concluded that the plant could result in carbon dioxide reduction 3885 kg yr^{-1}, which was equivalent to carbon reductions of 2.93 kg CO_2 kWh^{-1} electricity production or 0.762 kg CO_2 kg^{-1} waste treated.

7 Techno-economic analysis

7.1 Introduction

A techno-economic analysis (TEA) is a study in which a model of a system, in terms of its functional elements, inputs and outputs, is created to optimise the system with respect to a given functional unit. The purpose of the study is normally to support further development and improvement of the system (Zimmermann *et al.*, 2018).

In this TEA, the input was the feedstock for the anaerobic digestion (AD) process, the functional elements were the processes and equipment that are used to digest the feedstock, and the outputs are the digestate and biogas. A mass and energy-based model of the system was constructed and was used to quantify each part of the system.

As discussed in the literature review (section 2.2), one of the main reasons for a lack of implementation of anaerobic digestion (AD) at the micro scale (below 50kW in this study), is that the economics are seen as unfavourable (Yaman, Theaker and Walker, 2017), despite the evident enthusiasm for the technology at this scale. As AD can form an important part of a circular economy approach (The Ellen Macarthur Foundation, 2017), it is possible that one way of making micro-scale AD more economically favourable would be to investigate it as part of an integrated 'circular' system. This might add value to the outputs, or supply inputs at lower cost. The purpose of this TEA was therefore to construct a model that could be used to investigate an integrated system, and different processes within the system, to see if a more profitable process for micro-AD could be designed.

The starting point for the TEA was the set of core processes that are common to all anaerobic digestion systems (figure 7-1).

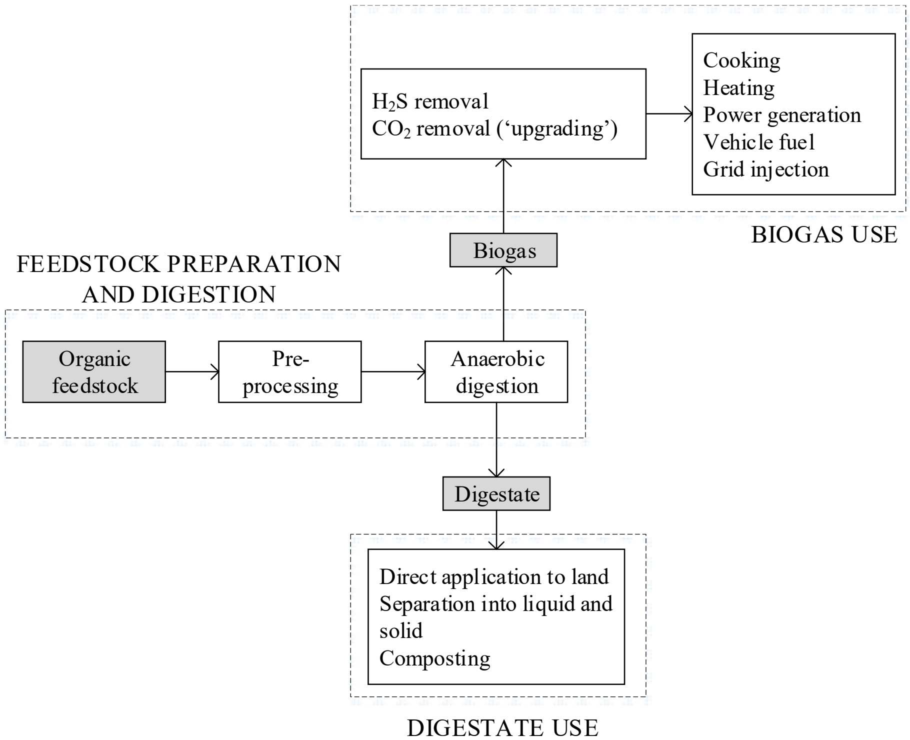


Figure 7-1: Generalised diagram of the anaerobic digestion process.

The amount of digestate produced by an AD plant can be large - estimated to be 82% to 87% by weight of the input feedstock (Turley *et al.*, 2016), and due to its scale, the treatment, transport and disposal costs are an important consideration in terms of the economics of the project. It was therefore included in the TEA study.

7.2 Methodology

The working method that was used for this TEA (figure 7-2) was derived from Zimmermann *et al*. (2018). Reference was also made to other TEA and life-cycle analysis examples (Patterson *et al.*, 2011; Sanscartier, MacLean and Saville, 2012; Diego, Bellas and Pourkashanian, 2018; Khan *et al.*, 2014).

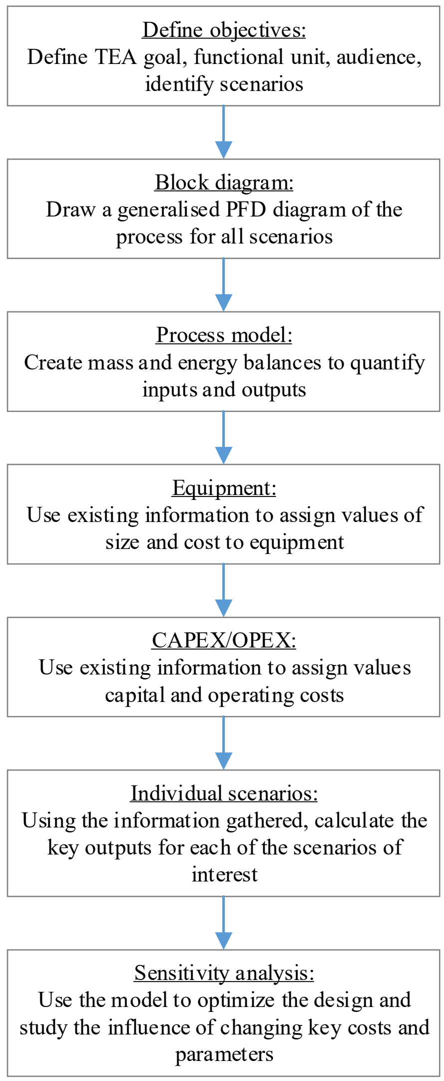


Figure 7-2: Process flow diagram for this TEA study, adapted from (2018).

7.3 Objectives

The goal of the TEA was to optimise the design of a micro-AD system in terms of its economic performance. The functional unit chosen to compare between scenarios was the simple payback time, in years, as it was a commonly used functional unit that incorporated the capital cost and revenue of the whole system. The levelized cost of energy (LCOE, in £ kWh^{-1}) was also analysed and used to compare different instances of a scenario. It was noted that the various forms of energy produced in the scenarios (heat, electricity, vehicle fuel) had different values, both economically and energetically. The usefulness of the LCOE as a comparison tool was therefore limited, and the simple payback time was preferred, as it could take into account this difference in value of energy forms. British pounds was used as the currency, because the setting was the UK and included UK-specific revenues such as the Renewable Heat Incentive (RHI).

The location of the system was envisaged as urban or peri-urban, with access to good transport links and utilities. The audience for the TEA was expected to be national and local government policy makers, academics, industrial engineers, funding agencies and investors. The TEA was written from the perspective of a person planning and designing a new micro-scale AD plant and connected systems.

7.4 Scope

There are a number of key features that determine the viability of an anaerobic digestion facility in a specific scenario. For example, a consistent supply of good quality feedstock, the availability of local expertise to manage the plant, the existence of outlets for the digestate, sufficient space, and the availability of capital funds. The scenario chosen as a focus for the study was a university. This represented an establishment in which there was continuous occupation of the facility, therefore a constant supply of food waste and other organic wastes, with some seasonal variation, and a constant heat and electricity demand. It should be noted that during university holidays, the campus was occupied by staff and visiting students, and so although there was a reduction in activity, the feedstock supply continued, albeit at a lower level. The university scenario had an additional advantage that there was accurate historical data available for use and the plant, when built, would provide a teaching and learning opportunity. A number of the key features were noted (table 7-1).

Table 7-1: Features of the university scenario that was used in the TEA.

Feature	University
Size	Large
Feedstock	Food waste and green waste
Attendance	Continual, variable
Heat demand?	Yes
Digestate usage?	Limited
Location	Outbuilding
Management	Estates management, research staff, students
Learning opportunity	Yes

The scope of a TEA normally defines the system's boundaries in terms of what elements are included. However, the purpose of this TEA was to investigate how the functional unit was affected by moving the system's boundaries. Therefore, the system diagram (figure 7-3) includes optional elements, which are marked in dashed lines. This system diagram was used as a basis to develop the inventory.

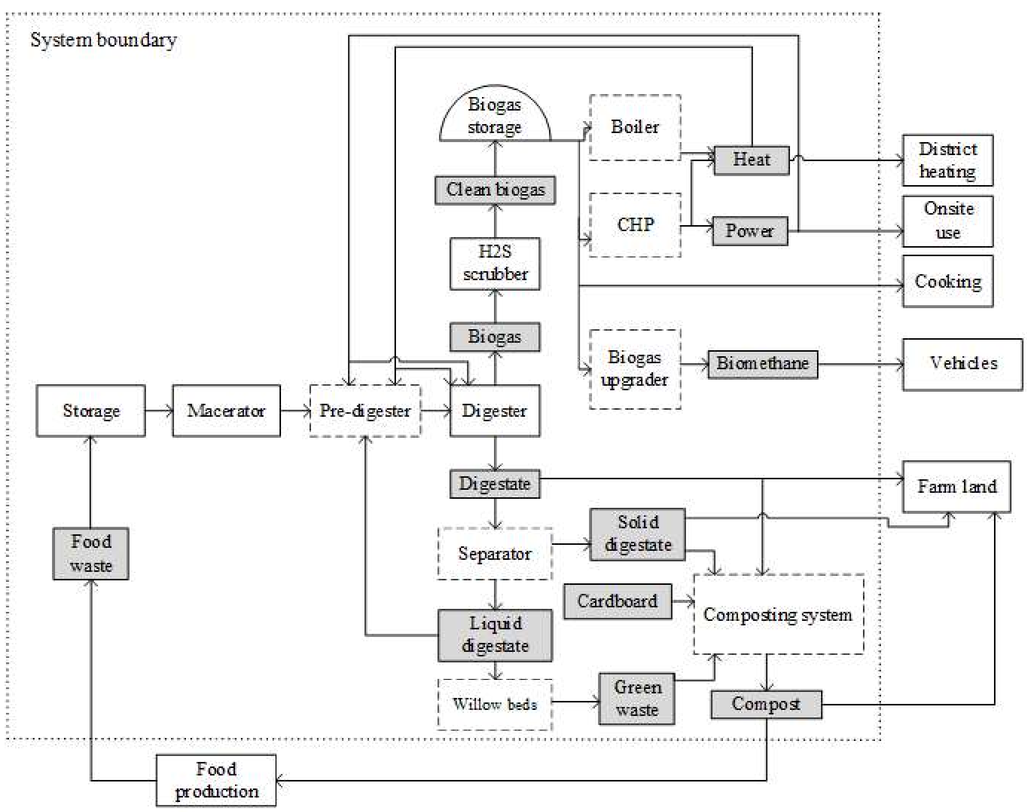


Figure 7-3: General diagram of a micro-AD plant. Optional elements are marked in dashed lines, processes/equipment are shown in white boxes and inputs and outputs are shown in greyed boxes.

7.4.1 Scenarios

The scenarios considered for the study were defined in terms of the equipment that was included in the process. Initially, nine scenarios were considered.

Table 7-2: Initial scenarios for the TEA analysis.

	Pre-digester	Boiler	CHP	Biogas upgrader	Separator	Composting whole digestate	Composting separated digestate
1a							
1b							
1c							
2a							
2b							
2c							
3a							
3b							
3c							

Each scenario was defined in terms of the equipment and processes it includes. Note that if a separator was included in the scenario, a pre-digester was also necessary, to ensure that all parts of the separated digestate had either undergone a pasteurization process or a composting process, to eliminate pathogens. For this reason, there were no scenarios considered that had a separator but no pre-digester.

These were the scenarios studied initially; more scenarios were added during the progress of the TEA to further explore promising areas of interest.

Scenario 1

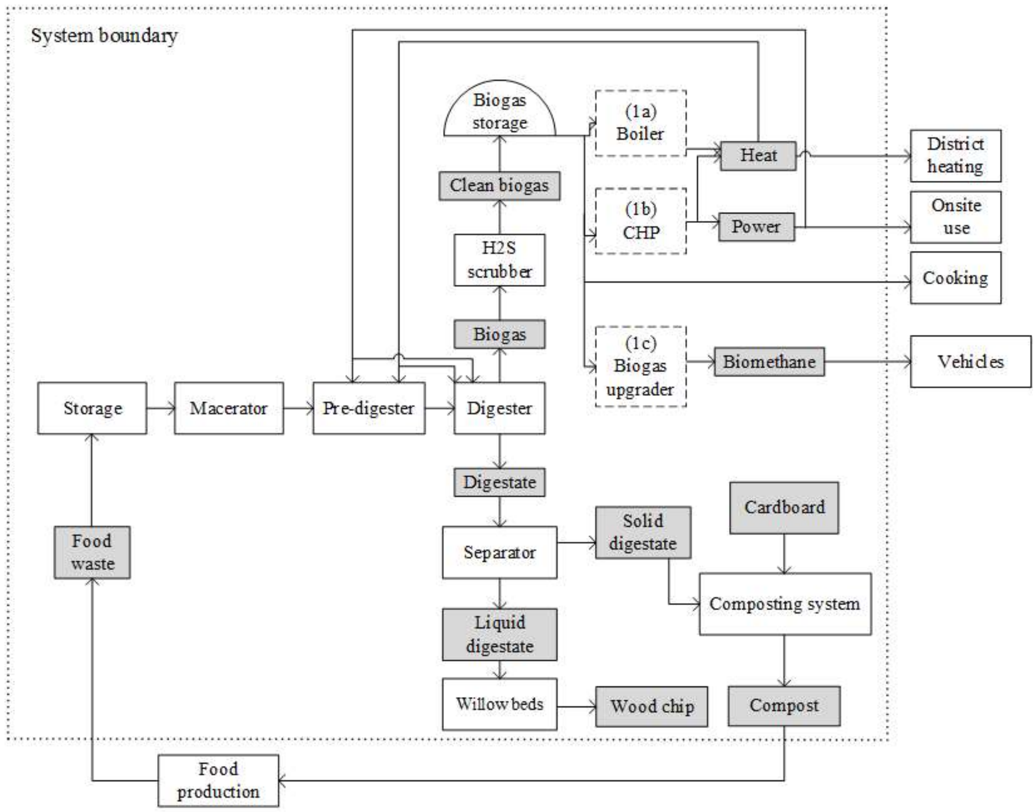


Figure 7-4: Schematic of scenario 1 with biogas use options shown in dashed lines.

In scenario 1 (figure 7-4), the feedstock was fed through a macerator into a pre-digester, where it was heated to 63°C for 7 days (on average) which will pasteurize the feedstock. After the digester, the digestate was separated and composted with cardboard (solid fraction) or sent to a willow bed (liquid fraction). The liquid fraction was converted by the willow bed into clean water (which is released through transpiration) and organic matter in the growing willow. Three instances of the scenario were considered: 1a, with a boiler, 1b, with a CHP, and 1c, with a biogas upgrader.

Scenario 2

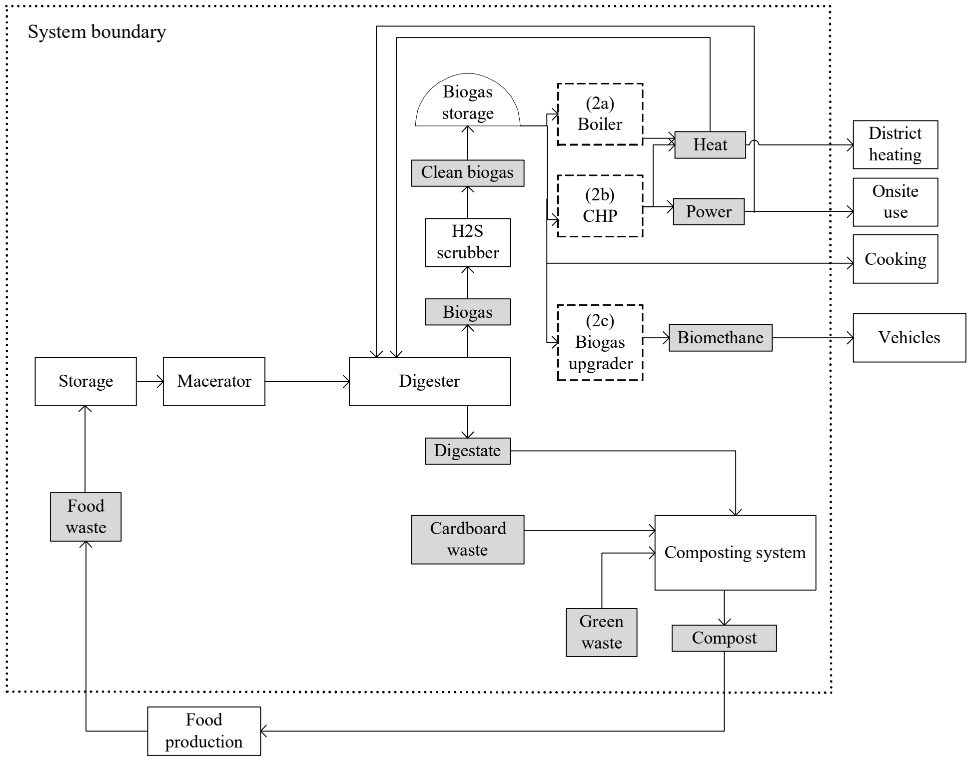


Figure 7-5: Schematic of scenario 2 with biogas use options shown in dashed lines.

In scenario 2 (figure 7-5), the pre-digester stage was not included and the digestate was composted whole rather than being separated. This was to ensure that the animal by-product regulations are met by all products of the system by going through a heating stage (the compost system was expected to reach 65°C)(Irvine, Lamont and Antizar-Ladislao, 2010). Three instances of the scenario were considered: 2a, with a boiler, 2b, with a CHP, and 2c, with a biogas upgrader.

Scenario 3

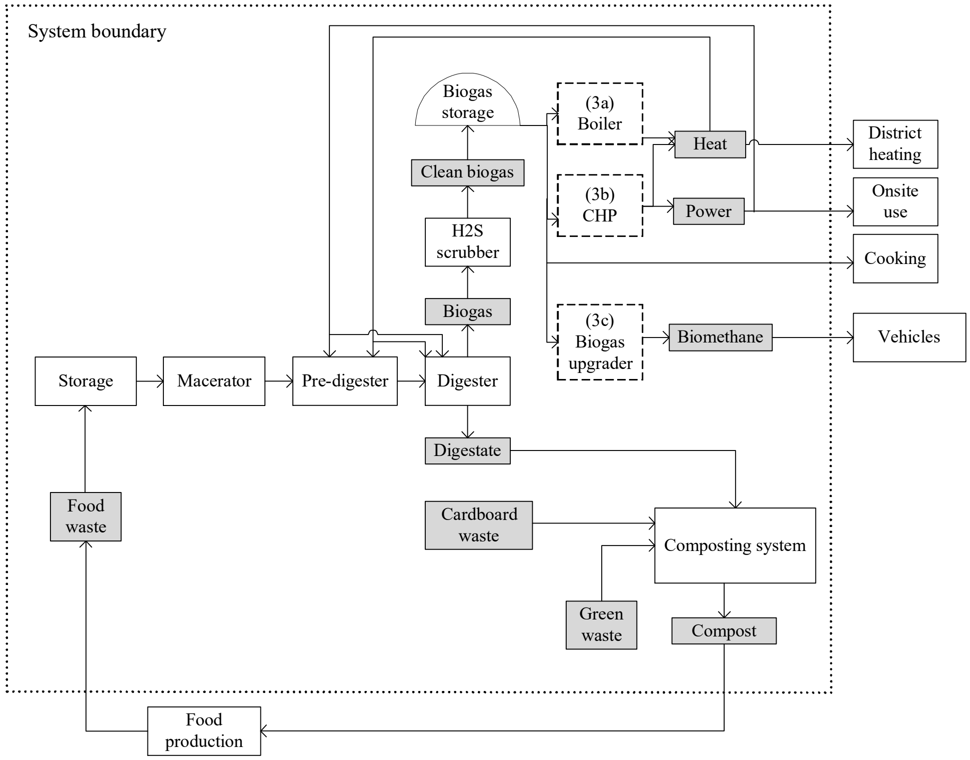


Figure 7-6: Schematic of scenario 3 with biogas use options shown in dashed lines.

In scenario 3 (figure 7-6), the pre-digester stage was included and the whole digestate was composted. This was done to investigate the difference between scenarios 2 and 3 (without and with a pre-digester). Again, three instances of the scenario were considered: 3a, with a boiler, 3b, with a CHP, and 3c, with a biogas upgrader.

7.5 Inventory: feedstock preparation and digestion

The inventory is a listing and calculation of the inputs and outputs of the system given the scope. All parts of the system, both standard and optional, were assessed.

7.5.1 Feedstocks

The feedstocks for the AD process were food waste and vegetable oil. Green/garden waste was not included as an input to AD as it would be likely to contain ligno-cellulosic material, which is difficult to digest (Fan *et al.*, 2019). However, in the scenarios where the impact of combined AD and composting was considered, this waste stream was used as a co-composting substrate. Cardboard was included as a composting input to add carbon-rich material. The amounts of each feedstock were derived from real data provided by the University of Sheffield (table 7-3).

Table 7-3: Feedstock masses (in tonnes per year) used in the TEA.

	Units	**University (3-year average)**
Population	people	36,000
Food waste	tonnes yr^{-1}	119.0 ±13%
Vegetable oil waste	tonnes yr^{-1}	6.0 ± 20%
Green waste	tonnes yr^{-1}	90.6 ± 13%
Cardboard waste	tonnes yr^{-1}	42.8 ± 13%

This amount of food waste represents 3.3 kg $person^{-1}$ yr^{-1}. The average amount of food waste produced per person per day in Europe is 173 kg $person^{-1}$ $year^{-1}$ (Banks, 2018). The estimated food waste figure for Europe includes all post-farm food waste (production losses, spoiled food, by-products or co-products, trimmings and scraps, leftovers). However, even taking this into account it is likely that the food waste captured at the University could be increased significantly.

7.5.2 Mass balance

A mass balance was adapted from the one previously created for the analysis of the laboratory work (Appendix A) to quantify the inputs and outputs for the TEA model. The adapted model added vegetable oil as a feedstock, and also added calculations of the carbon and nitrogen content of the digestate so this information could be used to model the composition of the compost. A ‘separator’ section was also added, to provide estimations of the amounts of post-separation solid and liquid fractions, and the solids content of each, which were used to model the compost and willow bed processes.

Food waste	
3.7	gVS/L/day OLR
119.0	tonnes/yr
325.9	kg/day
24%	TS
22%	VS
1.00	kg/L food waste density
118.95	m3/yr volume in
325.9	L/day volume in
26.2	T/yr VS input
2.4	T/yr ash input
90.4	T/yr water input
471	m3 CH4/tonne VS BMP
54.6%	% CH4
90%	% VS removal

Vegetable oil	
6.0	tonnes/yr
17.9	L/day volume
100%	VS
5%	% by weight of feedstock
16.4	kg/day
100%	% removal

Inside digester	
24,062	L digester capac.
70.0	days retention time
88.1	kg/day VS

Discharge (digestate)	
2.9	VS out T/yr
2.4	Ash out T/yr
80.8	Water out T/yr
87.2	Total mass out T/yr
6.1%	TS in digestate
3.6%	VS in digestate
2.7%	Ash in digestate
12.475	g/L NH3 concentration
1.0	Digestate density kg/L
88.1	Digestate out m3/yr
1.307	T/yr Digestate C content
0.995	T/yr Digestate N content
1.50%	% C content
1.14%	% N content

Separator	
20%	Solids fraction (by weight)
25%	Solid fraction dry matter
5.29	T/yr solids out
17.44	T/yr Solid fraction weight
4.4	T/yr Solid fraction DM
13.1	T/yr Solid fraction water
69.8	T/yr liquid fraction weight
67.7	T/yr Liquids fraction water
0.9	T/yr Liquids fraction solids
1.4%	Liquids DM

BUSWELL CALCULATIONS	
Food waste	
0.2481	g water consumption/gVS consumed
0.168	g water consumption/g biogas produced
0.3609	g CH4 production/gVS consumed
0.8389	g CO2 production/gVS consumed
0.0021	g H2S production/gVS consumed
0.046	g Ammonia production/gVS consumed
0.832	gVS consumed/g biogas produced
54.6%	% methane in biogas
Vegetable oil	
0.5752	g water consumption/gVS consumed
1.57	g biogas output/g vegetable oil
0.365	g water consumption/g biogas produced
0.7033	g CH4 production/gVS consumed
0.8708	g CO2 production/gVS consumed
0.0002	g H2S production/gVS consumed
0.000	g Ammonia production/gVS consumed
0.635	gVS consumed/g biogas produced
69.0%	% methane in biogas

Energy - food waste	
12006.8	m3/yr methane from food waste (Buswell)
1.272	kg/m3 biogas density
21976.9	m3/yr biogas
60.2	m3/day biogas
2.51	m3/hr biogas
32.9	m3/day methane
1,486	ppm H2S concentration
1.086	T/yr NH3 production
1.033	T/yr lost as water vapour
28.0	T/yr lost as biogas
23.3	T/yr VS consumed to make biogas
4.7	T/yr water consumed to make biogas
5.43	Biogas CV KWh/m3
Energy - vegetable oil	
5,938.1	m3/yr methane from veg oil
1.09	kg/m3 biogas density
8,611.4	m3/yr biogas output from veg oil
23.6	m3/day biogas
0.98	m3/hr biogas flow
16.3	m3/day methane
98	ppm H2S concentration
0.002	T/yr NH3 production
0.0174	g/L NH3 concentration
3.4	T/yr water consumption
0.405	T/yr lost as water vapour

Energy - combined	
17945	m3/yr methane
49.2	m3/day methane
30588	m3/yr biogas
58.7%	% methane in biogas
1095	ppm H2S concentration
5.83	kWh/m3 biogas CV
45.0%	Engine thermal efficiency
30.0%	Engine electrical efficiency
92.0%	Engine availability
5.622	Electricity production kW
8.434	Electricity production kW

Figure 7-7: Mass balance for the TEA, with food waste and vegetable oil feedstocks.

The solid fraction weight and dry matter after separation was estimated using data from a digestate processing review (figure 7-8).

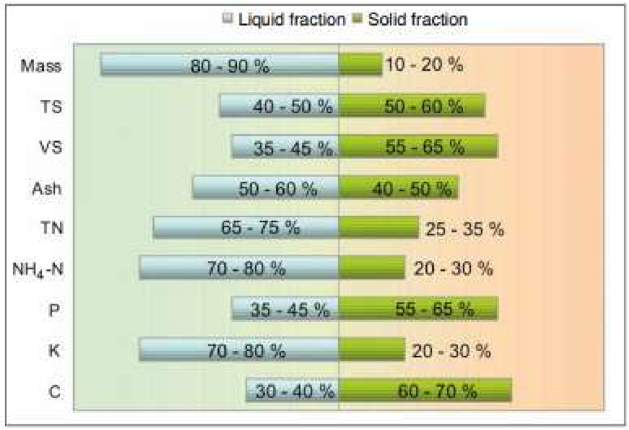


Figure 7-8: Distribution of the principal constituents after solid–liquid separation (Drosg et al., 2015).

Vegetable oil was included as a feedstock, and the Buswell equation (Buswell and Mueller, 1952) was used with a published compositional analysis of vegetable oil (San Jose, Arroyo and Sanz-Tejedor, 2019) to determine the expected biogas production, water consumption and biogas composition from its digestion (table 7-4).

Table 7-4: Calculation of the sample molar ratio for vegetable oil.

Element	N	C	S	H	O
% of sample (by weight) (San Jose, Arroyo and Sanz-Tejedor, 2019)	0.03	76.5	0.02	11.2	12.2
Molecular weight	14	12	32	1	16
Molar ratio (moles in 1g sample)	$2.143\text{x}10^{-5}$	0.0638	$0.625\text{x}10^{-5}$	0.1120	0.00763

The stoichiometry of the decomposition of vegetable oil was calculated (table 7-5).

Table 7-5: Stoichiometry for vegetable oil decomposition using the Buswell equation.

		Biomass sample	**H_2O consumed**	**CO_2 produced**	**CH_4 produced**	**Ammonia produced**	**H_2S produced**
Molar ratio	Mol	-	0.0320	0.0197	0.0440	$2.14\text{x}10^{-5}$	$0.625\text{x}10^{-5}$
Mass	g	1	0.575	0.871	0.703	0.000364	0.000213
Volume (gases)	L	-	-	0.422	0.424		0.0035

This resulted in a predicted biogas composition of 69.0 % methane, 98 ppm H_2S and a mass of 1.574 g of biogas for every gram of vegetable oil digested.

The addition of fats, oils and greases (FOGs) has been shown to increase the production of biogas by 30-80% (Long *et al.*, 2012; Amha *et al.*, 2017), and so when co-digesting with vegetable oil, the food waste % VS removal was increased to 90%. Some inhibition of the process has been reported when FOGs comprise above 30% of the feedstock by weight, so the proportion of oil was kept below this level (Meng *et al.*, 2015; Amha *et al.*, 2017). The minimum recommended hydraulic retention time was 30 days – it was noted that the digestion time for fats is longer than average (Meng *et al.*, 2015) – but in order to keep the organic loading rate below the highest recommended loading rate, 3.5 gVS L^{-1} day^{-1} (Gerardi, 2003b), the hydraulic retention time was 74 days.

The mass balance produced a profile of the scenario (table 7-6).

Table 7-6: Results of the mass balance using 100% of the estimated food waste and vegetable oil for the University of Sheffield.

Value	Unit	
Organic loading rate	gVS L^{-1} day^{-1}	3.5
Hydraulic retention time	days	74
Digester working capacity	m^3	25.4
Biogas production from food waste	m^3 day^{-1}	60.2
Biogas production from vegetable oil	m^3 day^{-1}	23.6
Methane concentration of biogas	%	58.7
Methane production	m^3 day^{-1}	49.2
Hydrogen sulphide concentration	ppm	1095
CHP electricity rating	kW	5.6
CHP heat rating	kW	8.4
Boiler heat rating	kW	16.9
Digestate total solids (TS)	% of whole by mass	6.1
Digestate volatile solids (TS)	% of whole by mass	3.6
Digestate ammonia concentration	g L^{-1}	12.475

These data were used as the inputs to the TEA model. The predicted ammonia concentration is very high and in a working plant it might be lower (due to ammonia evaporation, and ammonia uptake for microorganism growth), otherwise it would need to be mitigated to avoid toxicity.

7.5.3 Feedstock storage

The feedstock storage was envisaged as a concrete container made on-site, with drainage and a lid fitted, which would contain the food waste only. A separate storage container would be needed for the vegetable oil. The food waste storage would need to be sealed to reduce odours at ground-level but vented to a higher level (3-4 metres) to prevent gas build-up. Drainage into a sump outside the storage container was required to allow leachate from the feedstock to be collected. The storage would therefore need to be constructed bespoke to the scenario, according to the size calculated. A storage time of 14 days was estimated, as a minimum weekly collection was expected, with a relatively steady monthly food waste production amount over the course of a year. This was predicted by studying the food waste collections for the University of Sheffield Union over 3 years, which does not show any discernible pattern of variance or predictable periods of high or low output (figure 7-9). The Union food waste made up approximately half of the total food waste collection, so could be used as a predictor of the variation in food waste production overall.

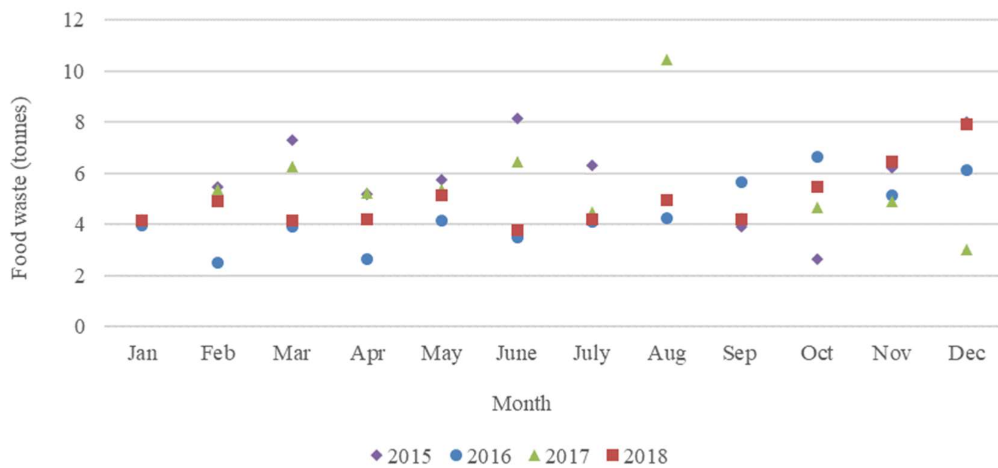


Figure 7-9: Monthly food waste amounts in tonnes collected from the University of Sheffield Union building, 2015 to 2018.

7.5.4 Macerator

Food waste must be macerated before being added to an anaerobic digester, to reduce the particle size and hence to prevent blockages and improve the digestion process. The equipment must be handled by an operator to prevent the ingress of unwanted items such as cutlery and packaging. As manual handling is required, the operational time must be limited to prevent risks such as repetitive strain injury, back problems and accidents caused by tiredness. The operational time was assumed to be half an hour per day, which was estimated as a reasonably short time period to perform the task, and this was used as a basis to calculate the size of macerator required. The type of macerator that would be suitable for this task was an industrial meat grinder such as a Quattro MG32SS Heavy Duty Meat Grinder.

7.5.5 Pre-digester and digester

A pre-digester is an extra tank that contains the feedstock before it is fed into the digester. It can be mixed and heated, and is sometimes referred to as a hydrolysis tank, as the feedstock will start to hydrolyse under these conditions. A pre-digestion tank is often included in food waste anaerobic digestion processes as it can be used to pasteurise the feed input, which would be necessary (EC-European Commission, 2003). To achieve pasteurisation if there was no composting stage (i.e. the digestate was sent straight to land, either whole or separated), the contents of the pre-digester need to be kept at over 63°C for at least 30 minutes (Engineering Toolbox, 2010; Thwaites *et al.*, 2015). The pre-digester can also be used to pre-heat and mix the feedstock, and provide additional storage volume in a controlled environment, and to balance out any fluctuations in the feedstock supply (Walker *et al.*, 2017). The pre-digester was envisaged as an insulated cylindrical stainless steel stirred tank with a water jacket or heating coil. The size of the tank was calculated from the food waste and vegetable oil input masses and the expected residence time in the tank and was used to calculate the expected heating loss and heating requirement of the tank. The expected residence time in the tank was estimated at 7 days, which would provide more than the required length of time for pasteurization.

The digester was envisaged as an insulated, continuously-stirred tank reactor (CSTR) which was heated and stirred, with a diaphragm pump at the inlet and outlet. If there was a pre-digestion stage, the heat requirement of the digester would reduce as the inlet feedstock would be at 63°C or just below. The heat requirement of the digester was calculated by assuming a temperature of 37°C, and calculating the heat input from the pre-digester and the heat loss by convection from the digester (Coulson and Richardson, 1999).

Due to the small size of the system, the heated vessels (the digester and pre-digester) could be housed in a protected atmosphere (for example, a shed or greenhouse) for a relatively small cost, which would reduce the heating cost. This approach was used in a London-based micro-scale digester, with the result that it produced a 49% saving in the heat requirement of the digester (Walker *et al.*, 2017). As the digester location will be comparable (in the UK), is has been assumed that adding a housing around the digester could save the same amount in heat requirement.

The heat requirements over both tanks were calculated with and without a pre-digester, and with and without a greenhouse (table 7-7). The calculations and data assumptions for these figures are provided in Appendix B.

Table 7-7: Results of the pre-digester design calculations.

	Unit	Value
Pre-digester residence time	days	7
Volume of pre-digester required	m^3	2.7
Temperature of pre-digester	°C	63
Volume of digester required	m^3	28.0
Temperature of digester	°C	37
Heat requirement without greenhouse, with pre-digester	kWh day^{-1}	52.7 (10.8%)
Heat requirement without greenhouse, without predigester (digester only)	kWh day^{-1}	20.1 (4.1 %)
Heat requirement with greenhouse and predigester	kWh day^{-1}	25.8 (5.3 %)
Heat requirement with greenhouse, without predigester.	kWh day^{-1}	9.9 (2.0%)

The parasitic heat requirements are shown as a proportion of the gross energy output from the digester (table 7-7, shown in brackets). These are commonly calculated for AD plants as they can be a key factor in its profitability – parasitic loads can be up to 29% of the gross energy output (Banks, 2018). The parasitic heat and electricity in the model seem small compared to this figure, so may be higher in practice. The parasitic heat demand was increased with the addition of a pre-digester, but this extra energy use may be necessary to ensure that legal health and safety requirements are met.

7.6 Inventory: biogas use

The potential uses of the biogas appropriate for this size of plant were considered to be boiler, micro-CHP, and upgrading for vehicle fuel. Biogas storage was included in the calculations for the TEA, with an estimated storage volume equivalent to half a day of biogas production. A flare was also included in the system to allow for the flaring of biogas in the case in which biogas production exceeds usage capacity or in emergencies.

7.6.1 Boiler

A boiler would provide a reliable, high-efficiency (typically 90%) outlet for the biogas so that it could be combusted as it is produced, with the heat energy that is produced being used to heat an accumulator tank. A predictable heat demand with a reasonably steady load (for example, average ±30% over a week) would be required on the site, for example a district heat network, washing facilities or a drying room. If the demand for heat greatly exceeded the supply, this could be regarded as a reliable outlet. Gas boiler technology is well established, but the boiler would need to be modified for biogas, which would make it more expensive.

7.6.2 Micro CHP (combined heat and power) engine

To run efficiently, a CHP needs to run continuously as close to its rated output as possible, and so benefits from a steady load. The availability of appropriately sized technology affected whether this outlet could be used and there has been recent growth in interest in micro-CHP for natural gas on a domestic scale. Three types of CHP were commercially available: Sterling engine, internal combustion engine and fuel cell. An example of a micro-CHP was a fuel cell based unit rated at 0.75kW with a 37% electrical efficiency (The Renewable Energy Hub UK, 2018). A 7.5kW gas-fired CHP was available commercially (Helec, 2020), but this would need to be adapted to biogas, and a 10 kW biogas CHP was available in the UK market (direct communication, source confidential). The technology is developing and may increase in range, efficiency and availability in the future.

7.6.3 Biogas upgrading

Biogas (approximately 60% methane) can be ‘upgraded’ to biomethane (>95% methane) by removing the water vapour, carbon dioxide and hydrogen sulphide, which would make it useable as a vehicle fuel or suitable for grid injection. There are a number of different upgrading technologies, with different process efficiencies, only some of which are scalable to ‘micro’ level. The most suited to small scale, low-technology applications would be water scrubbing, which is a well-researched technique that can produce biogas containing >97% methane (Petersson and Wellinger, 2009). This method also carried the advantages that it used innocuous and cheap substrates (as opposed to a strong alkali), did not produce substances that are difficult to dispose of, and has low electricity demand. Another micro-scale solution would be membrane filtration, however the availability of this technology was one which could be operated at 50 m^3 biogas hr^{-1} at the lowest possible flow (Prodeval, 2020), and so not suitable for this study. According to research, installations over 5m^3 biogas hr^{-1} could justify the addition of a biogas upgrading system, the smallest of which have a capacity of 10m^3 biogas hr^{-1} (Chesshire and Llewellyn, 2012; Metener, 2020).

Upgraded biogas can be used as a transport fuel by especially adapted vehicles and if used within the organisation could replace diesel. It can also be sold as a transport fuel to other biomethane users and attracts Renewable Transport Fuel Certificates (RTFCs), which are price indexed and can be sold to companies that need to fulfil their renewable fuel obligations (GOV.UK, 2020b). If the biogas was produced and used in a vehicle on site it would also produce a saving in fuel costs. This system is practicable on a small scale and so the output (in kg yr^{-1}) and fuel miles were calculated.

7.6.4 Summary of the biogas use options

Using the information collected, a summary of the different options was compiled to be used to calculate revenue.

Table 7-8: Summary of biogas use options for the university scenario.

	Unit	Value	Source
Biogas production	$m^3\ yr^{-1}$	30588	Figure 7-7, 'Energy – combined'
Biogas methane concentration	%	58.7%	Figure 7-7, 'Energy – combined'
Methane production	$m^3\ yr^{-1}$	17945	Figure 7-7, 'Energy – combined'
Gross energy production rate	kW	20.4	Methane production, CV of methane
Boiler heat production rate	kW	18.3	Calculated from gross energy production, using 90% boiler efficiency
CHP heat production rate	kW	8.4	Figure 7-7, 'Energy – combined'
CHP electricity production rate	kW	5.6	Figure 7-7, 'Energy – combined'
Biogas upgrading biomethane production	$kg\ yr^{-1}$	12583	Calculated from gross methane production using 97% final methane content of biomethane
Biomethane fuel miles equivalent	$miles\ yr^{-1}$	125833	Calculated using biomethane production x 10 miles kg^{-1} (estimate from CNG professional, personal communication)

It is essential that all heat, electricity or biomethane that is produced is made use of in order to maximise the benefit from the AD plant. For example, the boiler heat production would provide 6.3 m^3 of hot water per day (calculated from the amount of energy required to raise a quantity of water from 10°C to 70°C and the boiler's rated energy output), which would be easily used. The biomethane fuel production is equivalent to over 125,000 miles. The use of biogas as a vehicle fuel would require investment in buying or converting several vehicles (assuming each vehicle travels on average about 20,000 miles per year), adding to the CAPEX requirement. Otherwise, the fuel could be sold to other biomethane users, but this would introduce a new 'business' into the organisation which would add complexity and may not be practicable.

7.6.5 Hydrogen sulphide removal

It is necessary to remove the hydrogen sulphide from biogas for safety and to avoid corrosion of the pipework and other equipment (Okoro and Sun, 2019). The amount of hydrogen sulphide in the biogas depends on the feedstock used and can be up to 10000 ppm or 1% (Zulkefli *et al.*, 2016). The level of hydrogen sulphide in the biogas as calculated by the mass balance was 1026 ppm.

There is a range of hydrogen sulphide removal options available, for example using biofilters or activated carbon (adsorption), micro-dosing of air/oxygen or adding chemicals (Okoro and Sun, 2019). The cheapest option of these is adding chemicals, such as Iron (III) hydroxide, to the digester or the pre-digester to precipitate sulphides. Due to the small volumes involved and the very low initial capital input, this was chosen as the hydrogen sulphide removal method. The cost is approximately £0.0077 m^{-3} biogas (Okoro and Sun, 2019), however as the same amount of biogas was produced in all variations of a scenario (because the same amount of feedstocks were used), it was assumed to be included in the OPEX costs, which were estimated as a percentage of the CAPEX and therefore increased and decreased relative to the size of the plant and the amount of biogas produced.

7.7 Inventory: digestate use

7.7.1 Digestate separation

The use or disposal of digestate produced from anaerobic digestion is a vital issue, as it contains important nutrients (nitrogen, potassium, phosphorus and fibre) and therefore can be valuable, but can also be difficult to dispose of as it is produced in high volume (approximately 80% of the input feedstock) as a slurry, and therefore difficult to store and transport. This issue is particularly key in urban areas because there is less likely to be a ready receiver of the digestate – for example, large areas of land onto which the digestate can be spread (Fuldauer *et al.*, 2018). There are also limitations as to when the digestate can be spread on land in nitrate vulnerable zones. However, if the digestate can be transformed into a useful product, it could be marketed and sold, which would improve the profitability of the process.

Digestate can be mechanically separated into two waste streams; a ‘solid’ fraction and a ‘liquid’ fraction. The ratio of the two fractions and the resulting solids content of each depends on the total solids content of the digestate and the method used (table 7-9).

Table 7-9: Favoured separation type for digestate at a range of TS (total solids, in %) and the proportions and solids content of the separated fractions (derived from (Drosg et al., 2015)).

Separation type	Digestate TS %	Solid fraction (by weight)	Liquid fraction (by weight)	Solid fraction TS	Liquid fraction TS
Screw press	High (9-12%)	40%	60%	35%	calculated
	Medium (5-9%)	25%	75%	30%	calculated
	Low (1-5%)	10%	90%	25%	calculated
Decanter centrifuge	Low (1-5%)			28%	2.3%

From the mass balance, the predicted dry matter content of the digestate was around 5% solids, which meant that the most appropriate separation technology was a screw press, and (from table 7-9) that the ratio of solid to liquid fraction by weight would be roughly 20:80, with a solid fraction TS of 25%.

Table 7-10: Results of digestate separation from the mass balance for the university scenario.

	Unit	Value
Digestate production rate	T yr^{-1}	87.2
Liquid fraction production rate	T yr^{-1}	69.8
Liquid fraction total solids content	%	1.4
Solid fraction production rate	T yr^{-1}	17.4
Solid fraction total solids content	%	25

The separation produced a solid fraction that could be directly applied to land or composted (WRAP, 2012a), and a liquid fraction of a low enough solids content that it could be used to feed a willow bed (Labrecque, Teodorescu and Daigle, 1997), recirculated to add liquid to the front end of the AD process, or used in hydroponics (Ronga *et al.*, 2019). No liquid input (i.e. fresh water) was needed, and hydroponics was out of scope of the study, so the willow bed option was investigated further.

7.7.2 Willow/reed bed

The liquid fraction of separated digestate could be passed through a willow bed (figure 7-10) to produce water that is clean enough to dispose of through standard drainage or reused.

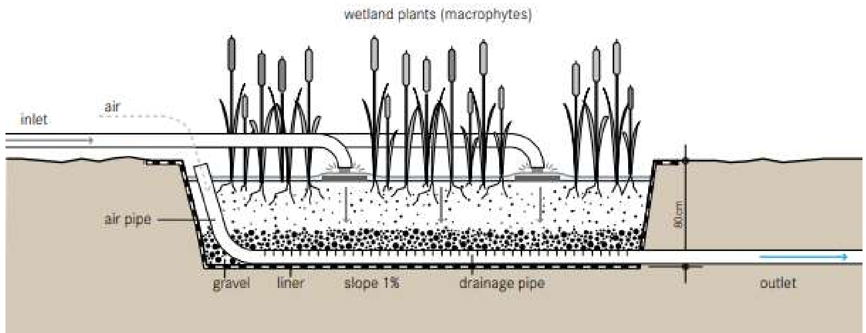


Figure 7-10: Side view of a vertical-flow constructed wetland for water purification (Tilley, 2014).

There is a limit to the amount of nitrogen that can be absorbed by a willow bed; this was assumed to be 300 kgN Ha^{-1} yr^{-1} (Labrecque, Teodorescu and Daigle, 1997). The footprint of the willow bed was calculated (table 7-11).

Table 7-11: Results of willow bed size and output calculations from the mass balance.

	Unit	**University**
Willow bed square footprint	hectares	11.3
Willow chips production	T yr^{-1}	11.4
Willow chips production volume	m^3 yr^{-1}	37.7

The willow bed would have a large footprint, therefore would only be viable there was sufficient land available. It would also require management and would therefore have a relatively high running cost. The wood chippings produced could be sold or composted, used on the gardens as a mulch, or burned as a biofuel, which may attract RHIs. The willow bed effluent would be a small amount of clean water, and so could be drained into a soakaway, or would evaporate (Caslin *et al.*, 2015).

7.7.3 Composting

Composting of organic waste was considered in this study as a method of processing the digestate output from the AD process. The composting process incorporates the added advantages of producing a stable soil-conditioning material with a higher carbon-nitrogen ratio than digestate that is better for the soil, producing heat for hot water production, and satisfying animal by-product regulations when managed properly (section 2.3.8).

The ideal moisture in a compost heap is 45-60% (solids content 40-55%). If the compost is too moist, water will take up air spaces, reducing oxygen availability. However, if the compost is too dry, the composting microorganisms will not be able to function properly (Cooperband, 2000; The Compost Gardener, 2019). The optimal C:N ratio is 25:1 to 35:1 (Cooperband, 2000). To ensure the compost process was viable, the moisture content and C:N ratio were calculated for each combination of feedstocks.

Initially, two composting alternatives were considered; composting whole digestate, or composting the solid fraction of separated digestate. Whole digestate has a high moisture content (85-95%) and a low carbon-nitrogen ratio (from the mass balance, about 1.3:1). When separated, the solid fraction of digestate has a lower moisture content (about 75%) and a slightly higher C:N ratio (table 7-12). Both substances therefore needed to be co-composted with other materials to achieve the ideal C:N ratio and moisture content for composting.

The co-composting substrates available were garden waste (shredded/chipped) and shredded cardboard. Willow coppicing waste (shredded/chipped) was also available if a willow bed was included in the system. These waste streams were readily available and had properties that were complimentary to digestate in composting terms (table 7-12). Composting digestate with wood chips (similar to shredded coppiced wood) has been shown to reduce the emission of nitrogen compounds such as ammonia (NH_3), nitrite (NO_2^-) and nitrate (NO_3^-) and help to stabilize digestate by slowing down its degradation (Zeng, De Guardia and Dabert, 2016).

Table 7-12: C:N ratio and moisture content values for the composting substrates used in the TEA, from references.

Property	**Substance**	**Value**	**Source**
C:N ratio	Digestate	1.3	Mass balance
	Solid fraction of digestate	3.4	Mass balance
	Coppiced willow	167	(Whittaker *et al.*, 2018)
	Cardboard	183	(Capson-Tojo *et al.*, 2017)
	Garden waste	55	(Boldrin and Christensen, 2010)
Moisture content	Digestate	94%	Mass balance
	Solid fraction of digestate	25%	Mass balance
	Coppiced willow	56.4%	(Whittaker *et al.*, 2018)
	Cardboard	7%	(Capson-Tojo *et al.*, 2017)
	Garden waste	60%	(Boldrin and Christensen, 2010)

The solid fraction C:N ratio was derived from the carbon and nitrogen content of the digestate and the constituent distribution supplied in a digestate processing review (figure 7-8). Composting is an exothermic reaction, and the heat that it creates can be captured in a closed-vessel system (section 2.3.8). The heat recovery that can be achieved is a proportion of the energy contained in the composting feedstock, and is measured in energy per kg of dry matter (kJ $kgDM^{-1}$). With a closed-vessel composting system, the average heat recovery was reported as 7084 kJ $kgDM^{-1}$ (Smith, Aber and Rynk, 2017), which is roughly 3.1 kWh $kgDM^{-1}$. This was a commercial, non-peer-reviewed source at industrial scale, and would need the support of an academic study to be ratified for further research.

7.8 System costs and revenue

7.8.1 CAPEX

The initial investment required in the project, i.e. the capital expenditure or CAPEX, was calculated based on the methods described in a selection of papers (Zimmermann *et al.*, 2018; Christensen *et al.*, 2005; Towler and Sinnott, 2012). There are five classes of CAPEX estimate according to the Association for the Advancement of Cost Estimating International (AACE), for use at different stages of a project which indicate how accurate the cost estimate is likely to be based on the type information used to derive it (Towler and Sinnott, 2012).

Table 7-13: AACE cost estimate classes, adapted from (Towler and Sinnott, 2012).

Class	Synonyms	Accuracy	Features
Class 5 'Order of magnitude'	Ballpark Guesstimate	±30-50%	No design information, based on other processes. For feasibility/screening.
Class 4 'Preliminary'	Approximate Study Feasibility	±30%	Based on limited cost data and design detail. To choose between design alternatives, to write feasibility studies.
Class 3 'Definitive'	Authorization Budgeting Control	± 10-15%	For authorization of funds. Rough P&ID needed.
Class 2 'Detailed estimate'	Quotation Tender Firm estimate	± 5-10%	For project cost control. Require front-end engineering design, near complete process design, full purchase list.
Class 1 'Check estimate'	Tender As-bid	± 5-10%	Based on completed design.

The TEA CAPEX estimate was categorized as 'class 4' according to these references, meaning that there would be approximately a ±30% range of cost accuracy, since no specific data about the project was known.

In the TEA model, to calculate the CAPEX value for each separate piece of equipment, the known cost for the same equipment in a similar size was converted to a cost for the appropriate size using the cost-curve equation from (Towler and Sinnott, 2012) (equation 7-1).

$$C_2 = C_1 \times \left(\frac{S_2}{S_1}\right)^n$$ *Equation 7-1*

Where: C_1 is the cost of the equipment with capacity S_1, in £.
C_2 is the cost of the equipment with capacity S_2, in £.
n is the exponent (no unit).

The exponent, n, is given by Towler and Sinnott as 0.7-0.9 for mechanical processes, 0.4-0.5 for highly instrumented processes, and 0.6 as an average over the whole chemical industry.

The CAPEX estimation method (equation 7-1) was verified by using the equation to predict the cost of the small and large composting units using different sources of comparison data, with the exponent n = 0.6 (table 7-14).

Table 7-14: Assessment of the CAPEX estimation method by estimation of composter cost using different sources of comparison data.

	Capacity m3 yr^{-1}	**Cost £**	**Source**
Comparison plant 1	7.6	11662	(Irvine, Lamont and Antizar-Ladislao, 2010)
Comparison plant 2	2800	414260	(Smith and Aber, 2017)
Small composter (using plant 1)	29.6	26457	
Small composter (using plant 2)	29.6	27016	
Large composter (using plant 1)	594	159969	
Large composter (using plant 2)	594	163352	

The two estimations were very similar for both sizes of composter and supported the use of this equation and the exponent n = 0.6.

A sensitivity analysis of the exponent n in the comparative cost equation (equation 7-1) was made for small and large composting units (figure 7-11 and figure 7-12 respectively). Two different comparison plants were used to calculate the estimated cost of the two composters. The sizes of each plant are provided in table 7-15.

Table 7-15: Sizes of composters in the TEA and comparison plants used to estimate the CAPEX cost.

	Capacity m3 yr^{-1}
Small composter	68.8
Large composter	614
Comparison plant 1	7.6

Comparison plant 2	2800

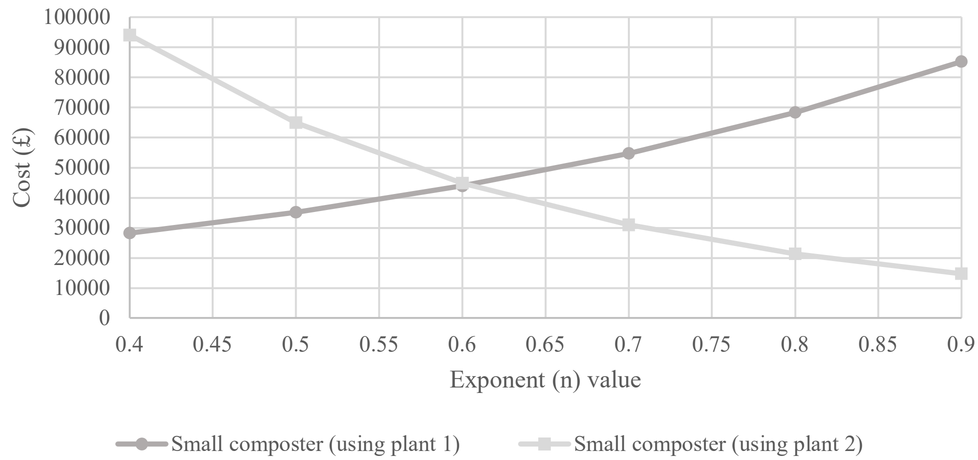


Figure 7-11: Sensitivity analysis of the exponent (n) in the prediction of the cost of the small composter using comparison plants 1 and 2.

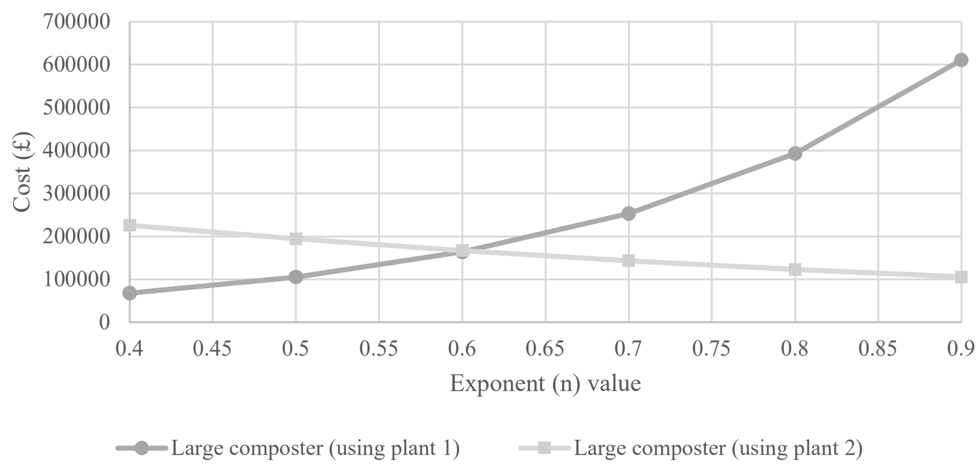


Figure 7-12: Sensitivity analysis of the exponent (n) in the prediction of the cost of the large composter using comparison plants 1 and 2.

The sensitivity analysis showed that if the size of the comparison plant was closer to the size of the plant the cost of which was to be estimated, then effect of changing the exponent n was less pronounced. If the large comparison plant was used to predict the cost of the small composter, the the outcome varied a lot more with the value of exponent used. Therefore, to attain more accurate CAPEX estimations, similarly-sized comparison plants were used to derive equipment costs wherever possible.

This CAPEX estimation method was used to calculate a cost for all large items (table 7-16).

Table 7-16: Reference and calculated CAPEX costs for large equipment in the TEA model.

	Reference plant size	Reference plant cost (£)	Model size	Model cost (£)	Reference
Grinder	350 kg hr^{-1}	600	651.8 kg hr^{-1}	871	(eCatering, 2020)
Pre-digester	3 m^3	12650	2.7 m^3	11734	(PPML, 2020)
Digester	2 m^3	7266	28 m^3	35383	(Walker *et al.*, 2017)
Greenhouse	135 m^3	14289	91.9 m^3	11343	(The Greenhouse People, 2020)
Boiler	30 kW	5000	18.3 kW	7442	(Screwfix, 2020)
Micro-CHP	10 kW	14807	5.6 kW	10481	Personal communication (SEAB)
Biogas upgrader	5 m^3 hr^{-1}	135000	3.5 m^3 hr^{-1}	108839	(Chesshire and Llewellyn, 2012)
Biogas storage	25 m^3	3371	42 m^3	4595	(BRE/WRAP, 2013)
Flare	20 m^3 hr^{-1}	11000	3.5 m^3 hr^{-1}	3860	(Alibaba.com, 2020a)
Separator	150 kg hr^{-1}	2695	30.2 kg hr^{-1}	1029	(Alibaba.com, 2020b)
Composting unit	7.6 m^3 yr^{-1}	11662	614 m^3 yr^{-1}	163221	(Irvine, Lamont and Antizar-Ladislao, 2010)
Control system	69 T yr^{-1}	11236	124.9 T yr^{-1}	14248	(BRE/WRAP, 2013)

The exchange rates used were 0.85 £ $€^{-1}$ and 0.77 £ $\$^{-1}$ (January 2020, exchangerates.org.uk). The same value of n (0.6) was initially used for all conversions, and later studied in a sensitivity analysis to see the effect of changing the cost estimation. An exception is the control system, which is highly technical and therefore used a lower n exponent, 0.4 (Towler and Sinnott, 2012).

A number of previous projects with costings were studied to ascertain a range of reference CAPEX and OPEX values, for comparison with the results of the model (table 7-17). The figures quoted in these reference projects did not give sufficient specific information (for example, exactly what costs were included in the OPEX and how each of the CAPEX costs were calculated) to be able to use them as a reference for calculations in the TEA. Instead, they were used as a sense check to verify that the costs in the TEA were roughly as might be expected.

Table 7-17: Key parameters of micro to small-scale food waste anaerobic digestion projects.

Item	Unit	Camley St, London (Walker et al., 2017)	University of Wisconsin Pilot (Gough et al., 2016)	BRE, Watford, UK (BRE/WRAP, 2013)
Yearly feed	kg	5220	22353	69000
Electrical output	kW	0.221	3.3	1.72
Heat output	kW	0.442	4.9	2.87
CAPEX	£	28309	73150	138094
OPEX	£	1935	4890	5600
OPEX as % of CAPEX	%	6.8%	6.7%	4.1%
LCOE	£ kWh^{-1}	0.577	0.120	0.311
Item	**Unit**	**Personal communication from UK company (source confidential)**	**St. Barnard, Louisiana (Moriarty, 2013)**	**Barcelona Central Market (Mata-Alvarez et al., 1992)**
Yearly feed	kg	182500	5695307	22009500
Electrical output	kW	7.9	424	147
Heat output	kW	14.5	0	221
CAPEX	£	196968	2460202	2222220
OPEX	£	3450	210499	257950
OPEX as % of CAPEX	%	1.75%	8.6%	11.6%
LCOE (20 yrs)	£ kWh^{-1}	0.086	0.090	0.114

It should be noted that in this table, the LCOE was in terms of electricity produced from a CHP.

7.8.2 OPEX

The operational expenditures, or OPEX, are the costs associated with the running of the plant, measured on a yearly basis. They can be split into variable and fixed costs; variable costs are those that change with the output of the plant, such as electricity costs and raw materials, and fixed costs are those that are the same whatever the plant input, such as insurance and salaries (Zimmermann *et al.*, 2018), but are influenced by the plant size.

OPEX can be calculated item by item, or the entire OPEX (both fixed and variable) can be calculated as 7% of CAPEX (Oreggioni *et al.*, 2017). The example studies (table 7-17) report that the OPEX is approximately 2-12% of the CAPEX. In this study, the heat and electricity use was a key factor that would differentiate between scenarios, because of the different equipment included and the optional use of a CHP or a boiler, all of which could reduce or increase the electricity and/or heat imported. The items of special interest in the OPEX were therefore the utility costs, and these were calculated specifically. The remainder of the OPEX was calculated at 5% of the CAPEX (this fraction was chosen as a reasonable estimate within the range stated above). This was termed the ‘OPEX factor’, in % of CAPEX. To allow for any inaccuracy in this estimation, a sensitivity analysis of this factor was performed to study the effect that changing the OPEX as a % of CAPEX had on the financial status of the scenario.

7.8.3 Revenues and avoided costs

The scenarios attracted a variety of revenues and avoided costs, which were used to calculate a project yearly revenue for the system (table 7-18).

Table 7-18: Values used to calculate revenues and avoided costs in the University scenario TEA.

Item	Unit	Value	Reference
Waste disposal - food	£ $tonne^{-1}$	Confidential	Personal communication, University of Sheffield
Waste disposal - card	£ $tonne^{-1}$	Confidential	Personal communication, University of Sheffield
Waste disposal - green waste	£ $tonne^{-1}$	Confidential	Personal communication, University of Sheffield
Heat energy	£ kWh^{-1}	0.0378	Bristol Energy (January 2020), personal communication
Electrical energy	£ kWh^{-1}	0.151	Bristol Energy (January 2020), personal communication
Heat RHI	£ kWh^{-1}	0.0474	(GOV.UK, 2019b)
RTFC rating for biomethane as fuel	Certificates kg^{-1}	3.8	(GOV.UK, 2020b)
RTFC certificate	£ $certificate^{-1}$	0.29	(GOV.UK, 2020b)
Diesel cost	£ L^{-1}	1.30	(Royal Automobile Club (RAC), 2020)
Diesel usage (medium van)	miles $gallon^{-1}$	48	(Powell, 2018)
Biomethane usage (medium van)	miles kg^{-1}	10	Personal communication, Red Kite Management Ltd.
Compost sale	£ kg^{-1}	0.167	(The Compost Shop, 2019)

The disposal costs for each waste stream were provided confidentially by the University of Sheffield and so cannot be explicitly presented. The disposal of vegetable oil was assumed to be free, as there are several companies in the UK that offer a free collection service (Waste Vegetable Oils, 2019). The savings made by producing biomethane as a fuel were calculated from the cost of diesel and average fuel usage of a medium-sized van. The amount of fuel saved was calculated using the average fuel usage per mile of a van running on biomethane.

7.8.4 Functional unit

The main chosen functional unit was the simple payback time, which was calculated as the CAPEX divided by the net yearly profit. This gives a simple method of comparing different scenarios, taking into account their CAPEX and OPEX.

Another method of calculating costs is to use the LCOE (Levelized Cost of Energy or Levelized Cost of Electricity). The LCOE is a common comparison factor between energy generation sources, and is normally specific to electricity generation (Business Energy and Industrial Strategy (BEIS), 2016b) (figure 7-13).

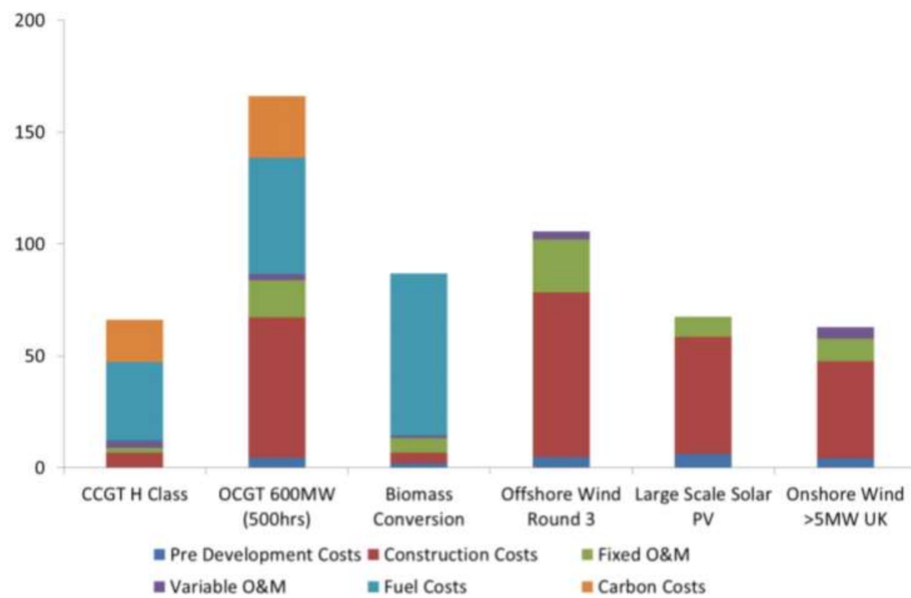


Figure 7-13: Levelized cost of energy in £ MWh_e^{-1} for low-carbon electricity generation technologies in 2020 (Business Energy and Industrial Strategy (BEIS), 2016b).

The expected LCOE for large-scale anaerobic digestion is 0.099-0.103 £ kWh_e^{-1} (0.116-0.121 € kWh_e^{-1}), which is mid-range compared to other renewable and non-renewable energy sources (Arup, 2016). For smaller installations, a larger LCOE would be expected. The LCOE was calculated for each of the example projects in table 7-17, and produced a range between 0.086 and 0.577 £ kWh_e^{-1}.

In this TEA, there were several different forms of energy produced in different scenarios; biogas as a fuel, electricity and heat from a CHP, heat from a boiler, and heat from composting. Different forms of energy have a different unit value; for example, electricity is more expensive than heat as it is more expensive to produce but has a wider range of uses. Because of the differing energy values, the LCOE could not be used to compare like with like, and so was not used as a comparison tool in this TEA.

7.8.5 Project Net Present Value

The simple payback for a project can be used to compare different scenarios, but to see whether a project is economically viable, the net present value (NPV) and the discounted cash-flow rate of return (DCFROR) must be calculated. These take into account the time value of money (payback sooner is worth more than payback later, as the money can be reinvested) and also consider the time needed for construction and ramping up of the system (Towler and Sinnott, 2012). The NPV calculates the value of the investment at the end of a given year, taking into account an interest rate (the DCFROR). If the NPV is positive, the project is in profit. The DCFROR can then be adjusted up or down to find the maximum allowable interest rate that the project can pay and still remain in profit (NPV ≥ 0).

7.9 Results

7.9.1 Key performance indicators of the scenarios

The CAPEX costs were estimated individually (figure 7-14) to generate a total CAPEX for each of the different scenarios.

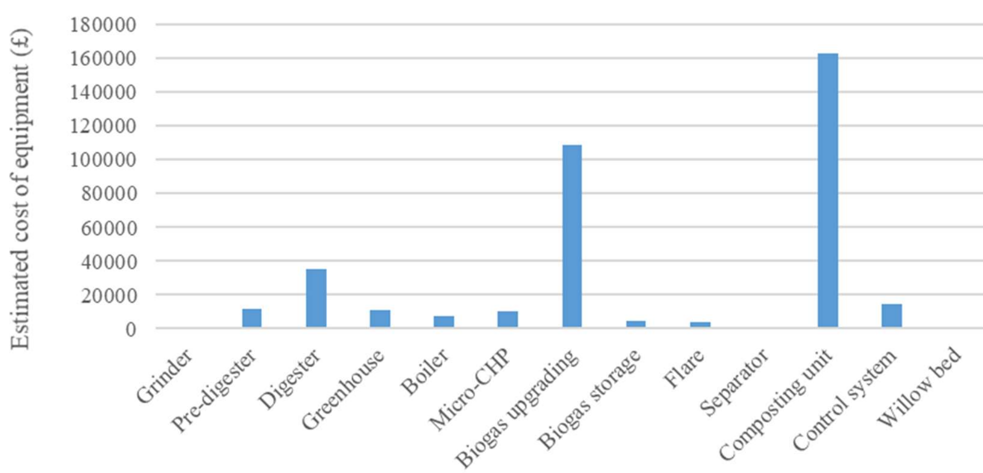


Figure 7-14: Estimated CAPEX equipment costs for large pieces.

The largest CAPEX investments in the project were the biogas upgrader and the composting unit (with heat recovery). These were both emerging technologies; this explains why the costs were high, and would be likely to reduce when the technology matured. In a real scenario, using these technologies might attract grants for technology development.

The engineering and project management costs were calculated at 30% of the equipment cost, and a contingency of 10% of the equipment cost was also added (Towler and Sinnott, 2012).

A summary of the key performance indicators of each scenario was generated using the information gathered in the inventory (table 7-19). The main functional unit used for comparison was the simple payback, calculated by dividing the CAPEX by the yearly net revenue, which therefore does not take into account any loan interest or discount factor (section 7.8.5). The simple payback cannot be used to predict the economic feasibility of a real project, however it was useful for comparison between scenarios.

Table 7-19: Key operational and economic parameters for each instance of the university scenario.

	Scenario	**Project CAPEX (£)**	**Yearly OPEX (£ yr^{-1})**	**Yearly revenue (£ yr^{-1})**	**Simple payback (years)**
1a	With pre-digester, separator and boiler	230643	16705	41620	9.3
1b	With pre-digester, separator and CHP	234894	13113	37864	9.5
1c	With pre-digester, separator and biogas upgrader	372594	26112	57326	11.9
2a	No pre-digester, composting whole digestate, with boiler	398565	21353	91970	5.6
2b	No pre-digester, composting whole digestate, with CHP	402820	20141	90593	5.7
2c	No pre-digester, composting whole digestate, with biogas upgrader	540520	30760	107676	7.0
3a	With pre-digester, composting whole digestate, with boiler	414229	23286	91741	6.1
3b	With pre-digester, composting whole digestate, with CHP	418484	20924	89204	6.1
3c	With pre-digester, composting whole digestate, with biogas upgrader	556184	32714	107447	7.4

The lowest CAPEX investment was required for a system that included a pre-digester, separator, boiler and small composting facility (1a). However, the revenue for this scenario, and other scenarios that incorporated separation and limited composting (i.e., 1b and 1c) was low, because less heat and less compost were produced, and there was less waste avoidance (figure 7-15). The energy produced in this case was heat energy only, which has a lower value than electricity (0.0378 £ kWh^{-1} for natural gas as opposed to 0.151 £ kWh^{-1} for electricity, estimated from energy bills in January 2020).

The fastest simple payback time was achieved by the scenario that included no predigester, a boiler, no separator and a large composter (2a). This would produce a simplified system without a pre-digester pasteurisation step – instead, the composting stage would sanitise the system's output to satisfy food waste regulations. The composter would need to have temperature sensors, to provide evidence that the digestate was being held at a hot enough temperature for long enough to bring about pasteurisation.

The revenues for scenarios 1a and 2a were compared (figure 7-15).

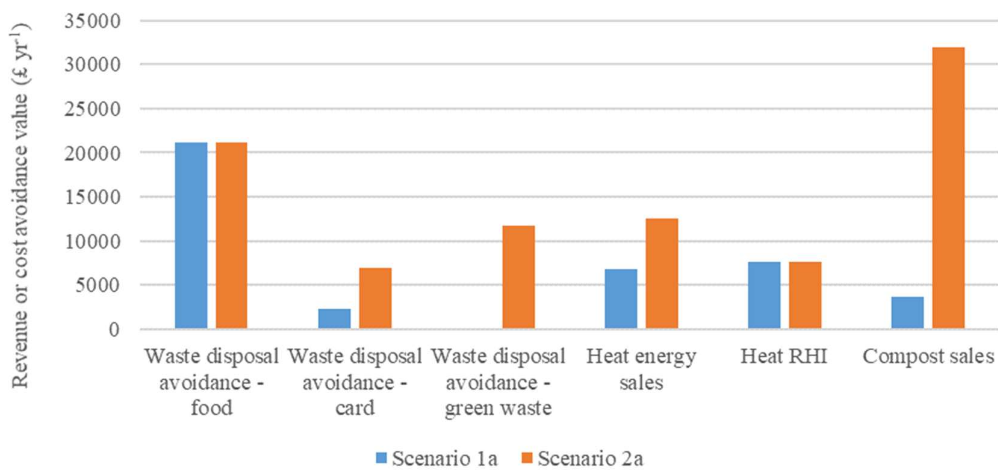


Figure 7-15: Revenue and cost avoidance comparison between scenarios 1a and 2a.

The analysis showed that the main differences in revenue between 1a and 2a were in waste disposal avoidance and compost sales. These would likely be sensitive factors in the TEA.

All versions of a scenario that incorporated a biogas upgrader were the most expensive option compared to those incorporating a boiler or CHP. This was due to the high CAPEX investment for a biogas upgrading unit.

The CAPEX and simple payback time for each of the scenarios are shown for comparison (figure 7-16 and figure 7-17). The TEA was a level 4 assessment, meaning that the certainty of the CAPEX estimate was ±30% (section 7.8.1), and the uncertainty for each indicator was calculated based on this error.

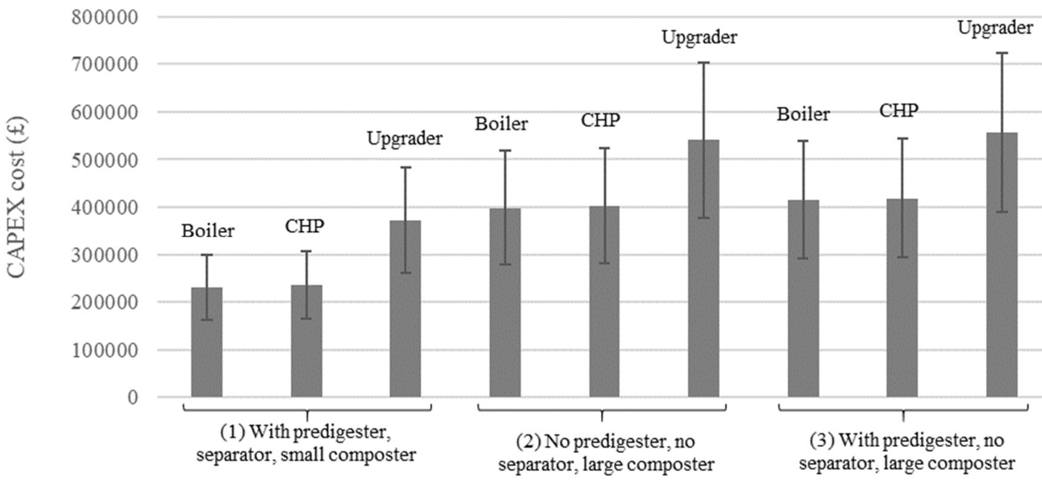


Figure 7-16: CAPEX costs for scenarios 1a to 3c. The error bars show the difference in capital costs effected by changing the CAPEX by ±30%.

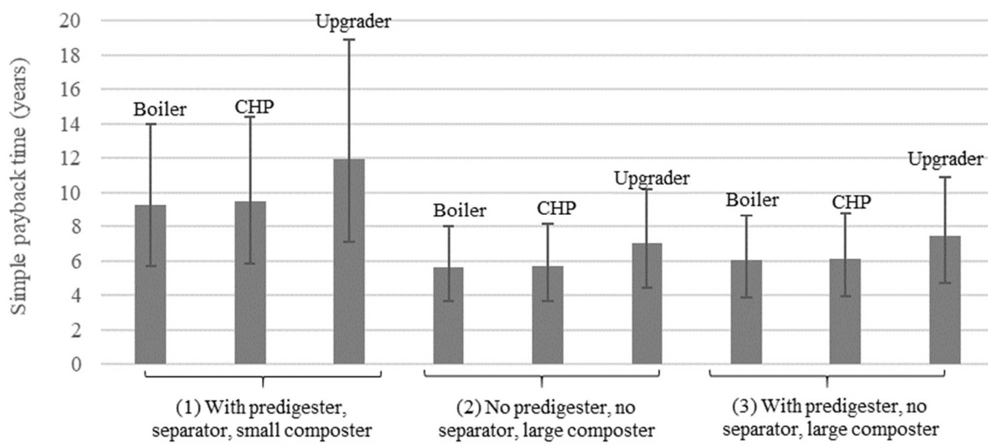


Figure 7-17: Simple payback time for scenarios 1a to 3c. The error bars show the difference in payback time effected by changing the CAPEX by ±30%.

The results of the TEA showed that the scenarios that included a large composter (composting whole digestate) and/or a biogas upgrading unit required a far higher initial CAPEX. However, the yearly revenues were also higher and therefore the simple payback time was shorter than scenarios that included a smaller composter (composting separated digestate). The large composting unit would process all of the digestate from the plant into compost, and therefore solve the issue of digestate disposal as well as processing the green waste and cardboard waste streams. However, it would also be a significant undertaking in that it would require trained operators, a large area to accommodate the machinery and the compost storage, and potentially would also require a small tractor to move solids and containers. These factors would all be important in deciding the system structure and would depend on the requirements and capabilities of the institution in question.

7.9.2 Outputs of the system

The energy and product outputs were analysed (figure 7-18, figure 7-19, figure 7-20). This analysis was a useful tool for assessing each scenario, but not for comparison between scenarios due to differences between the values and types of output.

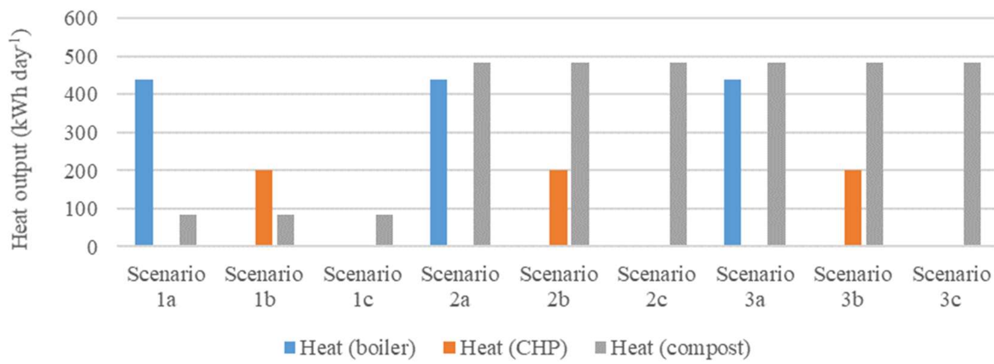


Figure 7-18: Heat produced in scenarios 1a to 3c, from a boiler, CHP or composting system.

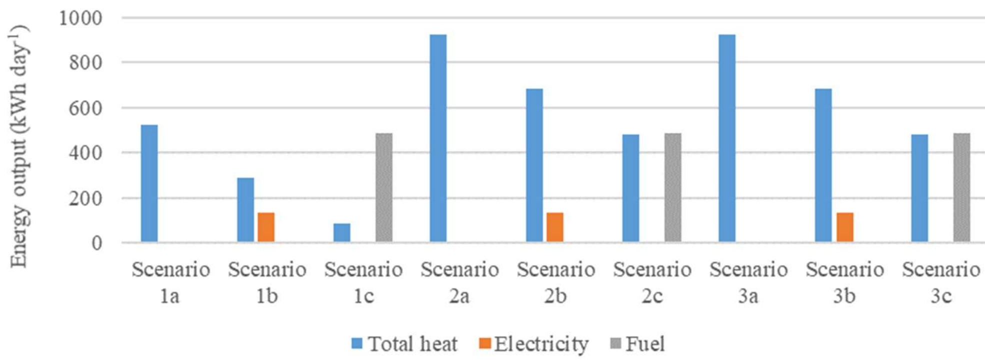


Figure 7-19: Energy outputs for scenarios 1a to 3c, as heat, electricity or fuel.

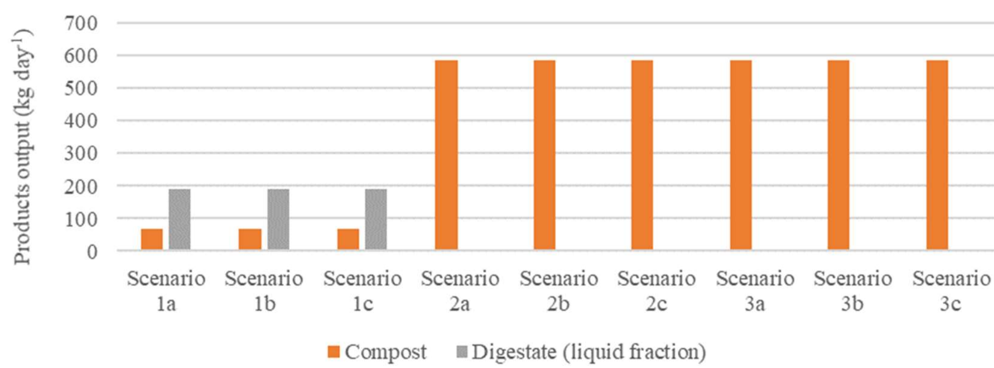


Figure 7-20: Product outputs from scenarios 1a to 3c, as biomethane, compost, or digestate.

The heat outputs from the scenarios (figure 7-18) showed that the compost system had the potential to produce as much heat as the boiler. This was dependent on achieving the expected heat recovery from the composting system, taken as 7084 kJ $kgDM^{-1}$, which was quoted in a review based on normal composting processes (Smith, Aber and Rynk, 2017). It is possible that when composting digestate, less energy would be available as some would have already been released as biogas in the anaerobic digestion process. This would have a larger effect on the scenarios in which digestate made up a larger proportion of the compost mix.

The heat from composting was expected to be ‘low-grade’ heat at approximately 47°C (Irvine, Lamont and Antizar-Ladislao, 2010), which has limited uses compared to the high-grade heat produced by a boiler or CHP (over 70°C), and is therefore less valuable. Similarly, when analysing the energy output of each scenario, electricity has a greater value than heat, but arguably biomethane has a greater variety of uses as it can be stored and converted.

The ‘solid fertiliser’ outputs of digestate and compost were also not comparable as their value also varies, according to their beneficial content such as fibre, nitrogen, phosphorus and potassium.

This analysis would be made use of in a real project by considering the markets available for each of the products, and whether they could be used in-house. Scenarios which included a more useable product might be favoured, even if less was produced.

7.9.3 Sensitivity to the exponent n in the comparative cost equation

The value ‘n’ is the exponent used in calculating the equipment CAPEX cost. The effect of changing the exponent n on the simple payback time was studied, for the scenarios with the lowest and highest simple payback time and the scenarios with the lowest and highest CAPEX investment (figure 7-21, figure 7-22).

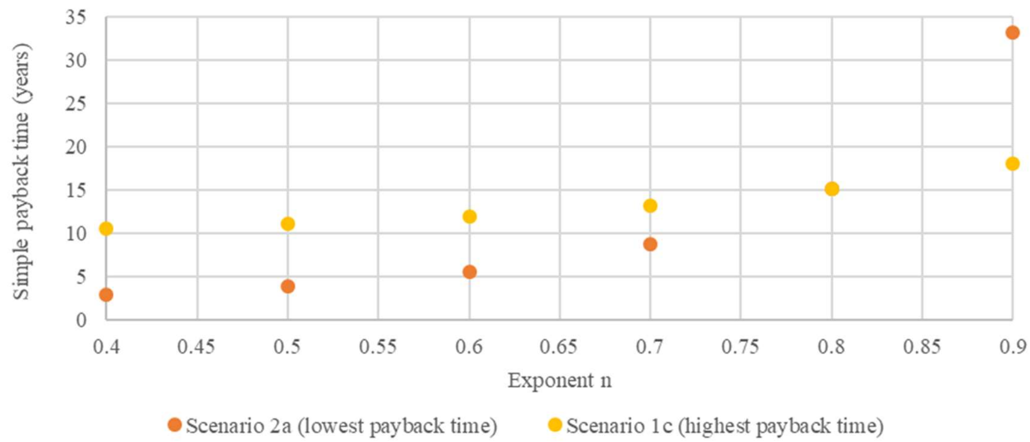


Figure 7-21: Sensitivity of the simple payback time to changes in the exponent n for scenarios with a low and high simple payback time.

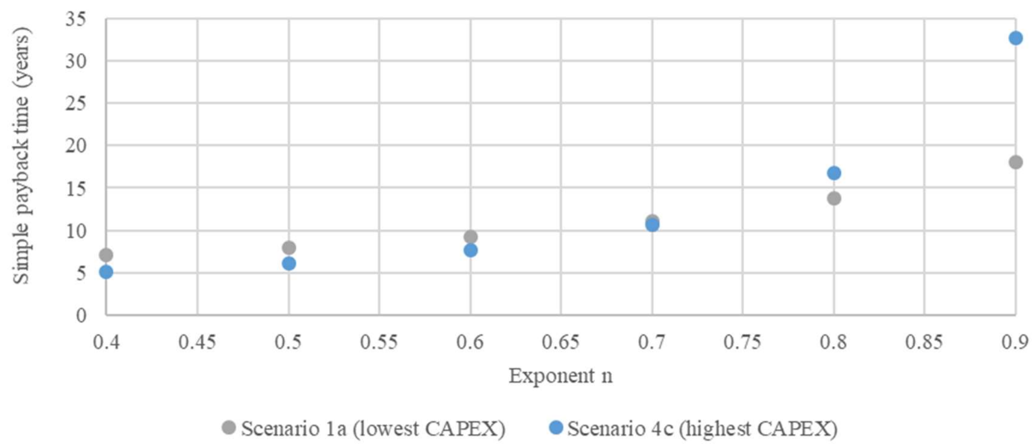


Figure 7-22: Sensitivity of the simple payback time to changes in the exponent n for scenarios with a low and high CAPEX cost.

This analysis shows that projects with a short simple payback time or high CAPEX were more affected by changes in the value of n. As n was an estimation, it would therefore be important when working in projects of this type (with a short simple payback time or high CAPEX) to obtain accurate quotes for equipment costs as soon as possible in the project timeline, to reduce this uncertainty.

7.9.4 Sensitivity to the OPEX factor

The OPEX was calculated by multiplying the CAPEX by 5% (referred to as the OPEX factor in this study), and then adding the significant scenario-specific charges (electricity, gas and willow tree purchase). This calculation is an estimation and therefore a sensitivity analysis was performed to see how much effect this 'OPEX factor' had on the simple payback time if it was increased or decreased.

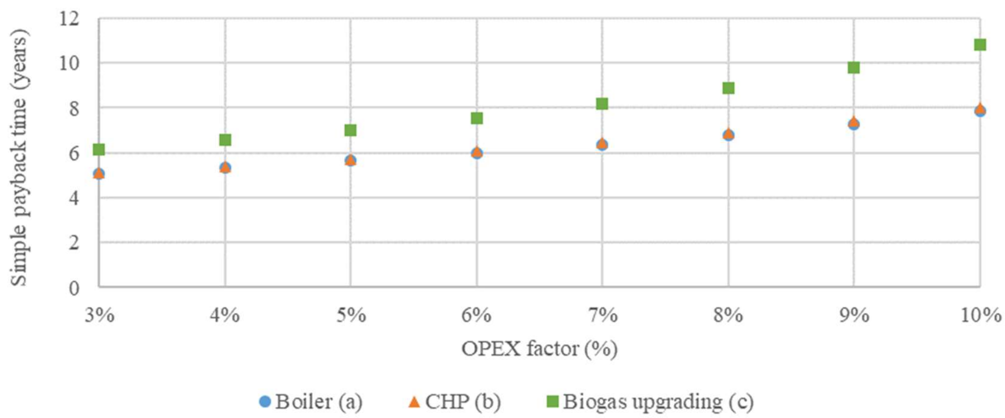


Figure 7-23: Sensitivity analysis for the effect on the simple payback time for scenarios 2a, 2b and 2c when changing the OPEX factor.

The analysis showed that the simple payback time increased exponentially with OPEX factor. Increasing the OPEX factor from 5% to 10% increased the simple payback time by between 52% and 67% depending on the CAPEX of the scenario.

Table 7-20: Financial statistics for scenarios 2a, 2b and 2c (without pre-digester, composting whole digestate, with boiler (a) CHP (b) and biogas upgrader (c).

	CAPEX (£)	**OPEX (£ yr^{-1})**	**Revenue (£ yr^{-1})**
Scenario 2a	398565	22778	91970
Scenario 2b	402820	20141	96605
Scenario 2c	540520	34495	107676

The change in OPEX factor had more of an effect on scenario 2c compared to scenarios 2a and 2b. This scenario had a much larger CAPEX, and therefore a much larger OPEX (table 7-20). The revenue was also higher but this would not be affected by the OPEX factor. Therefore, for projects with greater financial risk (i.e. higher CAPEX), the correct estimation of the OPEX factor is more important. If the OPEX factor is overestimated (i.e. it is set too high), this will cause less inaccuracy in the simple payback time than if the OPEX factor is underestimated. For high-risk projects it would therefore be prudent to overestimate the OPEX factor, or ideally obtain an accurate figure for the OPEX as early on in the project as possible.

7.9.5 Parasitic energy losses

The parasitic energy loss is the proportion of energy that was produced by the system that is needed to keep the system running, for example to power the equipment or heat the digester. The parasitic energy losses for other similar projects were studied in order to find an expected range (table 7-21).

Table 7-21: Parasitic loads as a % of plant output from a selection of references.

Feedstock	Gross energy output (kW)	CHP electrical efficiency (%)	CHP thermal efficiency (%)	Parasitic electricity	Parasitic heat	Reference
Food waste	272	32%	53%	27.3%	30.3%	(Banks *et al.*, 2011)
Grass	2789	25%	30%	7.0%	1.5%	(Salter and Banks, 2008)
Cattle slurry	344	25%	30%	56.9%	12.4%	(Salter and Banks, 2008)
Grass and cattle slurry	1566	25%	30%	12.5%	2.7%	(Salter and Banks, 2008)
Grass and food waste	2502	25%	30%	14.6%	1.7%	(Salter and Banks, 2008)
Food waste	0.884	25%	50%	31.7%	18.0%	(Walker *et al.*, 2017)
Cattle slurry	154	30%	45%	4.30%	4.60%	(Oreggioni *et al.*, 2017)
Food waste	-	32%		20.0%	-	(Patterson *et al.*, 2011)

The parasitic losses in this table were all reported as a proportion of the CHP output. The CHPs in different references had different heat and thermal efficiencies, so this represents an unfair comparison as the efficiencies were theoretical rather than practical and the parasitic losses are therefore subject to an estimation. For a fairer comparison the parasitic losses should instead be reported as a proportion of the gross energy output, which would remove this uncertainty.

The range of parasitic losses is very large for electricity (4.3 to 57%) and smaller for heat/thermal (1.5 to 30.3%). Because of this, it is not possible to state what an ‘expected’ value of parasitic electricity or heat would be.

After the initial study, further scenarios were investigated to explore two areas of interest (table 7-22). Firstly, the expansion of composting to use both separated and whole digestate, in order to balance the nutrients and the dry matter content of the compost (scenarios 4a, 4b, 4c, 5a). Secondly, for scenarios 2a and 5a, which were the most promising, the removal of the greenhouse to see the effect on the parasitic heat demand.

Table 7-22: Initial and later scenarios for the TEA analysis.

	Pre-digester	Boiler	CHP	Biogas upgrader	Separator	Composting whole digestate	Composting separated digestate	Green-house
1a								
1b								
1c								
2a								
2b								
2c								
3a								
3b								
3c								
4a								
4b								
4c								
5a								
6a								
7a								

The parasitic energy losses were calculated for each of the TEA scenarios, using the gross energy output as biogas and using a CHP electrical and thermal efficiency of 30% and 45% respectively (table 7-23). For all scenarios, the gross energy output (the energy content of the biogas produced) was 20.4 kW.

Table 7-23: Parasitic losses for each TEA scenario, in terms of gross energy output (as biogas) and as a % of theoretical CHP output. Assumes a CHP electrical and thermal efficiency of 30% and 45% respectively.

	Electricity use (kW)	**Heat use (kW)**	**Parasitic electricity demand (% of gross)**	**Parasitic heat demand (% of gross)**	**Parasitic electricity demand (% of CHP output)**	**Parasitic heat demand (% of CHP output)**
1a	2.9	1.1	14%	5%	47%	12%
1b	2.9	1.1	14%	5%	47%	12%
1c	4.6	1.1	23%	5%	76%	12%
2a	1.1	0.4	5%	2%	18%	5%
2b	1.1	0.4	5%	2%	18%	5%
2c	2.8	0.4	14%	2%	46%	5%
3a	1.9	1.1	10%	5%	32%	12%
3b	2.0	1.1	10%	5%	32%	12%
3c	3.7	1.1	18%	5%	61%	12%
4a	2.3	1.1	11%	5%	38%	12%
4b	2.2	1.1	11%	5%	36%	12%
4c	3.9	1.1	19%	5%	64%	12%
5a	1.4	0.4	7%	2%	23%	5%
6a	1.3	0.8	6%	4%	21%	9%
7a	2.2	2.2	11%	11%	36%	24%

The parasitic losses all fall in the same range as the comparison projects. The parasitic heat demand increase by the removal of the greenhouse was shown by scenarios 6a and 7a, which were analogous to scenarios 2a and 4a respectively with the greenhouse removed. The removal of the greenhouse doubles the parasitic heat demand in both cases.

The parasitic electricity demand was broken down by equipment for these scenarios to determine which were the biggest electricity users (figure 7-24).

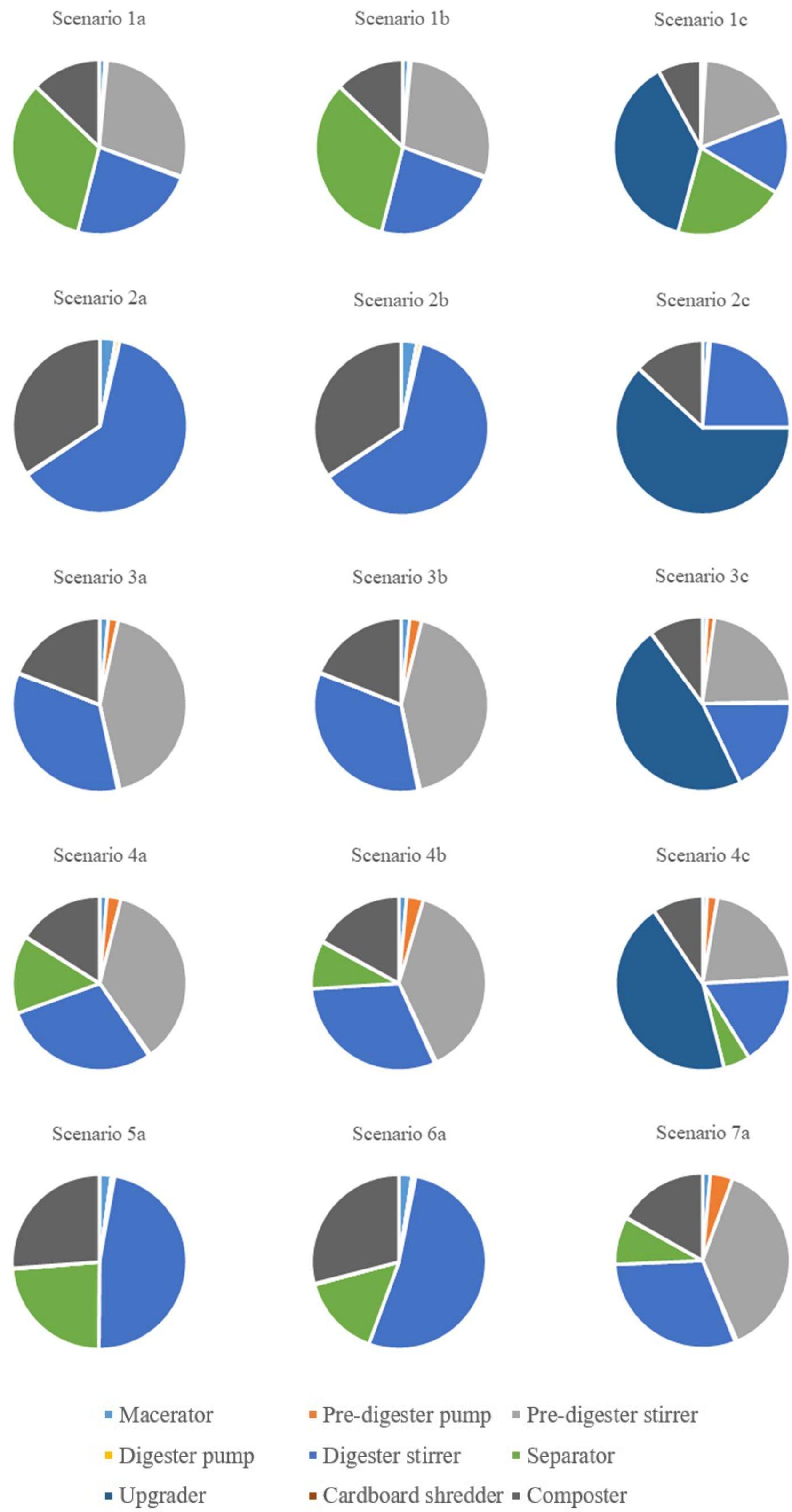


Figure 7-24: Parasitic electricity losses by equipment for each scenario.

This analysis shows that the pre-digester stirrer, digester stirrer, biogas upgrader, separator and composter were the highest electricity users in the system. Each of these pieces of equipment had a long operational time and so to reduce the parasitic electricity demand, the operational time should be reduced where possible.

A sensitivity analysis was performed on the effect on the simple payback time of changing the parasitic electricity and parasitic heat (figure 7-25). The scenario used was 3b (with pre-digester, without separator, with CHP, composting whole digestate) as it included a CHP and had a high simple payback time.

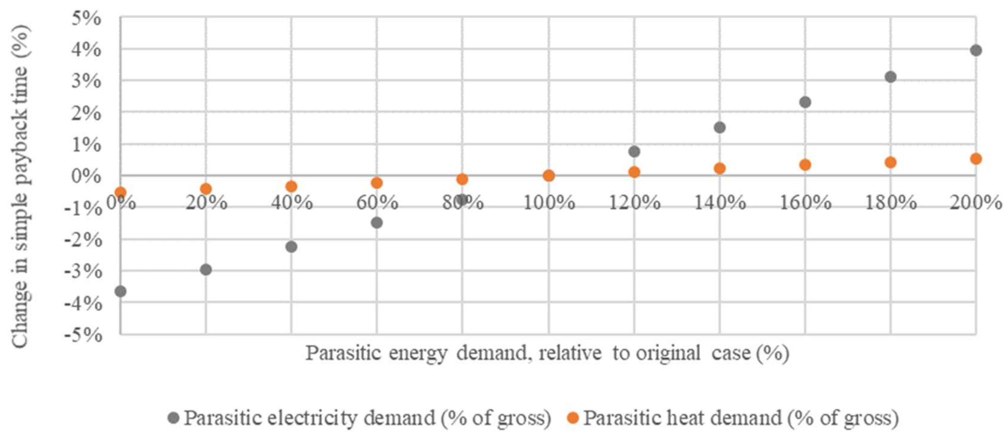


Figure 7-25: Scenario 3b: Sensitivity analysis of the effect of change of parasitic electricity and heat demand on the simple payback time (with pre-digester, without separator, with CHP, composting whole digestate). The original simple payback time was 6.1 years.

The analysis showed that the parasitic electricity demand had a greater effect on the simple payback time than the parasitic heat demand, however neither effect was very large. The sensitivity analysis was repeated with a scenario in which the heat and electricity were a larger proportion of the yearly OPEX – scenario 1b, which included a pre-digester, separator, CHP and was composting separated digestate solids (figure 7-26), and had the highest heat usage of all the scenarios.

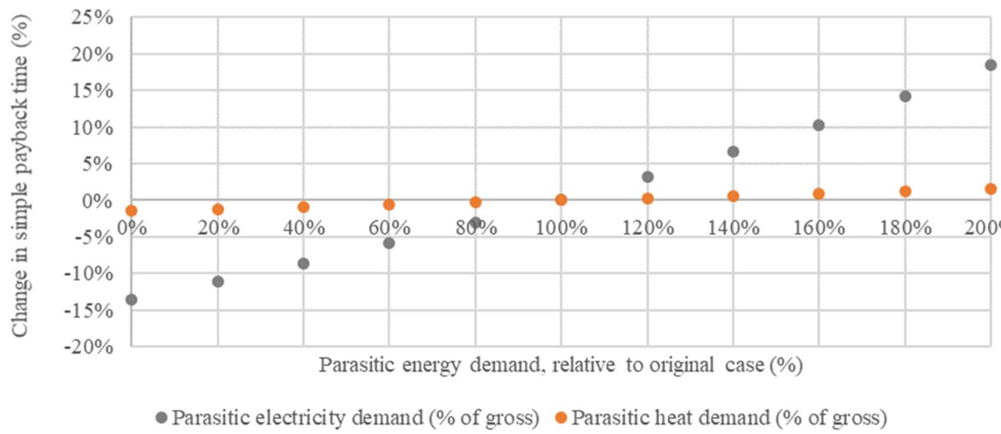


Figure 7-26: Scenario 1b: sensitivity analysis of the effect of changes in the parasitic electricity and heat demand on the simple payback time (with pre-digester, with separator, with CHP, composting separate digestate solids). The original simple payback time was 9.5 years.

The analysis was then repeated with scenario 1c, which had the highest electricity usage (figure 7-27).

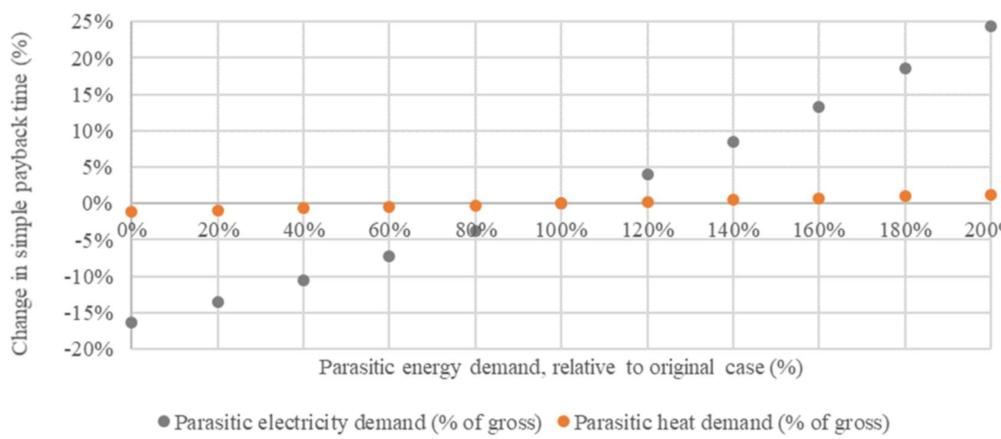


Figure 7-27: Scenario 1c: sensitivity analysis of the effect of changes in the parasitic electricity and heat demand on the simple payback time (with pre-digester, with separator, with CHP, composting separate digestate solids). The original simple payback time was 11.9 years.

The sensitivity of scenarios 1b, 1c and 3b to changes in the parasitic electricity demand were then compared, to see if more electricity demand had a more significant influence on the simple payback time (table 7-24, figure 7-28).

Table 7-24: CAPEX, OPEX and electricity use of parasitic electricity use comparison scenarios.

	Total CAPEX (£)	**Total OPEX (£ yr^{-1})**	**Total electricity use (kWh day^{-1})**
Scenario 1b	234894	13113	69
Scenario 1c	372594	26112	111
Scenario 3b	418484	20924	47

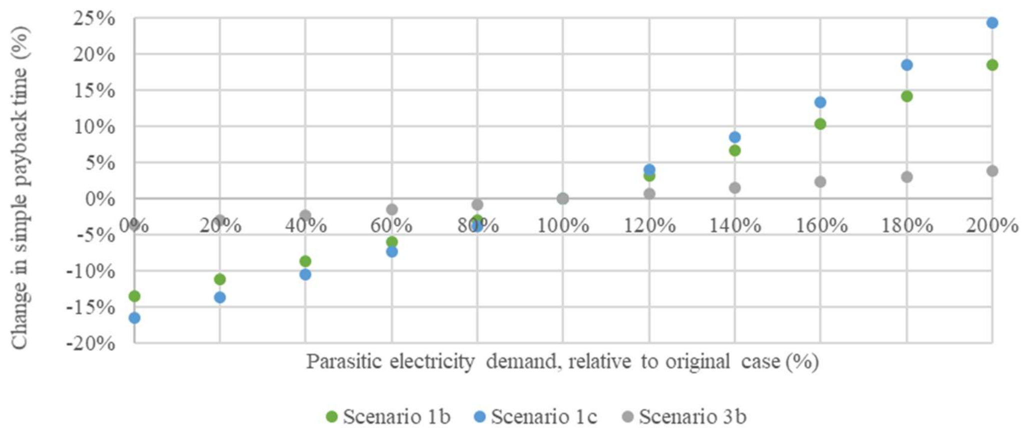


Figure 7-28: Comparison of the change in simple payback time with change in parasitic electricity demand, for scenarios 1c and 1b.

As would be expected, scenario 1c is most affected by changes in parasitic electricity demand, which is a result of the high total electricity use. The least affected by changes in parasitic electricity demand is scenario 3b, which has the lowest electricity use.

This study demonstrates that parasitic electricity use had a much greater effect on the simple payback time than the parasitic heat demand, particularly if the parasitic heat demand was high, and that to reduce this effect, it should be minimised where possible. The most effective way to do this, as shown by figure 7-24, would be to reduce the running time of the equipment that runs for long periods such as the digester stirrers and composter.

7.9.6 Sensitivity to electricity and gas prices

The variation of energy prices in previous years was studied to determine the likelihood of future changes and any trends (figure 7-29, figure 7-30).

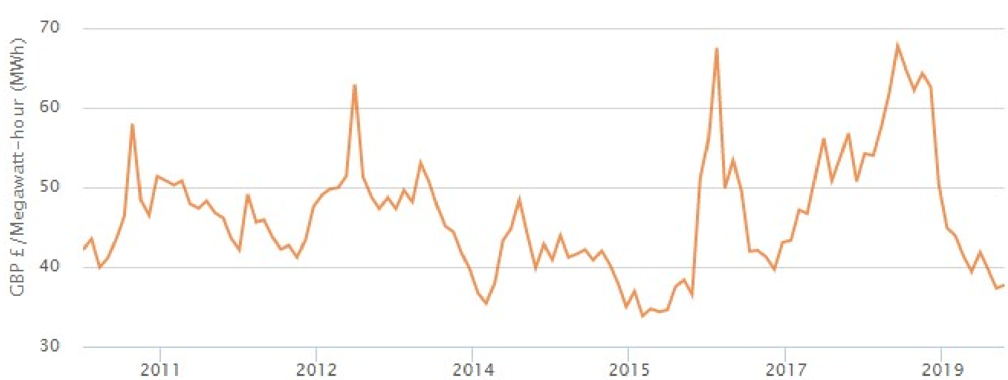


Figure 7-29: Electricity prices: Day-ahead baseload contracts - monthly average (GB) (OFGEM, 2020a).

Figure 7-30: Gas prices: Day-ahead baseload contracts - monthly average (GB)(OFGEM, 2020b).

The gas and electricity prices over the past ten years have been changeable but show no trend either increasing or decreasing. The electricity and gas prices show increases and decreases at similar times; this is because a large proportion of the UK's electricity is generated from natural gas (37% in 2017 (GOV.UK, 2018a)). As the penetration of renewable energy sources increases, these prices would be expected to 'uncouple'. However, it is not possible to predict at this stage how quickly this might happen and what the effect would be on either price.

The different scenarios consumed and produced varying amounts of heat and power, and so would be differently affected by any changes in gas and electricity price (it was assumed that where heat is required, this would be supplied by a gas boiler). Therefore, a sensitivity analysis was performed to study the effect that the electricity and gas price might have on the simple payback time. The gas and electricity supply prices used were £0.0394 kWh^{-1} (GOV.UK, 2017b) and £0.156 kWh^{-1} (Statista, 2020) respectively. The scenario used for comparison was the most profitable scenario (scenario 2, with no pre-digester, no separator and a large composter)(figure 7-31, figure 7-32). The price range used was 90 to 200% of the 'current' gas and electricity prices. This range was used as the price was more likely to increase than decrease.

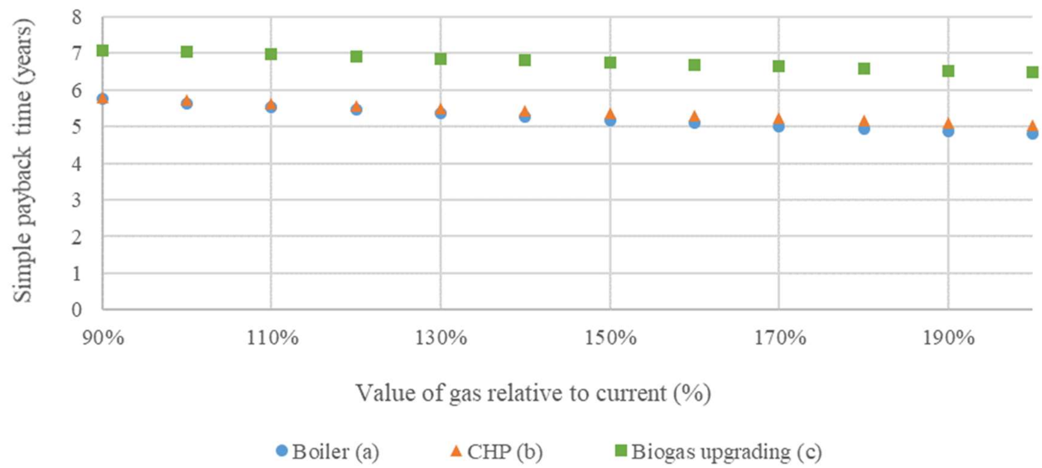


Figure 7-31: Analysis of the sensitivity of the simple payback time to the changing cost of supplied gas, relative to the current price, for instances of scenario 2 (no pre-digester, no separator, large composter).

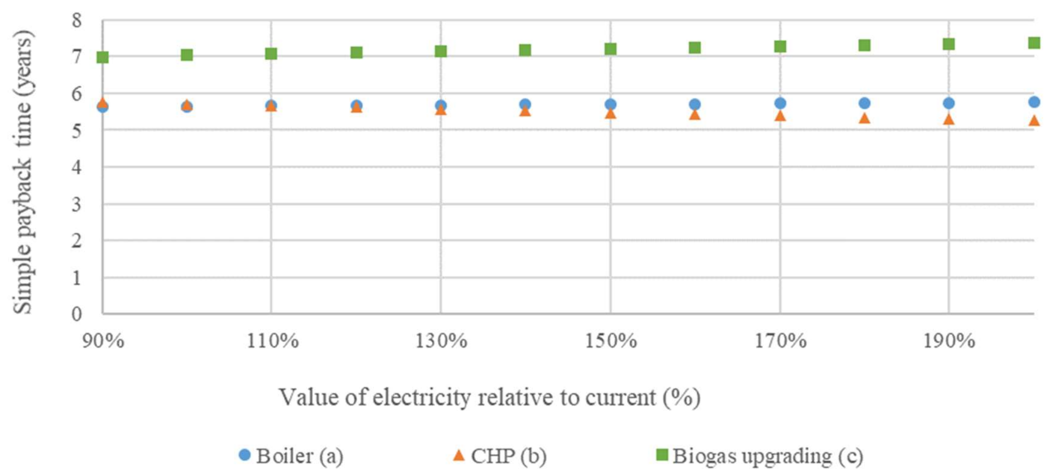


Figure 7-32: Analysis of the sensitivity of the simple payback time to the changing cost of supplied electricity, relative to the current price, for instances of scenario 2 (no pre-digester, no separator, large composter).

The analysis showed that as the gas price increased, the payback time of all the instances decreased, with the most effect on the scenario that used a boiler (2a). An increasing electricity price produced a slight increase in payback time for the boiler and biogas upgrading scenarios, and a slight decrease in payback time for the CHP scenario, as this instance generates electricity.

7.9.7 Sensitivity to RHI value

The UK government introduced the Renewable Heat Incentive (RHI) in 2011, at a rate of £0.083 for each kWh of heat produced from biogas plants under 200 kW_{th} (GOV.UK, 2019b). The rate has now dropped to £0.0474 kWh^{-1} (figure 7-33) and is set at this level until 31st March 2021, when it will be reviewed (ICAX, 2019).

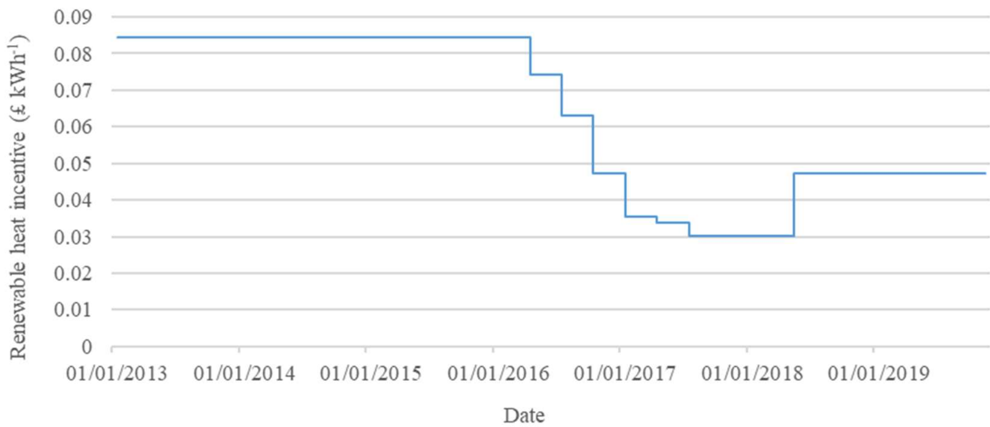


Figure 7-33: Renewable heat incentive payment rates in the UK from 2013 to present for biogas installations under 200 kW_{th}.(GOV.UK, 2019b).

As of 2018, the UK had achieved less than half the targets for heat and transport by 2020 set out in the 2009 Renewable Energy Directive (Ambrose, 2018; European Union, 2009) and therefore it is likely that the RHI will remain at the same value or increase after 2021, to stimulate investment in renewable heat and transport.

The feed-in tariff scheme, which ran alongside the RHI scheme and provided incentives for renewable electricity production, was closed to new applications in March 2019, and so was not included in this TEA study (OFGEM, 2019).

A sensitivity analysis of scenario 2 was performed (figure 7-34), which was selected as it was the scenario with the shortest simple payback time.

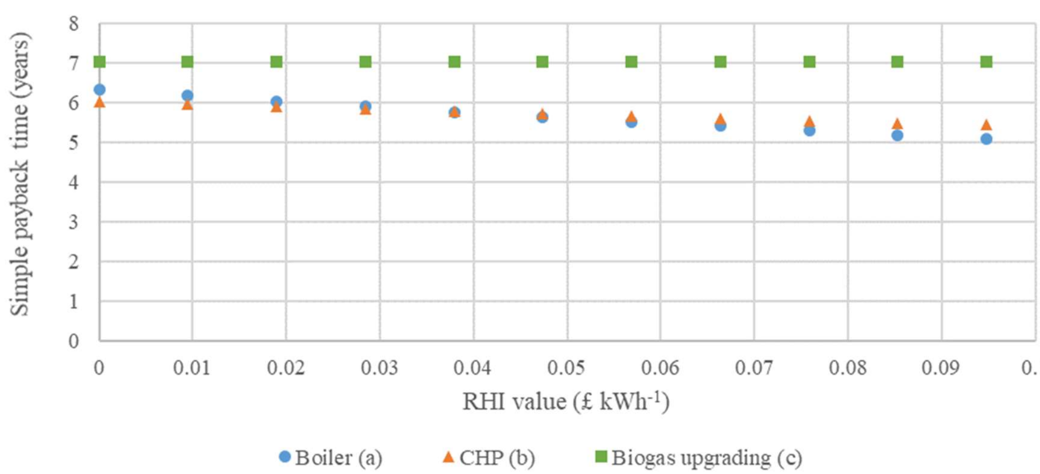


Figure 7-34: Sensitivity of different instances of scenario 2 (no pre-digester, no separator, large composter) to the variance in RHI value.

This analysis showed that the scenario that included a boiler was most sensitive to increases and decreases in the RHI, with a change of ±10.5% if the RHI increased or decreased by 100%

of the current value (0.0474 £ kWh^{-1}). If the RHI decreased, the CHP option became the most profitable, whereas at the current RHI rate and at higher rates, the boiler option was the most profitable. The choice of biogas utilisation technology is therefore important if it is deemed likely that energy incentives will change.

RHI payments are granted eligibility on a case-by-case basis, and heat from compost processes does not currently qualify for RHI payments. However, heat from composting is an energy product of interest in this study. Therefore an analysis was performed for the case of compost heat being eligible for RHI (figure 7-35).

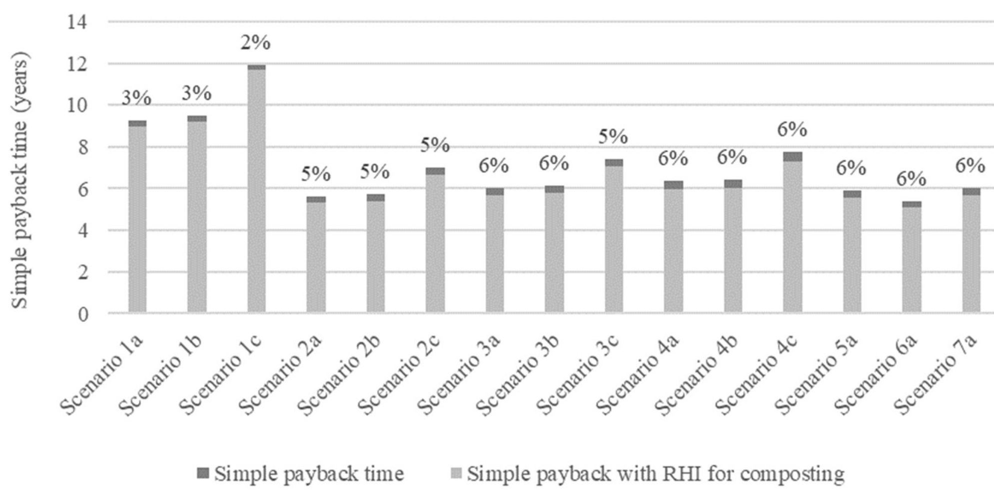


Figure 7-35: Simple payback time and payback time with RHI payments for compost heat added for all scenarios. The % difference between the two figures is shown above the bar.

The addition of the RHI payments for heat from compost would make a maximum of 6% decrease in the simple payback time.

7.9.8 Biogas upgrading scenario sensitivity to RTFC value

Biomethane produced for vehicle fuel attracts Renewable Transport Fuel Certificates (RTFCs) in the UK, at a rate of 3.8 RTFCs per kg (GOV.UK, 2020b). These can be traded with companies that have a renewable transport fuel obligation (RTFO) to fulfil, for example non-renewable fuel producers. The RTFCs are traded on an index, which is currently about £0.29 per RTFC (Energy Census, 2020). The effect of changing the two incentives was studied (figure 7-36).

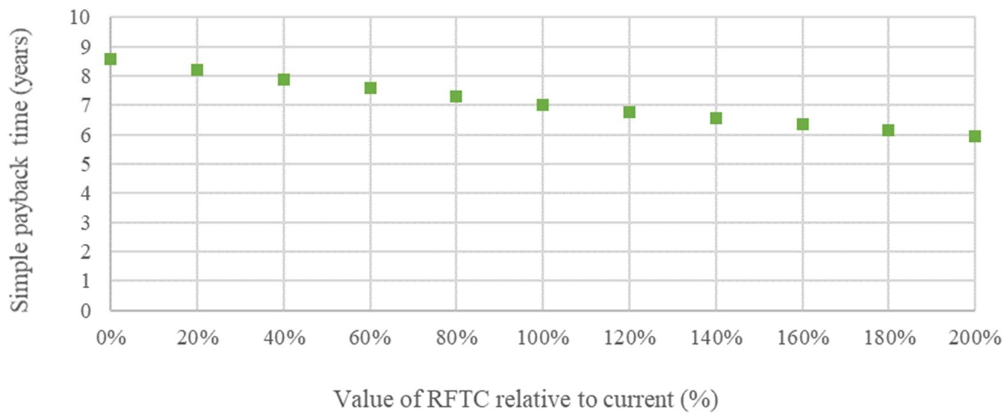


Figure 7-36: Analysis of the sensitivity of the simple payback time to changes in the RTFC and RHI values for scenario 2b (no pre-digester, no separator, large composter, biogas upgrader).

Setting the RTFC value at the current rate produced a simple payback time of 7.0 years. If the RTFC was reduced, the payback time increased and vice versa. This reflected the fact that the RTFC was a much greater proportion of the yearly revenue for the scenario (24% compared to 1.2%).

7.9.9 Biogas upgrading scenarios' sensitivity to onsite usage and fuel prices

The biogas upgrading unit would produce a large amount of biomethane for vehicle fuel (the equivalent of approximately 126,000 miles of fuel) and this fuel could be used within the organisation, sold or injected into the grid. The value of the biomethane depended on its end use. When used as an onsite replacement for diesel, it had a value of approximately 1.23 £ kg^{-1}, but when sold or injected into the grid, its value was approximately 0.85 £ kg^{-1}. Therefore, it was important to use as much biomethane as possible onsite.

A sensitivity analysis was performed to study the effect of reduced on-site biomethane usage (figure 7-37).

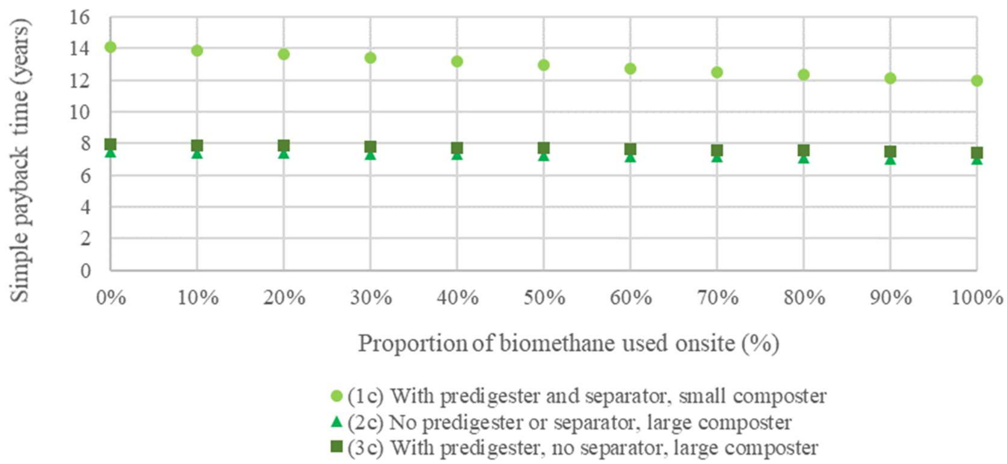


Figure 7-37: Sensitivity analysis of the simple payback time against the proportion of biomethane used on-site, for scenarios using a biogas upgrader.

This analysis showed that the scenario most sensitive to the amount of biomethane used on site was the one that included a pre-digester, separator and small composter (1c), that is, the scenario with the lowest CAPEX, in which the biomethane revenue made up a greater proportion of the overall revenue.

A second sensitivity analysis was performed on the simple payback time, this time by altering the price of diesel compared to its current price (figure 7-38).

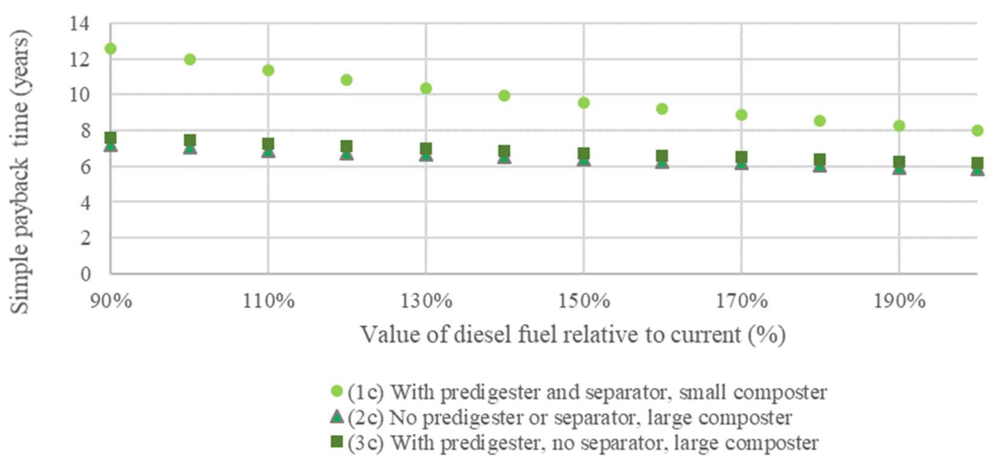


Figure 7-38: Sensitivity analysis of the simple payback time in scenarios that include a biogas upgrading unit (1c, 2c, 3c) to increases in diesel fuel prices.

The sensitivity analysis showed that increases and decreases in fuel prices had the most impact on the economics of the scenario that included a pre-digester, separator and small composter (1c). That is because in this scenario, a larger proportion of the revenue was from the biomethane – the other two scenarios had more income composting. However, even with a

200% increase in the cost of diesel, the scenarios that included whole digestate composting had a shorter simple payback time.

Reviewing the fuel prices for the last 10 years (figure 7-39), the petrol and diesel prices are somewhat volatile (ranging from 101.1p to 148.0p, a difference of 46%) but there is no pattern of either increase or decrease. It is therefore not possible to predict the likelihood that fuel prices will rise or fall in the future.

Figure 7-39: Petrol and diesel prices at the fuel pump from 2010 to 2020. The y-axis shows the cost in pence (RAC Foundation, 2020).

With a biogas upgrader, there would be 'external' expenses such as a grid connection (the cost of which varies greatly, depending on size and location) and one or more biogas-powered vehicles, to fully realise the investment. The biomethane standard would need to be tested regularly, to satisfy the standards for biomethane injection (DECC, 2009). Therefore the capital costs could be expected to be higher in a real scenario and the process would be prohibitively complex.

7.9.10 Sensitivity to the price of compost

The price of compost varies considerably depending on the quantity it is sold by (bags, bulk bags, loose per metre cubed) and depending on the quality and source. The value used in the TEA was 0.15 £ kg^{-1} (The Compost Shop, 2019), or 90 £ m^{-3} using a compost density of 600 kg m^{-3} (Severn Waste, 2018), which is a standard price for bulk compost. A sensitivity analysis was performed to study the effect of a change in the value of compost (figure 7-40).

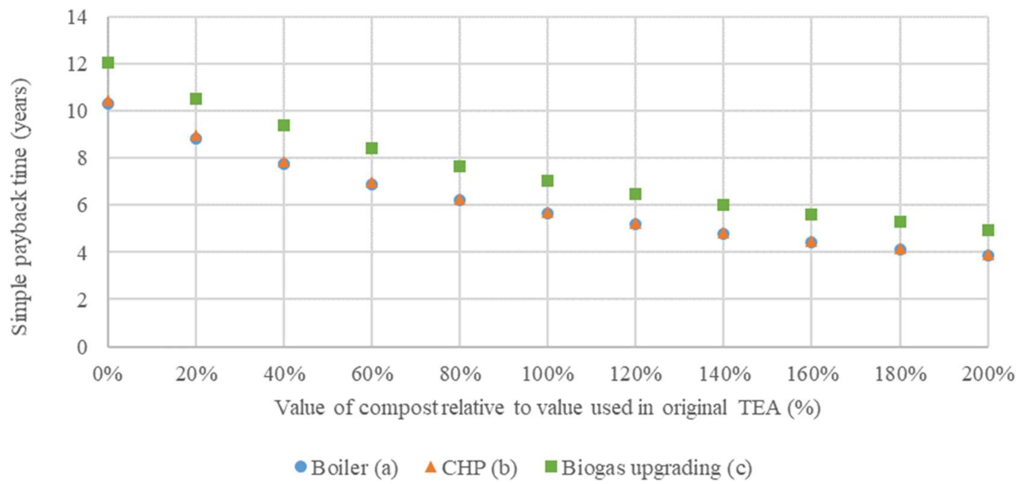


Figure 7-40: Sensitivity analysis of the simple payback time for scenarios 2a, 2b and 2c (no pre-digester, no separator, composting whole digestate).

The analysis showed a high sensitivity to the value of compost, particularly if the compost value decreased. If the compost was sold at twice the price, the simple payback time for scenarios 2a and 2b (with a boiler and CHP respectively) reduced to under 4 years, which is a reduction of 31.6%. If the compost was not sold (the price was 0 £ kg^{-1}), the payback time of the project increased by up to 84.2% (figure 7-41).

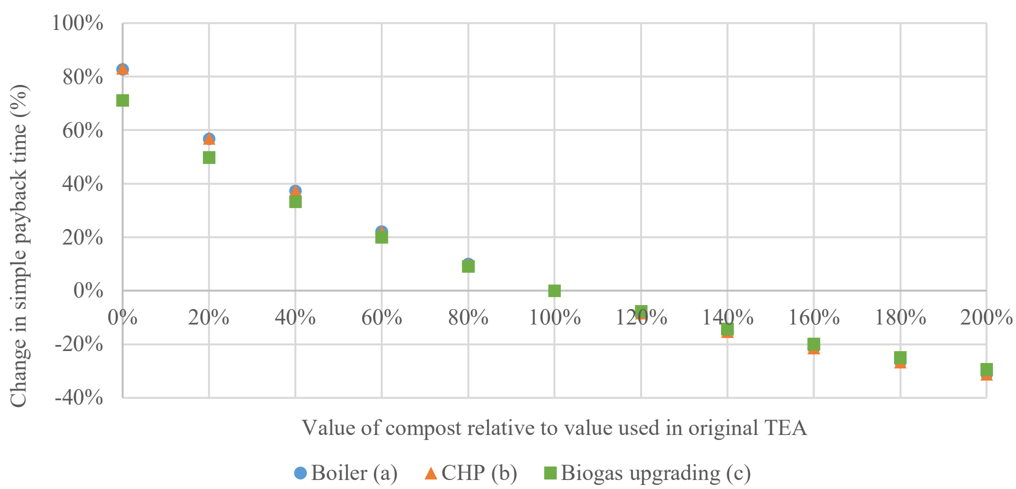


Figure 7-41: Sensitivity analysis of the simple payback time for scenarios 2a, 2b and 2c (no pre-digester, no separator, composting whole digestate).

This analysis shows that in all scenarios, securing a reliable market for the compost at a good price would be critical to the success of the project. This could be aided by government initiatives such as subsidies to establish a market and by achieving certification as an organic fertiliser.

7.9.11 Sensitivity to the heat production from composting

The heat production from composting used in the TEA was 7084 kJ $kgDM^{-1}$ (Smith, Aber and Rynk, 2017). This was an average heat production from a review, but the heat production reported has been variable and the technology is not well established, so this figure contained potential uncertainty. A sensitivity analysis on scenarios 2a, 2b and 2c to study the effect of an increase or decrease of heat production from composting was performed (figure 7-42).

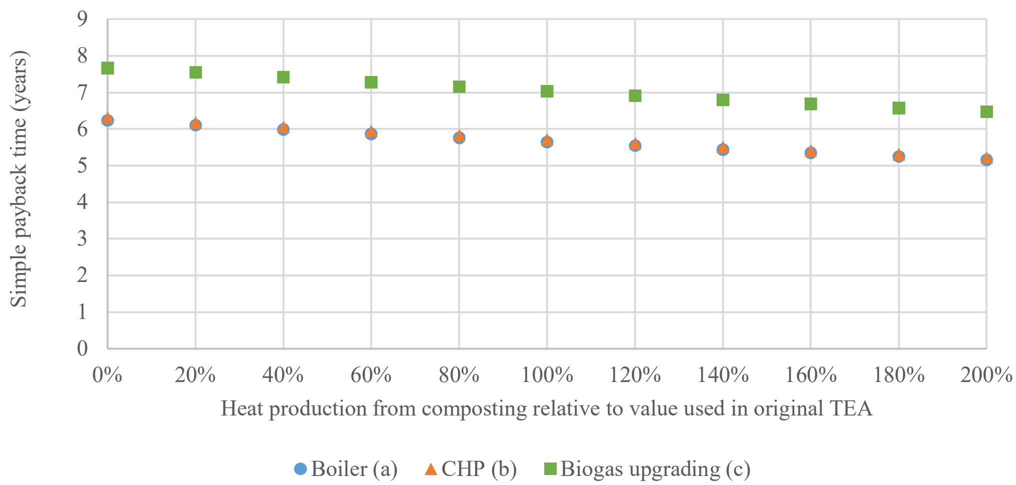


Figure 7-42: Sensitivity analysis of simple payback time variation with heat produced from composting for scenarios 2a, 2b and 2c.

The analysis showed that an increase or decrease of 100% in the heat production from composting would increase or decrease the simple payback time for each of the three scenarios by less than a year, which is not a large variation. If this technology were used however, it would be important to confirm the heat production experimentally.

7.9.12 Ideal composting mixtures

In first three scenarios of the study (scenarios 1, 2 and 3), it was assumed that the digestate produced by the AD process would be either completely separated into liquid and solid fractions, or left whole and composted with other inputs. There were advantages and disadvantages to both methods. When the digestate was separated, a very large area of willow bed was required to process the liquid fraction and only a small proportion of the other waste streams (cardboard, green waste, shredded willow cuttings) could be incorporated into the composting process, but the composter needed was relatively small and therefore less expensive compared to the composter for whole digestate. When the whole digestate was composted, more waste streams could be included but a very large composter was needed. An investigation was made to see whether it was advantageous to use a combination of both streams (table 7-25). As discussed previously (section 7.7.3) the ideal moisture in a compost heap is 45-60% (solids content 40-55%) and the optimal C:N ratio is 25:1 to 35:1 (Cooperband, 2000).

Table 7-25: Key features of 'boiler' scenarios using different combinations of whole and separated digestate in the composting stage.

Feature	Scenario 1a With predigester and boiler, composting separated digestate solids	Scenario 2a No predigester, with boiler, composting whole digestate	Scenario 3a With predigester and boiler, composting whole digestate	Scenario 4a With predigester and boiler, composting whole and separated digestate solids	Scenario 5a No predigester, with boiler, composting whole and separated digestate solids
Digestate (kg yr^{-1})	0	87206	87206	35754	35754
Solid fraction of digestate (kg yr^{-1})	17441	0	0	10290	10290
Willow trimmings (kg yr^{-1})	0	0	0	14764	14764
Cardboard (kg yr^{-1})	14108	42752	42752	42752	42752
Green waste (kg yr^{-1})	0	90613	90613	90613	90613
Compost C:N ratio	20.5	20.1	20.1	34.9	34.9
Compost solid content (%)	55%	45%	45%	55%	55%
Simple payback time (years)	9.3	5.6	6.1	6.4	5.9
Plant footprint (hectares)	2.51	0.04	0.04	1.51	1.51

When the separated digestate solids are composted (scenario 1a), only 33% (18383 kg yr^{-1}) of the cardboard waste stream can be added – any more cardboard would place the solids content outside the 'ideal'. However, the C:N ratio is in the required range. When the whole digestate is composted (scenarios 2a and 3a), 100% of all the available waste streams can be added, which brings about a significant cost benefit, and reduces the payback time to 5.6 years (or 6.1 years with a pre-digester included). However, the C:N ratio is below the ideal range, meaning that the mixture will not compost as well and will not produce a nutritionally optimal compost (Dimambro, Lillywhite and Rahn, 2006). When both whole and separated digestate are composted (scenarios 4a and 5a), both the C:N ratio and the solid content are within the required range, and the plant footprint is smaller than in scenario 1a. However, the simple payback time is longer than the scenarios with no separation step.

Therefore, if there was no space available to plant a willow bed and the C:N ratio of the resulting compost was not critical, then composting the whole digestate would be the best option. To produce a better quality compost (albeit at a slightly higher cost), composting a mixture of whole and separated digestate would be preferred.

7.9.13 Analysis of the effect of adding a greenhouse

In a previous study of a micro-scale AD plant, the digester and ancillary tanks were located in a greenhouse, which reduced the heating requirements of the digester by 49% (Walker *et al.*, 2017). The effect on the simple payback time of adding a greenhouse in the CAPEX for two different scenarios was investigated (figure 7-43).

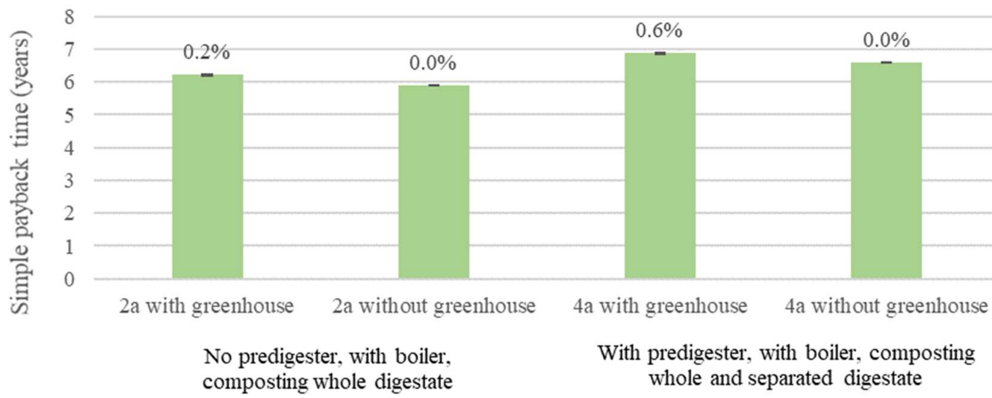


Figure 7-43: Comparison of the simple payback time of scenarios with and without a greenhouse. The effect of changing the amount of heat saving the greenhouse provides (from 0% to 100%) is shown by the error bars and labelled above the bar for clarity.

The results of the study showed that adding a greenhouse increased the simple payback time. The cost benefit brought about by the reduction in heating requirement was far outweighed by the cost of the greenhouse. The proportion of heat saving the greenhouse provides was investigated (shown by the error bars in figure 7-43) – even when the heat saving was 100% (meaning the digester needed no heating at all), the difference in simple payback time was only 0.2% less. Therefore it was concluded that a greenhouse should only be added to the system if it does not incur a capital cost (that is, it is already present).

7.9.14 Changing the feed amounts and the plant size

The feedstock amounts that were used in the TEA study were estimated according to the data provided by the university, and the equipment and containers for the plant were sized based on these amounts. The amount of feedstock in the model was altered to see the effect that reducing the size of the plant would have on the payback period (figure 7-44).Figure 7-44: Comparison of the simple payback time of three different scenarios with different sized plants.

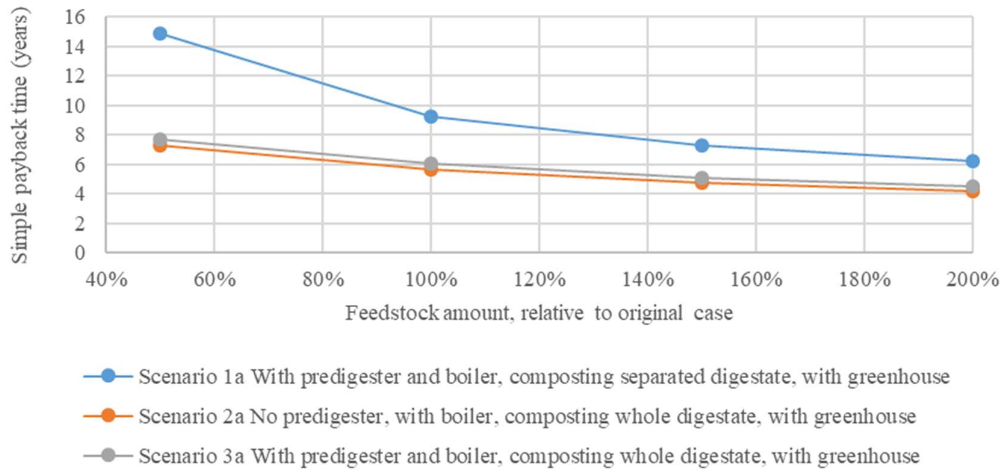


Figure 7-44: Comparison of the simple payback time of three different scenarios with different sized plants.

An increase in plant size decreased the payback time, and vice versa, showing that there is an economy of scale for the system. For scenario 1a, which had a lower CAPEX but a longer payback time than scenarios 2a and 3a, the effect of increasing or decreasing the plant size was more pronounced.

7.9.15 NPV and the effect of interest

For 'real-life' projects, the payback time must include interest to reflect the change in value of capital in the future compared to the present time, as capital in the present has a higher value. The difference is calculated using a discount factor. In the sensitivity analysis, scenarios were generally compared by simple payback time, but it was possible that the scenarios with a higher capital investment would be more affected by applying an interest rate, so this was investigated.

Using an interest rate of 9%, the payback time for each scenario was calculated, and then compared to the simple payback time (figure 7-45).

Table 7-26: Scenarios for the TEA analysis.

	Pre-digester	Boiler	CHP	Biogas upgrader	Separator	Composting whole digestate	Composting separated digestate	Green-house
1a								
1b								
1c								
2a								
2b								
2c								
3a								
3b								
3c								
4a								
4b								
4c								
5a								
6a								
7a								

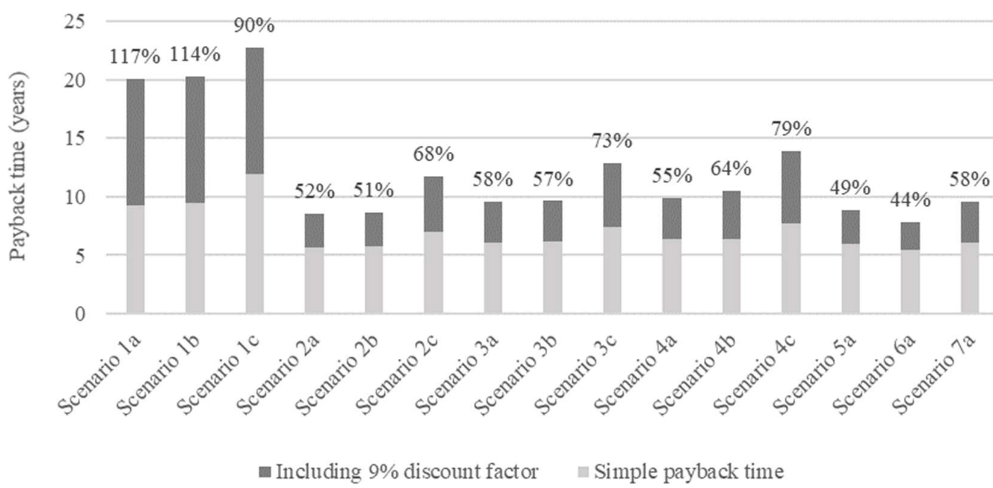


Figure 7-45: Simple payback time and payback time with 9% discount factor applied for all scenarios. The % difference between the two figures is shown above the bar.

The graph shows that the effect of applying a discount factor is to increase the payback time by between 44 and 117%. The increase in payback time was plotted against CAPEX and then against simple payback time to see if there was a relationship between these factors.

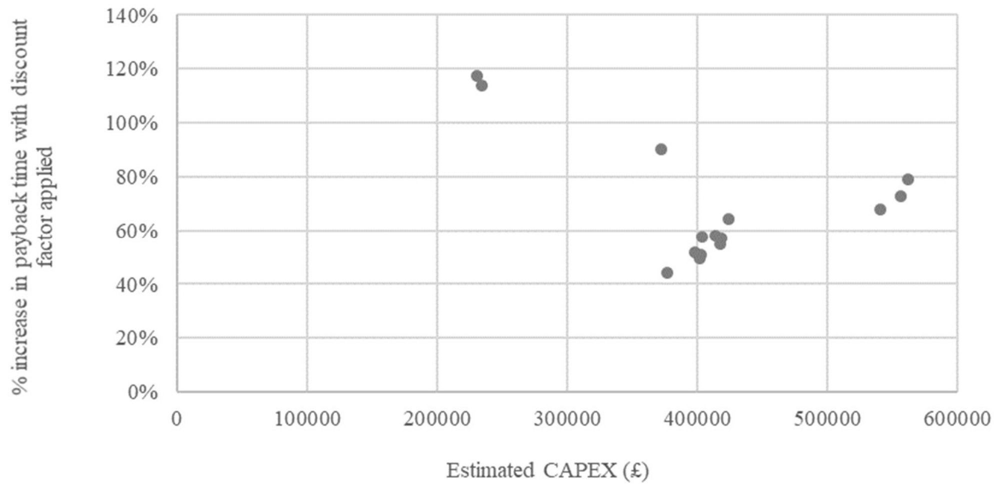


Figure 7-46: Increase in payback time against estimated CAPEX for all scenarios.

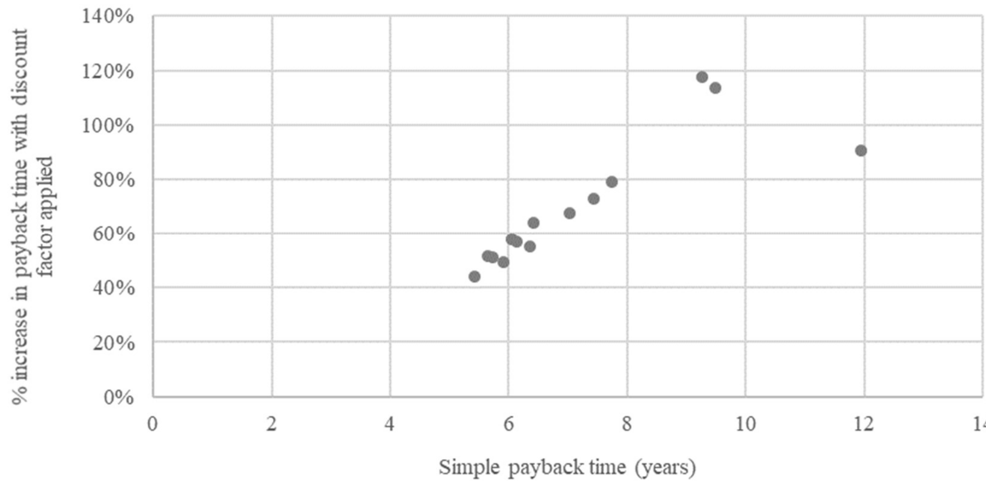


Figure 7-47: Increase in payback time with discount factor applied against simple payback time for all scenarios.

These comparisons showed that the % increase in the payback time with a 9% discount factor applied had no discernable relation to the estimated CAPEX (figure 7-46) – that is, if a scenario had a higher CAPEX, the payback time did not increase more that other, cheaper, scenarios. However, if the simple payback time was longer, then the payback time would increase by more when a discount factor was applied, in a roughly linear pattern (figure 7-47). This means that the simple payback time is closely related to the payback time with a discount factor applied. This added validity to the use of simple payback time to compare the scenarios, and showed that the yearly revenue, rather that the initial investment, was the more important factor in profitability of the scenario.

7.9.16 Best-case scenario

Collating the information gathered in the TEA, the solution with the quickest payback (2a, 5.7 years) was the scenario that did not include a pre-digester or separator, instead composting the whole digestate, with a boiler running on the biogas. However, this scenario may not be ideal, as the compost it produces would have a low C:N ratio (the starting C:N ratio is 20.1, which is below the 'ideal' range) and it may not comply with animal by-products regulations in terms of pasteurisation. The best-case scenario therefore would be to include a pre-digester and boiler in the system, but to compost both whole and separated digestate (figure 7-48).

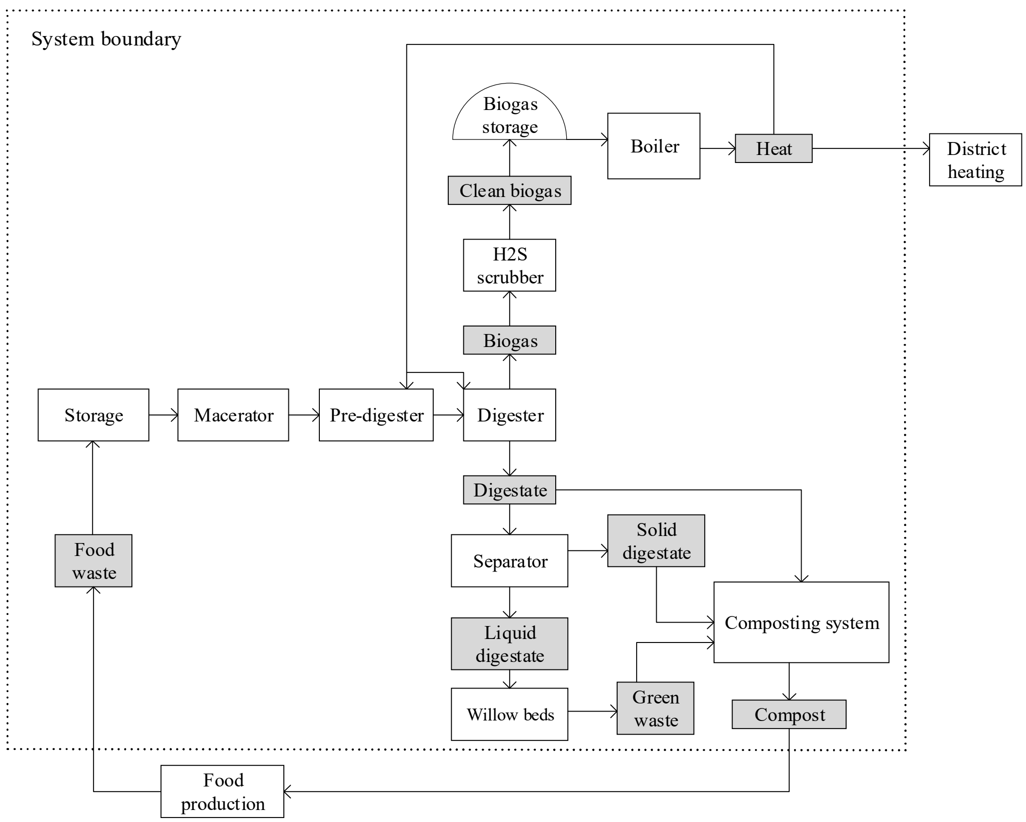


Figure 7-48: Diagram of the 'best-case scenario' for the TEA study.

The key parameters of this scenario are provided in table 7-27.

Table 7-27: Key parameters of the 'best-case scenario' for the TEA study.

Parameter	Value
Food waste processed (T yr^{-1})	119.0
Vegetable oil processed (T yr^{-1})	6.0
Cardboard waste processed (T yr^{-1})	42.8
Green waste processed (T yr^{-1})	90.6
CAPEX (£ yr^{-1})	403849
Yearly revenue (£ yr^{-1})	90424
Simple payback time (years)	6.1
Boiler output (kW)	18.3
Compost heat output (kW)	21.3
Footprint (hectares)	1.16
Compost production (T yr^{-1})	204
Compost solid content (%)	53%
Compost C:N ratio	30.2

This scenario produced a large amount of compost, for which an outlet would need to be secured before the project started. The footprint of the plant (including the willow bed) is 1.16 hectares, and so a relatively large space is required. If the required space was not available, then the best option to choose would be to compost the whole digestate instead.

7.9.17 Composting only

A scenario was modelled in which the AD system was removed completely and the inputs were process solely through a composting system with heat recovery. The system included a macerator before the composter and a large compost storage container, with enough room for 180 days' worth of compost, to allow for an accumulation at the times of the year when spreading is not permitted (figure 7-49).

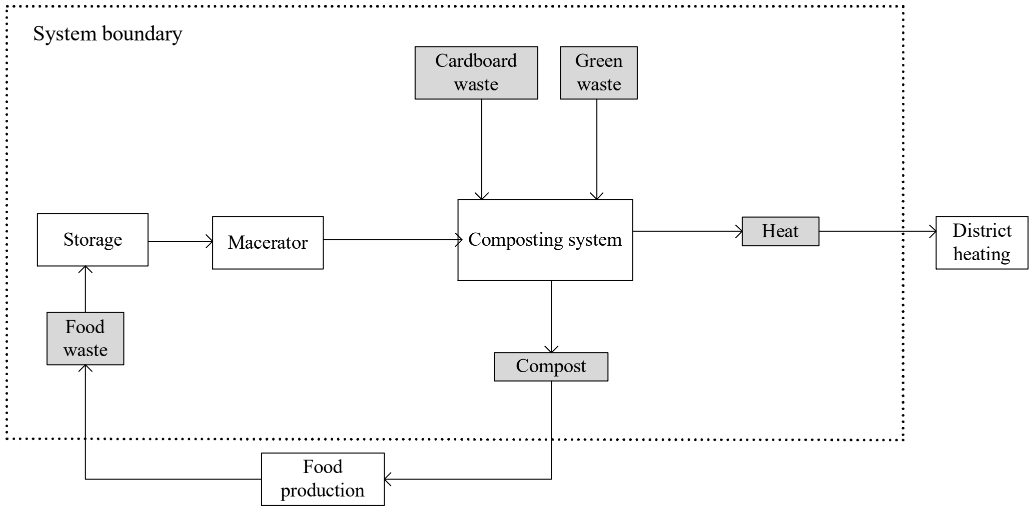


Figure 7-49: Composting system with heat recovery.

The composting system was compared to the 'best-case' scenario in terms of its capacity to process the waste input, its economics, and its output (figure 7-50, table 7-28).

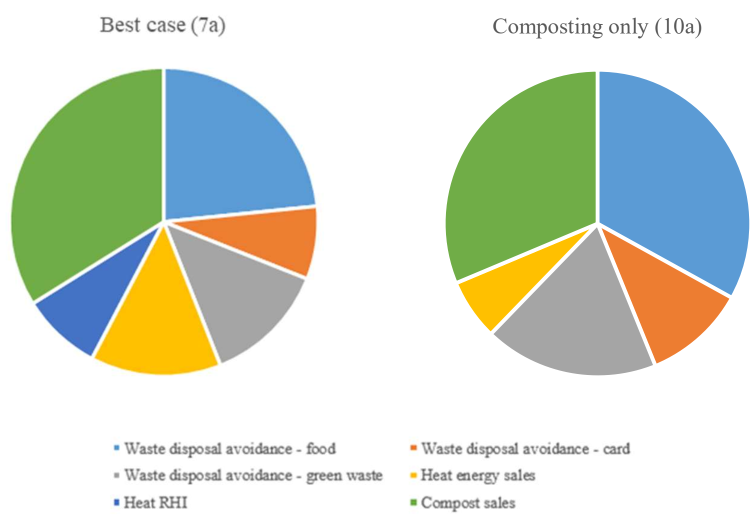


Figure 7-50: Revenue by category for the 'best-case' and composting only scenarios.

Table 7-28: Comparison of the key parameters of the 'best-case' and composting scenarios for the TEA study.

Parameter	Best case (7a)	Composting only (10a)
Food waste processed (T yr^{-1})	119.0	119.0
Vegetable oil processed (T yr^{-1})	6.0	6.0
Cardboard waste processed (T yr^{-1})	42.8	42.8
Green waste processed (T yr^{-1})	90.6	90.6
CAPEX (£ yr^{-1})	403849	304959
Yearly revenue (£ yr^{-1})	90424	88122
Simple payback time (years)	6.1	4.2
Boiler output (kW)	18.3	0.0
Compost heat output (kW)	21.3	24.8
Footprint (hectares)	1.16	0.04
Compost production (T yr^{-1})	204	267
Compost solid content (%)	53%	49%
Compost C:N ratio	30.2	33.1

Both scenarios produced a large amount of good-quality compost (that is, with a solids content of 45-60% and a C:N ratio between 25:1 and 35:1). The composting scenario relied heavily on the compost sales as a revenue, making up 46% of the total revenue, but was independent of revenue from RHIs, which might vary in the future. The composting scenario footprint was a lot smaller than that of the 'best-case' scenario. A significant advantage for the 'best-case' scenario was the heat produced – both high-grade heat from the boiler, which would have more applications, and low-grade heat from the compost system. The CAPEX and simple payback time were 24% and 31% smaller respectively for the composting only scenario, however this scenario relied on a relatively new and unproven technology, so these parameters were liable to increase or decrease.

7.10 Discussion and conclusions

7.10.1 CAPEX and OPEX

The TEA and sensitivity analysis have shown that the CAPEX of the system is not the sole factor to determine its profitability: more expensive systems can have a quicker payback if the potential 'revenue' of the system (from the energy produced, waste disposal cost avoidance, compost and renewable energy incentives) is optimised. Even when the initial CAPEX is much larger, the yearly revenue can affect the finances of the scenario to the point where the payback time is much shorter than a cheaper solution. This means that making the best possible use of the outputs, thereby optimising the revenue, is of critical importance. The TEA investigated different options within the system to use biogas, but also investigated the production of heat from composting and sale of compost, which were key to increasing the yearly revenue. The most profitable method of using the biogas was via a boiler; this is not only a relatively cheap and well understood technology but also produces a form of high grade heat (hot water) at a scale where it will be easy to use, so the highest possible benefit may be realised.

Where the yearly revenue had been optimised, the second most important factor in determining the simple payback time was the CAPEX. The TEA results showed that the CAPEX could be reduced by not including a pre-digester or a greenhouse, and this reduced the simple payback time. However, removing the pre-digester from the system could cause issues with animal by-product regulations and therefore a pre-digester is recommended to be included.

7.10.2 Cost-curve equation

A key part of the CAPEX prediction in the TEA model was the use of the cost-curve equation (Towler and Sinnott, 2012) (equation 7-1), which allowed equipment costs to be derived from equipment with the same function but of a different size. This is widely used in the process engineering industry and resulted estimated project costs that aligned well with costs of other AD projects. A sensitivity analysis of the exponent (n) used in this calculation showed that more accurate estimations were possible if the comparison equipment was close in size to the equipment for which the cost was to be estimated.

7.10.3 OPEX factor

The TEA used the CAPEX value to estimate the OPEX, using an 'OPEX factor', reflecting the fact that if a plant has more equipment and if the equipment is larger, then the operating expenditure will increase proportionally. As this was an estimate, a sensitivity analysis was performed for the OPEX factor set between 3 and 10% for the scenario that had the shortest simple payback time (without pre-digester, composting whole digestate, with either a boiler, CHP or biogas upgrader). The analysis showed that increasing the OPEX factor can significantly increase the simple payback time (increasing up to 67% by increasing from 5% to 10%). Therefore, in a 'real-world' project, this would be an important cost to calculate accurately as close as possible to the start of the project, as it will affect the project's viability.

7.10.4 Sensitivity analyses

Some factors, such as the cost of gas, electricity and fuel, and the value of government incentives, could affect the simple payback time if they change in the future. Sensitivity analysis showed that the factor that had most effect on this was a change in the RHI, which could increase the simple payback time by 19% (1 year) if it was removed entirely, or reduce the payback time by 14% (0.7 years) if it was doubled. The cost of gas and electricity had very little effect on the payback time, which reflected the fact that the amount of gas and electricity used was only a small proportion of the yearly revenues and costs. A sensitivity analysis on the parasitic energy losses showed that these similarly had only a small effect on the simple payback time compared to other factors. This was supported by the fact that adding a greenhouse increased the simple payback time rather than reducing it: the amount of CAPEX added to the project by adding a greenhouse did not compensate for the cost savings it brought about, because those savings are relatively minor compared to the savings in, for example, waste avoidance.

7.10.5 Composting

Composting technology was included in the TEA as a method of making good use of the digestate produced by the anaerobic digestion process. There were two 'alternatives' – composting whole, unseparated, digestate, and composting the separated solids from digestate (with the liquids being processed by a willow bed). It was found that neither solution produced 'ideal' results: composting whole digestate produced a low-nutrient compost, whereas the scenario for composting separated digestate solids produced a nutrient rich compost but required a large area of land for the willow bed. By changing the amounts of digestate that were separated, a compromise solution was found, in which 59% of the digestate was separated. This solution had a smaller willow bed area requirement, would process all the waste streams, and would produce a better quality compost. The simple payback time of this scenario was slightly higher than the scenario in which 100% of the whole digestate was composted (6.1 years compared to 5.7 years).

A scenario that removed the AD processing stage and used only composting was lower in simple payback time than all other scenarios due a relatively low CAPEX and relatively high revenue. This indicates that in some scenarios, for example when biogas production is not a key requirement of the system, a composting-only scenario might be preferable when processing organic waste at this scale.

7.10.6 Project viability

This TEA has shown that anaerobic digestion at a micro scale is technically feasible, and that it is most profitable when the technology is coupled with a second form of energy capture. However, the payback times are slightly outside the normal expected payback for a capital investment (3-5 years) and so the project would not be economically viable. However, an increase in the RHI or a decrease in the CAPEX (for example, by reducing the cost of the digester or composter) would make the project viable.

7.10.7 Recommendations and possible future developments

This TEA relied heavily on the relatively new technology of capturing heat from compost. To proceed with any scenario outlined in the TEA, composting with heat recovery would need to be studied in greater detail to ascertain how reliable and useable it was and how much heat and compost it would produce. A stable market for the compost would also be critical.

8 Discussion

The aim of this thesis was to investigate whether the viability of micro-scale anaerobic digestion in developed countries could be improved. The viability is based around the economics of the system, which are affected by a number of factors – for example, the cost benefit balance of the system, the value of the outputs, and whether lessons learnt in theoretical or lab-scale projects can be applied to a working plant. The individual objectives to study these factors were to investigate the effect of feeding a digester at a variable load, and to study the techno-economics of a micro-AD plant as part of a larger system based on a circular economy. A case study of a micro-scale AD plant was also conducted to gain insight into the practical operation of a plant of this type.

8.1 Experimental work on variable feeding rates

The experimental work looked at variable feeding rates in a digester, with the reasoning that being able to operate a plant at a variable load could bring about an economic advantage. For example, electricity from a CHP might be given premium rates when produced at peak times, or heat or electricity might be used on site rather than being exported, if it is produced only when it is needed, thus optimising the cost avoidance.

The experimental work employed a dual-stream automatically-fed laboratory scale anaerobic digestion system to investigate the effect of a variable feeding regime on an anaerobic digester. In the experiment, one stream was fed in a variable pattern, and the other at a steady rate, over the course of approximately five months. The stability in both digesters remained constant throughout the experiment period, as it had in previous studies of variable feed patterns (Mauky *et al.*, 2016; Laperrière *et al.*, 2017) but differences were observed in the performance.

The first effect noted was that the variable-rate digester showed an increasingly large peak in biogas production after each feed event, where the steady-rate digester response to feed events was roughly the same throughout. This was likely to be caused by an increased level of immediate breakdown of the input feed. There could have been several reasons for this – better functioning of the microbial population, an increased microbial population size, or smaller molecules in the feed. As the same feed was given to both digesters, this last explanation can be disregarded, leaving a difference arising from the microbial populations between the two digesters. To distinguish between these explanations, it is necessary to look at results from different tests.

The volatile solids (VS) content in both digesters was similar at the end of the testing period (3.9% and 4.0% in the variable-rate and steady-rate digester respectively). The VS content in a digester consists of microorganisms plus undigested feed, and a VS analysis cannot distinguish between the two. However, when a BMP analysis was performed, more methane was produced by blank digestate from the steady-rate digester compared to that from the variable-rate digester. The extra methane could have been a result of undigested feed in the steady-rate digester, either as VS or as VFAs. The extra methane is unlikely to be the result of the breakdown of dead microorganisms as this would happen more slowly. From this it is possible to conclude that the steady-rate digester had not broken down as much of the feed as the variable-rate digester.

In contrast, the digestate from the variable-rate digester produced less methane in the BMP test but responded with a larger peak of biogas production after each feed event, despite having the same VS content. As the steady-rate digester contained extra feed, as reasoned above, but the VS in both digesters was the same, it is possible to conclude that the balance VS in the variable-rate digester was ‘extra’ microorganisms.

Combining these results, it would be reasonable to conclude that there had occurred a build-up of microorganisms in the variable-rate digester compared to the steady-rate digester, and a build-up of feed had occurred in the steady-rate digester. However, to confirm this, further tests would be needed.

To verify this conclusion, an assessment could be performed of the microbial population in both digesters, in terms of species and size, by 16s rRNA testing or Most Probable Number (MPN) analysis (Cysneiros *et al.*, 2012). In previous studies, variable feeding has been found to increase the microbial population diversity (Bonk *et al.*, 2018; Mulat *et al.*, 2016a; De Vrieze, Verstraete and Boon, 2013) but the population size was hard to ascertain because of the limitations in current microbial population analysis techniques (De Vrieze, Verstraete and Boon, 2013). In which case, a method would need to be found that could accurately quantify the population size.

When the feeding rate was increased in the variable-rate digester, this was accompanied by a drop in the methane concentration in the biogas, i.e. a decrease in methane production. This drop in methane production was likely caused by an imbalance in the reaction rates within the digester – the earlier stages of digestion (hydrolysis, acidogenesis, acetogenesis) acting faster than the methanogenesis stage. However, after several weeks of feeding in a variable pattern, the decrease in methane production at increased feeding rates in this digester was less pronounced. This suggests that the digester had started to adapt to the imbalance, possibly by an increase in methanogen population, or by selective growth of microorganisms that were able to thrive in conditions with variable feed rates. This could be further investigated by analysis of the mix of species in the microbial population as mentioned previously.

8.2 Case study

The case study presented a micro-scale AD plant in London that was operating a 2 m^3 digester. The site was monitored for 319 days, with a throughput of 5.1 tonnes yr^{-1} of organic waste, producing 228 m^3 biogas $tonne^{-1}$ with an average methane content of 60.6%.

The case study showed that the feed to the digester could be varied considerably without affecting its stability, and that the pre-digester tank was an important addition to the system to blend the feedstocks being added so that these variations could be minimised. The study monitored the VFA, alkalinity, pH, biogas production and biogas methane concentration. Towards the end of the monitoring period, a rise in VFA was noted, accompanied by a rise in ammonia concentration in the digester. This issue was successfully resolved by adding trace elements to the feed, as recommended by previous research of food waste anaerobic digesters (Banks *et al.*, 2012). Compared to a large scale plant (Banks *et al.*, 2011), the micro-scale AD plant achieved a higher specific biogas production, a similar volumetric yield (biogas per volume of digester) and a lower biogas methane content.

The site energy use was monitored and found parasitic energy and heat use of 31.7% and 18.0% respectively, which are both high compared to other plants (table 7-21). However, the parasitic heat was found to have been reduced by 49% by the plant being built in a greenhouse. This feature increased the thermal coefficient of performance from 2.72 to 5.55.

The economics of the plant showed a very long simple payback time (148 years), which was partly due to the lower than expected gate fees drawn as revenue. The total capital cost was £28,035, 29% higher than predicted, due to a higher than expected cost to set up a system for gas use on site. The site did not draw any revenue from sales of digestate, which could be a potentially valuable output if a suitable market could be identified.

The operators of the site reported that a lack of space and excessive odour were both key issues that should be addressed in future projects. They noted that the output of digestate was problematic because despite having potential as a fertiliser, the market for it was limited in an urban environment.

8.3 Techno-economic analysis

The TEA generated an array of scenarios of a micro-scale AD system based in a university. The feed input of all instances was 326 kg day^{-1} (119 TPA) of food waste and waste vegetable oil, and the potential CHP output was 5.6 kWe at 30% electrical efficiency and 8.4 kWth at 45% thermal efficiency.

The scenarios explored in the TPA varied in the technology used to process the biogas (boiler, CHP, upgrading for vehicle fuel), the details of the composting process (whole or separated digestate used, co-composting with waste cardboard and green waste), and inclusion/exclusion of optional items (a separator, a willow bed and a greenhouse as a plant housing).

The simple payback time (SPT) was determined to be the best target for comparison as it could be used to compare different forms of energy output and was currency agnostic. The SPT was calculated for each instance, with sensitivity analyses to determine the important factors that could affect the plant's economics. The instance with the shortest SPT (5.4 years) had the lowest CAPEX through omission of a pre-digester, greenhouse and separator and by composting whole, rather than separated, digestate. As the process was handling food waste however, including a pre-digester to pasteurise the feedstock was a necessity. The solution with the shortest SPT that satisfied this requirement included a boiler, pre-digester and a composter for whole digestate, but omitted a greenhouse (5.84 years). A greenhouse or similar building to house the plant was found to reduce energy usage, but had a negative net cost benefit, due to the cost of the building itself.

A biogas boiler proved to be economically the best option for biogas use compared to CHP or upgrading to biomethane. However, the payback difference between the boiler and CHP was marginal, and if the price of electricity purchase increased, then a CHP would become the better option, as it would bring larger benefits in terms of avoidance of utilities expenditure. A key consideration is the usage of the output (heat, electricity, biomethane), and any external costs that might be incurred by setting up a system for the usage of the output. For example, a district heating system or an electrical connection might be required, or the organisation might need to purchase a fleet of CNG vans for biomethane use. From this point of view, combustion of the biogas in a boiler would require the lowest CAPEX investment. It was also considered that the volume of biomethane that would be produced by this system (the equivalent of 126,000 fuel miles) could be in excess of the amount that could feasibly be used by the university.

A significant addition to the system was a composter, which would generate low-grade heat from composting and would provide a safe and reliable method of using the digestate produced from the anaerobic digestion process. The scenarios compared composting whole digestate, composting separated digestate, and composting a combination of the two, with all instances including different amounts of the other available organic waste streams (shredded cardboard, garden waste). The most cost-effective solution was to compost whole digestate, and this also resulted in a smaller plant footprint. A scenario with no composting system had a slightly shorter simple payback time (5.79 years compared to 5.84 years), but this might cause a problem with digestate disposal, as identified by the case study in chapter 5. A final scenario tested a system with only a composting system and no anaerobic digester and resulted in the shortest SPT from all scenarios (4.22 years).

The TEA showed that the yearly revenue and CAPEX both had a significant effect on the overall cost, but that a project with a lower CAPEX would not necessarily have a quicker payback time. This has implications with government funding, both in terms of incentives such as RHI and the provision of large capital funds, the expansion of which could both stimulate growth in this field.

8.4 Further work

8.4.1 Expansion of the variable feed pattern experimental work

The variable feed pattern in digester 1 was set to an average OLR of 2 gVS L^{-1} day^{-1}, which is a low to medium load. This OLR was chosen as the basis for the experimental study as it was well within the range of OLR that an anaerobic digester would easily be able to cope with, and therefore any signs of instability in the digester fed at a variable rate would show up in contrast. The results of this study showed that the digester seemed to adapt to the variable feed pattern by exhibiting a quicker response to overload but did not show any difference in stability. In order to investigate this effect further, the experiment could be repeated at a high OLR, for example 6 gVS L^{-1} day^{-1}. This would be more likely to destabilise the digesters and might show if one digester is more stable than the other. If the variable-rate digester had adapted to higher feed rates, it would remain more stable than the steady-rate digester. Alternatively, the OLR for both digesters could be raised each week until the digesters fail – the hypothesis being that the digester that had been acclimatised with the variable feeding rate would be able to withstand a higher steady feeding rate.

The data that should be recorded is the same as was recorded in this experimental work. The biogas flow and methane content of the biogas would be used to show the response of the digester microbial consortium to the feeding/overload incidents, and the alkalinity, VFA levels and ammonia measurements would be used to monitor the digester's stability. If the hypothesised mechanism is correct (that the population of microorganisms increased or otherwise changed because of the variable pattern), the expected outcome would be that a steady-feed digester would become unstable at higher OLRs, but a variable-rate digester would not.

8.4.2 VFA testing

The variable-rate digester appeared to show an adaptation to the changing feed, with an increased biogas production rate after each feed event. This could have been due to an increased microorganism population or an adaptation of the microorganism population to favour certain species. To study this effect more closely, the VFA levels after feed events could be studied through VFA GC analysis (in the original study, samples were taken before, rather than after, feed events). This would highlight the build-up of process intermediates through increased levels of different VFAs and might thereby allow the dynamics of the anaerobic digestion system to be studied.

The increased biogas production in the digesters in response to feed events in this study was approximately 5 minutes, therefore sampling would need to be performed in this time frame. The experiment could be repeated firstly at the same loading rates and loading patterns, with digestate samples taken 10 minutes before, then 5 minutes and 10 minutes after a feeding event. The sampling and sample preparation is work intensive and would need two researchers to manage the process. If the process could be automated, this would be advantageous.

The expected results to support the work in this thesis would be that in the first weeks under a variable feed pattern, the digester would show large spikes in VFA levels after a feeding event, and these spikes would become smaller as the experiment continued. Compared to the steady feed digester, the variable-feed digester would show smaller VFA level spikes after a feed event when fed at the same OLR.

8.4.3 Composting system

The TEA study showed that a composting system could improve the economics of an anaerobic digestion plant, but only if the composting system was set up with heat extraction and there was a local application for the heat. This type of composting system technology is not currently developed commercially but might present a marked improvement in the processing of organic waste in terms of practicality and cost-benefit, particularly at the scale studied in this thesis. Further investigation into this technology would be beneficial to support commercial development, determine whether the production of heat was useable, and look further into the design and economics. The scale of composting systems with heat recovery is important, with larger systems producing more heat per kg of organic matter than smaller ones (Smith, Aber and Rynk, 2017). Experimental work would be most effective and applicable to larger plants if it was conducted at least at pilot scale – for example, a size of over 1m^3.

The system would need to test the composting time and the heat produced by the process. Firstly, a 'pilot plant' composter would be constructed, consisting of a highly insulated container that can be injected with air from below, with a heat exchanger and temperature gauges in the roof space. Using different mixtures of compostable material, the starting dry solids and carbon:nitrogen ratio of the mixture would be calculated, then samples of the compost would be taken over time to determine how long the composting takes. Stability of the compost would be determined by measuring the temperature and changing moisture content - when these stop changing then the compost is stable, therefore finished. The final dry solids and carbon:nitrogen ratio of the compost would then be measured.

8.5 Exploring the broader context

This thesis found that when considered from a purely economic point of view, micro-scale anaerobic digestion is not financially attractive. This is supported by previous studies (Walker *et al.*, 2017; BRE/WRAP, 2013) that found that the payback time and levelized cost of electricity were higher for a micro-scale project than a large-scale project. The levelized cost of electricity for large-scale AD projects is competitive to other renewable energy systems (Arup, 2016).

There are, however, factors that make micro-scale AD attractive in other ways and could become more important in the future. Micro-scale AD is a form of distributed energy, which supplies at a local level, reducing capacity requirements on the national grid. The technology required to facilitate this, micro-scale biogas CHP, is newly available commercially (The Renewable Energy Hub UK, 2018; Helec, 2020). When considering distributed sources of organic waste, the transport reduction enabled by processing waste locally can reduce traffic, thus reducing capacity requirement on the traffic system and reducing carbon emissions. A micro-scale AD plant can be made accessible (in a controlled way) to the general public, who are often both the end users of the product and the providers of the feedstock. This provides an opportunity to engage this audience as stakeholders to make the most of resources, provide an education opportunity, and encourage an awareness of local resilience and the circular economy. In a best-case scenario, promotion of the circular economy concept in this way could help to show the value of organic waste as a source of nutrients and a soil conditioner as well as a source of energy. This could lead to the capture of marginal waste streams that would otherwise be lost, by enabling consumers to recognise the value of organic waste and establish ways to separate it at source.

The benefits of micro-scale AD described above are generally issues that are important at the national scale rather than for individual projects – education, environment and loading of the electricity and traffic networks. In order to realise these benefits, initiatives on a national scale would be required as they could be specifically targeted at these aims. The TEA found that renewable heat incentives (RHIs) had significant potential to lower the payback time of micro-scale AD projects, however there is no separate size category of RHIs for micro-scale AD (GOV.UK, 2019b) despite the difference in payback time. The RHIs are also not applicable to heat produced from composting (OFGEM, personal communication), which is also an effective technology at micro-scale. There is currently no premium in place for the production of organic fertiliser, although in the UK there are funding schemes in place for organic farming (GOV.UK, 2016), and a biofertilizer certification scheme (Renewable Energy Assurance Ltd, 2020).

Another incentive that could alter the economics of AD at all scales is the introduction of higher prices for electricity delivered at peak times (Hahn *et al.*, 2014a), which again could be a national initiative. AD can be a 'flexible' source of energy and can therefore be controlled to produce premium energy at peak times. This thesis investigated the effect of variable feeding of food waste and found that the stability of the plant was unaffected, which is supported by previous research (Mauky *et al.*, 2015; De Vrieze, Verstraete and Boon, 2013). The variation in feed appeared to improve the volatile solids destruction rate compared to a steady feed, but the response to a changing feed rate occurred over several days – this time frame has been observed in other studies (Laperrière *et al.*, 2017). A flexible feeding system would therefore be possible without making the plant unstable, and may even improve a plant's efficiency, but would need to be pre-planned to fit an expected daily pattern. Alternatively, a more easily digestible feedstock would be required to produce a quicker change in biogas production (Hahn *et al.*, 2014b).

The profitability of a micro-scale AD plant also requires it to be designed to make best use of the features of the site, and the site's general suitability. Remote sites for example will benefit more than well-connected ones by a reduced need for transport, local waste disposal and local fertiliser production – a form of 'local resilience', which is an important aspect of the circular economy. Urban sites may benefit from the reduced load on the national grid and on the transport network. Any plant would need to have a use for the products of the process – heat, electricity, fertiliser or biomethane – in order to make it profitable and effective.

As part of the TEA, this thesis considered composting as both a post-process for anaerobic digestion and as an alternative to anaerobic digestion. As a post-process for AD, composting can increase the C:N ratio in the product from roughly 3:1 to 30:1 and make it more stable by increasing the humus content (Jane Gilbert, Marco Ricci-Jürgensen and Ramola, 2020), both of which would make it a better soil conditioner. The disposal or use of digestate from micro-scale anaerobic digestion can be problematic (Walker *et al.*, 2017) and the transformation into compost could solve this problem. Additionally, compost could be marketed to a wider range of consumers as it could be spread on smallholdings, gardens, allotments and municipal parks as well as farmland.

As an alternative to AD, this TEA has shown that composting at this scale is more cost-effective, with an equivalent or smaller footprint. The composting process is simpler, would require less specialized expertise, and would encompass less risk to safety and the environment because it does not produce methane. As an educational tool and a facility that provides local waste handling, the two systems are equivalent. The principal disadvantage in composting compared to AD is that the process generates low-grade heat, up to a maximum of 65°C (Smith, Aber and Rynk, 2017). This is a form of energy which is more limited in its uses than biogas, and so would not be as valuable a resource for replacing fossil fuels and thus reducing carbon emissions. However, if a location has an established use for this heat, for example heating buildings or a swimming pool, this might provide the right conditions for a profitable project. The location of a project that includes composting is therefore key to its success. Composting with heat recovery is a relatively new technology, but open-air composting (in windrows) is established, as is anaerobic digestion.

In summary, micro-scale anaerobic digestion can be an asset that brings about numerous benefits. In the future, recommendations to promote this technology through legislation would be to create a separate, higher subsidy for heat (and electricity) from plants at this scale, and to recognise the potential environmental benefits through premiums for organic fertilisers. At the plant level, planners should carefully assess the plant site to ensure that the maximum benefits from the output can be realised.

9 Conclusion

The average size of an anaerobic digestion facility in the UK is about 500 kilowatts, with a capacity of up to 3000 m^3, using over 15 tonnes of feedstock per day. Typical feedstocks are food waste, energy crops, cow slurry and sewage. This size of operation has a large transport requirement and is inaccessible to the general public.

Micro-scale anaerobic digestion (under 50 kW) has several advantages over larger scale operations. It can minimize transport, it can capture 'marginal' waste streams that would otherwise be landfilled, it can promote the local economy by providing jobs, and it can promote community involvement and education in green issues. However, micro-scale anaerobic digestion is less economically viable than large-scale, due to an economy of scale, and so is not widely implemented – although examples already exist.

This thesis investigated the viability of micro-scale anaerobic digestion in developed countries, in terms of both its economics and practical factors such as the best use of the products from the process.

The experimental part of the project studied the effect of varying the organic load to the digester, in terms of its biogas production and stability, compared to that of a digester fed at a steady load. Over the course of five months, two digesters were fed at the same average load, 2 gVS L^{-1} day^{-1}, however one was fed at a steady load, and the other was fed with a variable load for part of the experimental period. Both digesters remained stable, but the digester under a variable load showed an increasingly large peak in biogas production after each feed event. The steady-rate digester showed a build-up of undigested feed compared to the variable-rate digester, and the variable-rate digester showed a possible change in microbial population, either in the species densities or the overall population density.

A case study of a micro-scale anaerobic digestion project in London, UK demonstrated the stable operation of a digester running on food waste for over a year. A potential problem with a build-up in volatile fatty acids and ammonia was resolved by the addition of trace elements. The plant had a very long simple payback time of 148 years, which was attributable to a lower than expected yearly gate fee revenue and a higher than expected CAPEX expenditure to set up a gas usage system on site. The operators noted the main operational issues to be excessive odour, a lack of space, and difficulties in disposing of the digestate output.

A techno-economic analysis was performed to determine whether additions or changes could be made to micro-scale anaerobic digestion projects to improve their financial viability. The scenarios explored in the analysis were varied by the technology used to process the biogas (boiler, CHP or upgrading for vehicle fuel) and inclusion/exclusion of optional items (a separator, a willow bed and a greenhouse as a plant housing). Composting was added as a post-processing mechanism to make use of the digestate and transform it into a more stable and useful product. The simple payback time was used as the main basis for comparison between the scenarios. The study found that a boiler was economically the best option for biogas use at this scale, but that the difference between simple payback time for projects using a boiler and using a CHP was marginal, and if the value of electricity increased, the CHP would become the better option. It was noted that a pasteurisation step in the form of a pre-digester was required to satisfy food waste handling regulations. A sensitivity analysis showed that the simple payback time was most affected by the value of the Renewable Heat Incentive and the value of compost. Overall, the simple payback time was more affected by the yearly revenue rather than the initial CAPEX investment.

The economic viability of micro-scale anaerobic digestion could therefore be improved by additional processing of the digestate into compost, and the development of government initiatives such as Renewable Heat Incentives specifically for plants under 50 kW. Variable biogas production could be employed by plant operators to produce electricity on demand without risk to plant stability, and this could be incentivised by the guarantee of higher rates for electricity produced at peak times. The practical viability of micro-scale anaerobic digestion can also be improved by the production of compost from digestate, and the development of heat recovery composting technology would help to achieve this.

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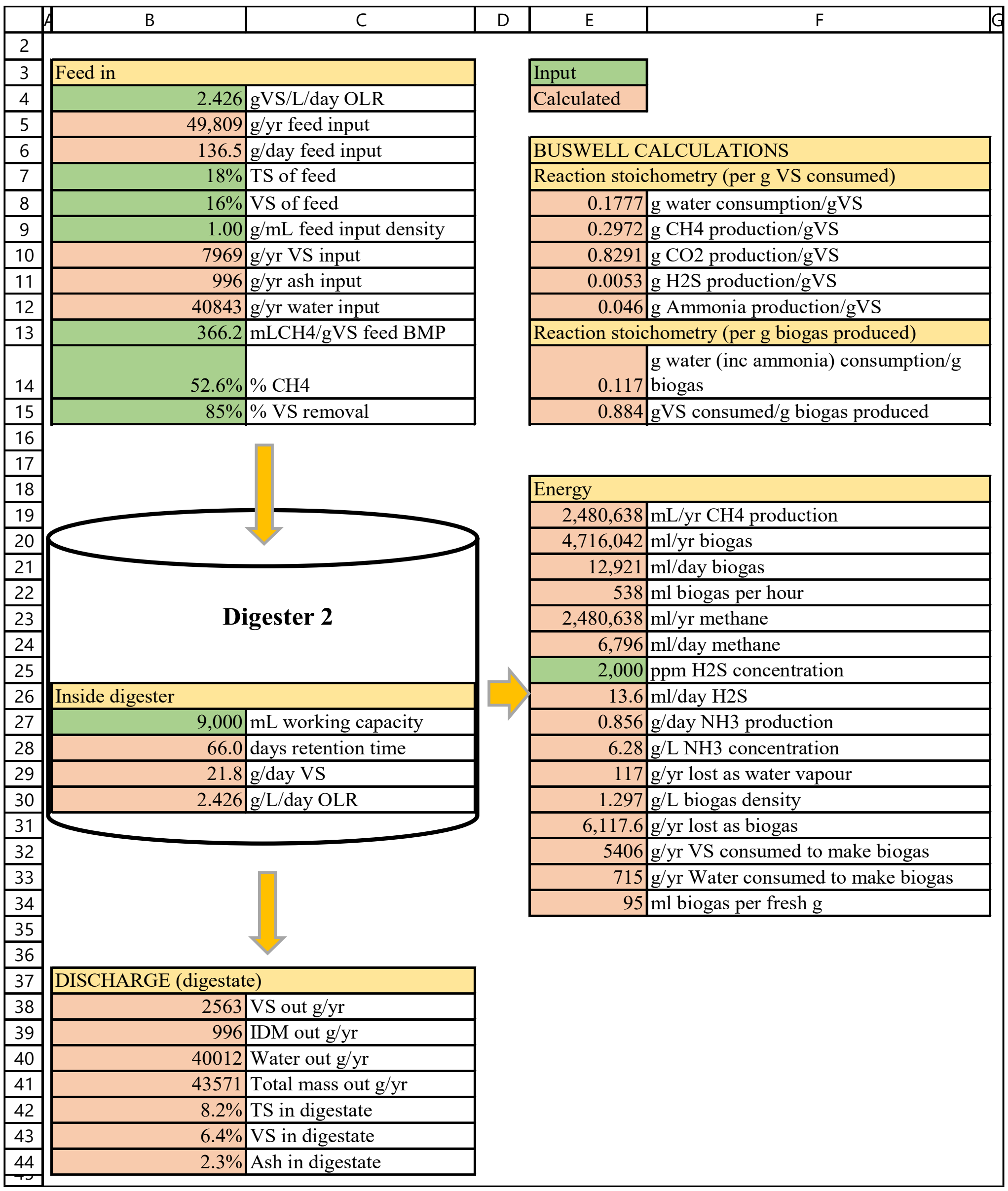
Appendix A: Mass balance calculations

A.1 Laboratory mass balance

A mass balance for the experimental work was created as a spreadsheet in Microsoft Excel. The inputs were the data assumptions and measured experimental values, and the outputs were the predicted key characteristics of the system, such as final TS and VS of the digestate and the biogas production rate (figure A-1). The mass balance assumed the use of a continuously-stirred single-tank reactor (CSTR).



Feed in	
2.426	gVS/L/day OLR
49,809	g/yr feed input
136.5	g/day feed input
18%	TS of feed
16%	VS of feed
1.00	g/mL feed input density
7969	g/yr VS input
996	g/yr ash input
40843	g/yr water input
366.2	mLCH4/gVS feed BMP
52.6%	% CH4
85%	% VS removal

Digester 2	
Inside digester	
9,000	mL working capacity
66.0	days retention time
21.8	g/day VS
2.426	g/L/day OLR

DISCHARGE (digestate)	
2563	VS out g/yr
996	IDM out g/yr
40012	Water out g/yr
43571	Total mass out g/yr
8.2%	TS in digestate
6.4%	VS in digestate
2.3%	Ash in digestate

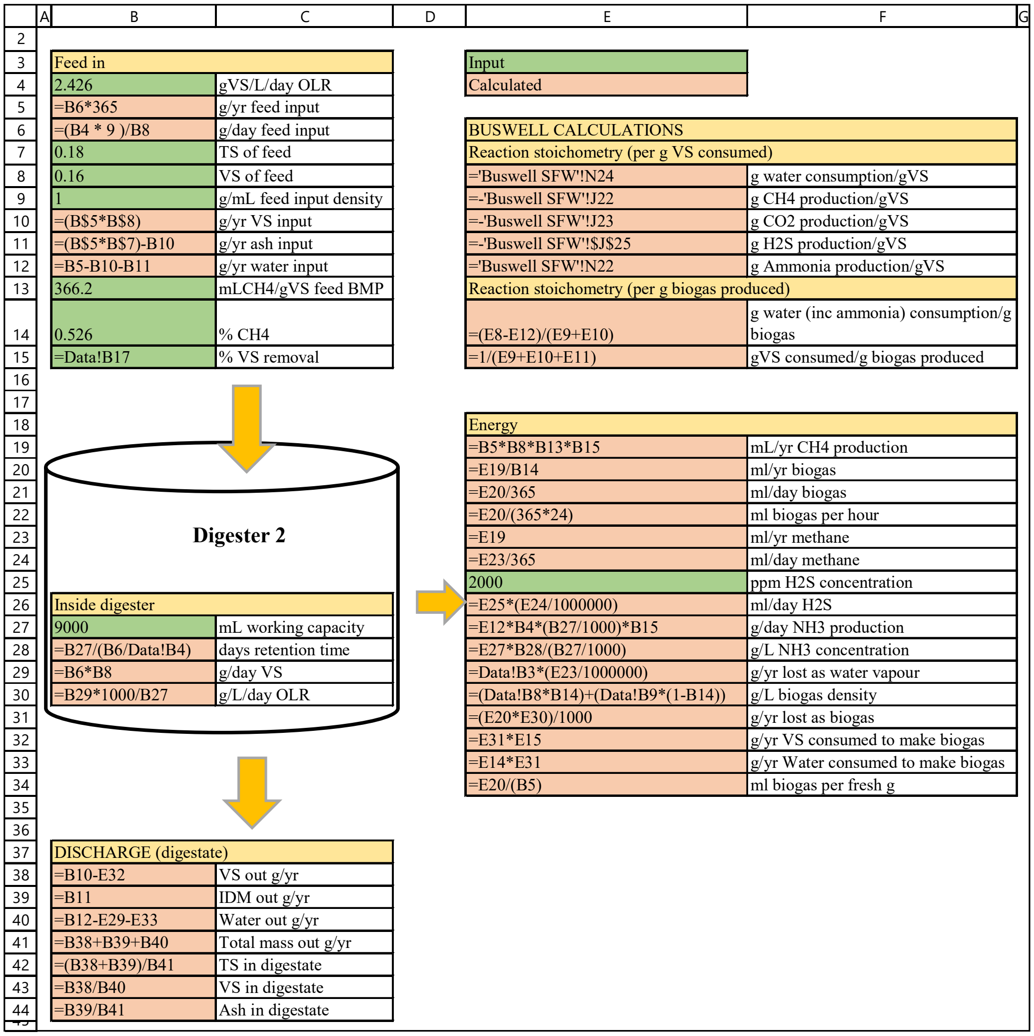
Input
Calculated

BUSWELL CALCULATIONS	
Reaction stoichometry (per g VS consumed)	
0.1777	g water consumption/gVS
0.2972	g CH4 production/gVS
0.8291	g CO2 production/gVS
0.0053	g H2S production/gVS
0.046	g Ammonia production/gVS
Reaction stoichometry (per g biogas produced)	
0.117	g water (inc ammonia) consumption/g biogas
0.884	gVS consumed/g biogas produced

Energy	
2,480,638	mL/yr CH4 production
4,716,042	ml/yr biogas
12,921	ml/day biogas
538	ml biogas per hour
2,480,638	ml/yr methane
6,796	ml/day methane
2,000	ppm H2S concentration
13.6	ml/day H2S
0.856	g/day NH3 production
6.28	g/L NH3 concentration
117	g/yr lost as water vapour
1.297	g/L biogas density
6,117.6	g/yr lost as biogas
5406	g/yr VS consumed to make biogas
715	g/yr Water consumed to make biogas
95	ml biogas per fresh g

Figure A-1: Mass balance for digester 2, in Microsoft Excel.

The formulas used for each calculation in the laboratory mass balance are shown in figure A-2.



	B	C
3	Feed in	
4	2.426	gVS/L/day OLR
5	=B6*365	g/yr feed input
6	=(B4 * 9)/B8	g/day feed input
7	0.18	TS of feed
8	0.16	VS of feed
9	1	g/mL feed input density
10	=(B$5*B$8)	g/yr VS input
11	=(B$5*B$7)-B10	g/yr ash input
12	=B5-B10-B11	g/yr water input
13	366.2	mLCH4/gVS feed BMP
14	0.526	% CH4
15	=Data!B17	% VS removal
23	Digester 2	
26	Inside digester	
27	9000	mL working capacity
28	=B27/(B6/Data!B4)	days retention time
29	=B6*B8	g/day VS
30	=B29*1000/B27	g/L/day OLR
37	DISCHARGE (digestate)	
38	=B10-E32	VS out g/yr
39	=B11	IDM out g/yr
40	=B12-E29-E33	Water out g/yr
41	=B38+B39+B40	Total mass out g/yr
42	=(B38+B39)/B41	TS in digestate
43	=B38/B40	VS in digestate
44	=B39/B41	Ash in digestate

	E	F
3	Input	
4	Calculated	
6	BUSWELL CALCULATIONS	
7	Reaction stoichometry (per g VS consumed)	
8	='Buswell SFW'!N24	g water consumption/gVS
9	=-'Buswell SFW'!J22	g CH4 production/gVS
10	=-'Buswell SFW'!J23	g CO2 production/gVS
11	=-'Buswell SFW'!J25	g H2S production/gVS
12	='Buswell SFW'!N22	g Ammonia production/gVS
13	Reaction stoichometry (per g biogas produced)	
14	=(E8-E12)/(E9+E10)	g water (inc ammonia) consumption/g biogas
15	=1/(E9+E10+E11)	gVS consumed/g biogas produced
18	Energy	
19	=B5*B8*B13*B15	mL/yr CH4 production
20	=E19/B14	ml/yr biogas
21	=E20/365	ml/day biogas
22	=E20/(365*24)	ml biogas per hour
23	=E19	ml/yr methane
24	=E23/365	ml/day methane
25	2000	ppm H2S concentration
26	=E25*(E24/1000000)	ml/day H2S
27	=E12*B4*(B27/1000)*B15	g/day NH3 production
28	=E27*B28/(B27/1000)	g/L NH3 concentration
29	=Data!B3*(E23/1000000)	g/yr lost as water vapour
30	=(Data!B8*B14)+(Data!B9*(1-B14))	g/L biogas density
31	=(E20*E30)/1000	g/yr lost as biogas
32	=E31*E15	g/yr VS consumed to make biogas
33	=E14*E31	g/yr Water consumed to make biogas
34	=E20/(B5)	ml biogas per fresh g

Figure A-2: Mass balance (with formulas shown) for digester 2, in Microsoft Excel.

The following equations were used:

$$Q_{CH4} = \frac{\dot{m}_{fw} \times VS_{fw} \times BMP}{365 \times 24} \qquad \textit{Equation A-1}$$

Where: Q_{CH4} is the predicted methane production rate in mL hr^{-1}.

$\dot{m}_{fw}$ is the mass of feed input, in g year^{-1}.

VS_{fw} is the volatile solids content of the food waste by mass, in %.

BMP is the biological methane potential in mL methane gVS^{-1}.

$$\rho_{bg} = (\rho_{CH4} \times P_{bg.CH4}) + (\rho_{CO2} \times P_{bg.CO2})$$ *Equation A-2*

Where: ρ_{bg} is the biogas density in g mL^{-1}.

ρ_{CH4} is the density of methane in g mL^{-1}.

$P_{bg.CH4}$ is the proportion of methane in the biogas, in %.

ρ_{CO2} is the density of carbon dioxide in g mL^{-1}.

$P_{bg.CO2}$ is the proportion of carbon dioxide in the biogas, in %.

$$\dot{m}_{bg} = \frac{Q_{CH4}}{P_{bg.CH4}} \times \rho_{bg}$$ *Equation A-3*

Where: $\dot{m}_{bg}$ is the mass of VS lost as biogas, in g $year^{-1}$.

Q_{CH4} is the methane production in mL $year^{-1}$.

$P_{bg.CH4}$ is the proportion of methane in the biogas, in %.

ρ_{bg} is the density of biogas in g mL^{-1}.

$$\dot{m}_{bg.H2O} = C_{H2O} \times Q_{bg}$$ *Equation A-4*

Where: $\dot{m}_{bg.H20}$ is the mass of water vapour lost in the biogas, in g $year^{-1}$.

C_{H2O} is the water vapour content of the biogas, in g m^{-3}.

Q_{bg} is volume of biogas, in m^3 $year^{-1}$.

$$R_{VS} = \frac{\dot{m}_{bg}}{\dot{m}_{fw} \times VS_{fw}}$$ *Equation A-5*

Where: R_{VS} is the volatile solids removal rate, or the proportion of the input volatile solids that are converted by the anaerobic digestion process, in % .

$\dot{m}_{bg}$ is the mass of volatile solids lost as biogas, in g $year^{-1}$.

$\dot{m}_{fw}$ is the mass of food waste input, in g $year^{-1}$.

VS_{fw} is the volatile solids content of the food waste by mass, in %.

$$\dot{m}_{dig.TS} = \dot{m}_{fw.TS} - \dot{m}_{bg}$$ *Equation A-6*

Where: $\dot{m}_{dig.TS}$ is the amount of total solids that leaves the digester in the digestate, in g $year^{-1}$.

$\dot{m}_{fw.TS}$ is the mass of total solids that enters the digester in the food waste, in g $year^{-1}$.

$\dot{m}_{bg}$ is the mass of volatile solids lost as biogas, in g $year^{-1}$.

$$\dot{m}_{dig.VS} = \dot{m}_{fw.VS} - \dot{m}_{bg}$$ *Equation A-7*

Where: $\dot{m}_{dig.VS}$ is the mass of volatile solids that leaves the digester in the digestate, in g year^{-1}.

$\dot{m}_{fw.VS}$ is the mass of volatile solids that enters the digester in the food waste, in g year^{-1}.

$\dot{m}_{bg}$ is the mass of volatile solids lost as biogas, in g year^{-1}.

$$\dot{m}_{dig.H2O} = \dot{m}_{fw.H2O} + \dot{m}_{bg.H2O}$$ *Equation A-8*

Where: $\dot{m}_{dig.H2O}$ is the mass of water that leaves the digester in the digestate, in g year^{-1}.

$\dot{m}_{fw.H2O}$ is the mass of water that enters the digester in the food waste, in g year^{-1}.

$\dot{m}_{bg.H2O}$ is the mass of water vapour lost in the biogas, in g year^{-1}.

$$\dot{m}_{dig} = \dot{m}_{dig.H2O} + \dot{m}_{dig.TS}$$ *Equation A-9*

Where: $\dot{m}_{dig}$ is the mass of digestate that leaves the digester, in g year^{-1}.

$\dot{m}_{dig.H2O}$ is the mass of water that leaves the digester in the digestate, in g year^{-1}.

$\dot{m}_{dig.TS}$ is the mass of total solids that leaves the digester in the digestate, in g year^{-1}.

$$TS_{dig} = \frac{\dot{m}_{dig.TS}}{\dot{m}_{dig}}$$ *Equation A-10*

Where: TS_{dig} is the proportion of total solids in the digestate, in %.

$\dot{m}_{dig.TS}$ is the mass of total solids that leaves the digester in the digestate, in g year^{-1}.

$\dot{m}_{dig}$ is the mass of digestate that leaves the digester, in g year^{-1}.

$$VS_{dig} = \frac{\dot{m}_{dig.VS}}{\dot{m}_{dig}}$$ *Equation A-11*

Where: VS_{dig} is the proportion of volatile solids in the digestate, in %.

$\dot{m}_{dig.VS}$ is the mass of volatile solids that leaves the digester in the digestate, in g year^{-1}.

$\dot{m}_{dig}$ is the mass of digestate that leaves the digester, in g year^{-1}.

$$HRT = \frac{V_{dig} \times \rho_{fw} \times 365}{\dot{m}_{fw}}$$ *Equation A-12*

Where: HRT is the hydraulic retention time of the digester, in days.

V_{dig} is the working volume of the digester, in mL.

ρ_{fw} is the density of the food waste, in g mL^{-1}.

$\dot{m}_{fw}$ is the mass of food waste input, in g $year^{-1}$.

$$NH3_{dig} = NH3_{Buswell} \times OLR \times VS_{dest} \times HRT$$ *Equation A-13*

Where: $NH3_{dig}$ is the ammonia concentration in the digester, in g L^{-1}.

$NH3_{Buswell}$ is the production rate of ammonia, calculated by the Buswell equation, in gNH3 gVS^{-1}.

OLR is the organic loading rate, in gVS L^{-1} day^{-1}.

VS_{dest} is the expected proportion of VS destruction, in %.

HRT is the hydraulic retention time of the digester, in days.

Inputs

The measured inputs used were the TS (%), VS (%), measured feed loading rate (OLR, in gVS L^{-1} day^{-1}), average methane concentration in the biogas (%) and BMP determined in the experimental work (LCH_4 gVS^{-1}).

Data assumptions

A number of data assumptions were used (table A-1). The densities used for methane and carbon dioxide were at standard temperature and pressure (0°C, 1 atm) as the biogas measurements were corrected to STP by the flowmeters for both the digesters and the BMP test equipment. A range of % VS removal values was reported in the literature for food waste so a mid-range value was assumed.

Table A-1: Data assumptions used in the mass balance.

Data	Unit	Value	Source	Cell ref. in formulas
Water vapour content	g m^{-3}	47	(Scott and Turra, 2019)	Data!B3
Feed density	kg/m^3	1	(WRAP, 2010)	Data!B4
Methane density	kg/m^3	0.708	(Engineering ToolBox, 2018b)	Data!B8
CO_2 density	kg/m^3	1.951	(Engineering ToolBox, 2018a)	Data!B9
VS removal	%	85	(Paritosh *et al.*, 2017) and (Banks, 2009)	Data!B17

Buswell equation

The relative proportions of methane and carbon dioxide in biogas change depending on the CHONS composition of the feedstock, which affects the energy production (Curry and Pillay, 2012). The Buswell equation can be used to calculate the amount of methane and carbon dioxide produced, and the amount of water used, when a known mass of VS is decomposed by anaerobic digestion (Buswell and Mueller, 1952). The equation (equation A-14) uses the known proportions of carbon, hydrogen, oxygen, nitrogen and sulphur in a feedstock (determined using a CHNS test) to calculate the stoichiometry of the feed decomposition into biogas. In this way the relative molar ratio for each of the substances in the equation can be determined.

$$C_cH_hO_oN_nS_s + \left(c - \frac{h}{4} - \frac{o}{2} + 3\frac{n}{4} + \frac{s}{2}\right)H_2O \rightarrow \left(\frac{c}{2} - \frac{h}{8} + \frac{o}{4} + 3\frac{n}{8} + \frac{s}{4}\right)CO_2 + \left(\frac{c}{2} + \frac{h}{8} - \frac{o}{4} - 3\frac{n}{8} - \frac{s}{4}\right)CH_4 + nNH_3 + sH_2S$$

Equation A-14

The subscripts c, h, o, n, and s denote the % weight of carbon, hydrogen, oxygen, nitrogen and sulphur in the sample, determined by CHNS analysis. The molar ratio of each element was calculated using the values measured experimentally (equation a-15, table a-2).

$$Molar\ ratio = \frac{element\ \%\ of\ sample\ by\ weight}{element\ molecular\ weight\ \times 100}$$ *Equation A-15*

$$Moles\ of\ H_2O\ consumed = \left(c - \frac{h}{4} - \frac{o}{2} + 3\frac{n}{4} + \frac{s}{2}\right)$$ *Equation A-16*

$$Moles\ of\ CO_2\ produced = \left(\frac{c}{2} - \frac{h}{8} + \frac{o}{4} + 3\frac{n}{8} + \frac{s}{4}\right)$$ *Equation A-17*

$$Moles\ of\ CH_4\ produced = \left(\frac{c}{2} + \frac{h}{8} - \frac{o}{4} - 3\frac{n}{8} - \frac{s}{4}\right)$$ *Equation A-18*

Table A-2: Calculation of the sample molar ratio for synthetic food waste.

Element	N	C	S	H	O
% of sample (by weight)	3.8	44.9	0.5	6.3	44.5
Molecular weight	14	12	32	1	16
Molar ratio (no.of moles in 1g sample)	0.002714	0.037417	0.000156	0.063	0.027813

The stoichiometry of the decomposition of SFW was calculated (table A-3).

Table A-3: Stoichiometry for synthetic food waste decomposition using the Buswell equation.

		Biomass sample	**H_2O consumed**	**CO_2 produced**	**CH_4 produced**	**Ammonia produced**	**H_2S produced**
Molar ratio	Mol	-	0.0099	0.0188	0.0186	0.0027	0.0002
Mass	g	1	0.1777	0.8291	0.2972	0.0461	0.0053
Volume (gases)	L	-	-	0.4221	0.4235	0.0608	0.0035

From these calculations, the weight of biogas produced per gram of biomass VS could be calculated (equation A-19), which could be used in the mass balance. This assumes 100% breakdown of VS, which would not be the case for food waste. For this mass balance, an initial value of 85% VS breakdown was used, which is a normal value for food waste (Banks, 2009; Paritosh *et al.*, 2017).

$$Biogas\ mass\ per\ gram\ of\ biomass\ VS\ (in\ g) = CO_2\ mass + CH_4\ mass + H_2S\ mass$$

Equation A-19

A.2 TEA mass balance

For the techno-economic analysis, a second mass balance was constructed, with additional calculations due to the use of two different feedstocks (equation A-20 to A-25).

$$\dot{VS}_{tot} = (\dot{m}_{fw} \times VS_{fw}) + (\dot{m}_{gw} \times VS_{gw})$$

Equation A-20

Where: $\dot{VS}_{tot}$ is the total volatile solids input into the digester, in tonneVS yr^{-1}.

$\dot{m}_{fw}$ is the mass input of food waste, in tonnes yr^{-1}.

VS_{fw} is the volatile solids content of the food waste, in %.

$\dot{m}_{gw}$ is the mass input of green waste, in tonnes yr^{-1}.

VS_{gw} is the volatile solids content of the green waste, in %.

$$BMP_{tot} = \left(\frac{VS_{fw} \times \dot{m}_{fw} \times BMP_{fw}}{\dot{VS}_{tot}}\right) + \left(\frac{VS_{gw} \times \dot{m}_{gw} \times BMP_{gw}}{\dot{VS}_{tot}}\right)$$ *Equation A-21*

Where: BMP_{tot} is the combined biological methane potential of the food waste and green waste, in m^3CH_4 tonneVS^{-1}.

VS_{fw} is the volatile solids content of the food waste, in %.

$\dot{m}_{fw}$ is the mass input of food waste, in kg yr^{-1}.

BMP_{fw} is the biological methane potential of the food waste, in m^3CH_4 tonneVS^{-1}.

$\dot{VS}_{tot}$ is the total volatile solids input into the digester, in tonneVS yr^{-1}.

VS_{gw} is the volatile solids content of the green waste, in %.

$\dot{m}_{fw}$ is the mass input of food waste, in kg yr^{-1}.

BMP_{gw} is the biological methane potential of the green waste, in m^3CH_4 tonneVS^{-1}.

$$\dot{m}_{dig.sf} = \dot{m}_{dig} \times DIG_{sf}$$ *Equation A-22*

Where: $\dot{m}_{dig.sf}$ is the solids fraction of the separated digestate, in tonnes yr^{-1}.

$\dot{m}_{dig}$ is the mass of digestate that leaves the digester, in tonnes year^{-1}.

DIG_{sf} is proportion of digestate that is separated off as a 'solid fraction', in % by weight.

$$\dot{m}_{dig.lf} = \dot{m}_{dig} \times DIG_{lf}$$ *Equation A-23*

Where: $\dot{m}_{dig.lf}$ is the liquid fraction of the separated digestate, in tonnes yr^{-1}.

$\dot{m}_{dig}$ is the mass of digestate that leaves the digester, in tonnes year^{-1}.

DIG_{lf} is proportion of digestate that is separated off as a 'liquid fraction', in % by weight.

$$CV_{bg} = CV_{CH4} \times BG_{CH4}$$ *Equation A-24*

Where: CV_{bg} is the calorific value of the biogas, in kWh m^{-3}.

CV_{CH4} is the calorific value of methane, in kWh m^{-3}.

BG_{CH4} is the proportion of methane in the biogas, in %, by volume.

$$\dot{E}_{bg} = Q_{CH} \times CV_{CH4}$$ *Equation A-25*

Where: $\dot{E}_{bg}$ is the energy production rate from biogas, in kWh day^{-1}.

Q_{CH4} is the methane production rate, m^3 day^{-1}.

CV_{CH4} is the calorific value of methane, in kWh m^{-3}.

A number of data assumptions were used in the mass balance (table A-4).

Table A-4: Data assumptions used in the TEA mass balance.

Data	**Unit**	**Value**	**Source**
Food waste BMP	mL CH_4 gVS^{-1}	471	Experimental (table 5-3), (Xu *et al.*, 2018)
Food waste TS	%	24	(Banks *et al.*, 2018)
Food waste VS	%	22	(Banks *et al.*, 2018)
Food waste density	kg/L	0.5	(WRAP NI, 2015)
Food waste VS removal	%	85	(Banks *et al.*, 2018)
Vegetable oil destruction	%	100	Estimate – no reference
Retention time	days	30	(Gerardi, 2003b)
CHP engine electrical efficiency	%	30	(U.S. Environmental Protection Agency, 2019)
CHP engine thermal efficiency	%	45	(U.S. Environmental Protection Agency, 2019)

Appendix B: Techno-economic analysis calculations

B.1 Feedstock storage

The storage volume calculated was just for the food waste. The volume required was calculated using equation B-1 and equation B-2 and the data assumptions (table B-1).

$$V_w = \frac{m_w}{\rho_w}$$ *Equation B-1*

Where: V_w is the volume of waste in m^3 day^{-1}.
m_w is the mass of waste in kg day^{-1}.
ρ_w is the waste density in kg m^3.

$$V_s = \sum V_W \times T_s$$ *Equation B-2*

Where: V_s is the storage volume required in m^3.
V_w is the volume of waste in m^3 day^{-1}.
T_s is the storage time in days.

Table B-1: Data assumptions for food waste storage calculations.

Data	Unit	Value	Source
Food waste bulk density	kg m^{-3}	500	(WRAP, 2010)
Maximum storage time	days	14	Estimated

B.2 Macerator

The macerator was sized by considering the mass of food waste that would be processed each day and how quickly it should be done. If a post-macerator hopper was required (if there was no pre-digester), the size was calculated by assuming that the feed would be kept in the hopper for a maximum of 7 days.

$$Q_M = M_{fw} \times T_M$$ *Equation B-3*

Where: Q_M is the macerator capacity, in kg hr^{-1}.
M_{fw} is the mass of food waste, in kg day^{-1}.
T_M is the maceration time, in hr day^{-1}.

Table B-2: Data assumptions for macerator calculations.

Data	Unit	Value	Source
Operational time	hours	0.5	Estimated.
Post-macerator hopper resting time	days	7	Estimated.

B.3 Pre-digester

The pre-digester was assumed to be a cylinder. The volume was calculated using the food waste volume throughput and the residence time (equation B-4, equation B-5) and the height and surface area were calculated using the standard dimension calculations for a cylinder (equation B-6, equation B-7). The specific heat capacity of the macerated food waste was calculated using an equation from (Törnwall *et al.*, 2017) which calculated the specific heat capacity of digestate, a similar sludge-like material (equation B-8).

$$V_{pd.tot} = V_{pd.w} \times 10\%$$ *Equation B-4*

$$V_{pd.w} = t_{pd.res} \times (Q_{fw} + Q_{vo})$$ *Equation B-5*

Where: $V_{pd.tot}$ is the total volume of the pre-digester, in m^3.
$V_{pd.w}$ is the working volume of the pre-digester, in m^3.
$t_{pd.res}$ is the residence time, in days.
Q_{fw} is the food waste volume loading rate, in m^3 day^{-1}.
Q_{vo} is the vegetable oil volume loading rate, in m^3 day^{-1}.

$$V_{cyl} = \pi r^2 h$$ *Equation B-6*

$$SA_{cyl} = \pi h d + 2\pi r^2$$ *Equation B-7*

Where: V_{cyl} is the total volume of a cylinder, in m^3.
r is the cylinder radius, in m.
h is the cylinder height, in m.
SA_{cyl} is the cylinder surface area, in m^2.
d is the cylinder diameter, in m.

$$SHC_{fw} = (1 - TS_{fw}) \times \left(SHC_{water} + (TS_{fw} \times 1.05)\right)$$ *Equation B-8*

(Törnwall et al., 2017)

Where: SHC_{fw} is the specific heat capacity of food waste, in kJ kg^{-1} $°C^{-1}$.

TS_{fw} is the total solids content of food waste, in %.

SHC_{water} is the specific heat capacity of food waste, in kJ kg^{-1} °C^{-1}.

The overall heat transfer coefficient (U-value) of the pre-digester wall was calculated by calculating the R-value (equation B-9) for both the pre-digester wall, made of stainless steel, and the insulation, and taking the reciprocal of the sum (equation B-10). Using the calculated values, the heat loss rate, in W, was calculated (equation B-11) using the outdoor temperature and the target temperature of the pre-digester (table B-3). The heat input requirement was then calculated using the calculated heat loss and the heat required to increase the temperature of the feed input to the target temperature (equation B-12).

$$R = \frac{x}{k}$$ *Equation B-9*

$$U_{pd} = \frac{1}{R_{wall} + R_{ins}}$$ *Equation B-10*

(Coulson and Richardson, 1999)

Where: R is the thermal resistance of a material, in m^2 K W^{-1}.

x is the thickness of the layer, in m.

k is the thermal conductivity of the material, in W m^{-1} K^{-1}.

U_{pd} is the U-value of the container, in W m^{-2} K^{-1}.

R_{wall} is the thermal resistance of the container wall, in m^2 K W^{-1}.

R_{ins} is the thermal resistance of the container insulation, in m^2 K W^{-1}.

$$Q_{loss} = U \times SA \times \Delta T$$ *Equation B-11*

(Coulson and Richardson, 1999)

Where: Q_{loss} is the heat lost rate, in W.

U is the U-value of the container, in W m^{-2} K^{-1}.

SA is the surface area of the container, in m^2.

ΔT is the temperature difference between the inside and outside of the container, in K.

$$Q_{req} = \frac{Q_{loss} \times 24}{1000} + \left(\frac{\dot{M}_{fw} \times SHC_{fw} \times \Delta T}{3600}\right)$$ *Equation B-12*

(Coulson and Richardson, 1999)

Where: Q_{req} is the heat requirement, in kWh day^{-1}.

Q_{loss} is the heat lost rate, in W.

$\dot{M}_{fw}$ is the mass flow rate of food waste, in kg day^{-1}.

SHC_{fw} is the specific heat capacity of food waste, in kJ kg^{-1} K^{-1}.

ΔT is the difference between the starting and ending temperatures, in K.

24/1000 is the conversion factor from W to kWh day^{-1}.

3600 is the conversion factor from kJ to kWh.

$$E_{el} = (t_{stirrer} \times P_{stirrer}) + (t_{pump} \times P_{pump})$$ *Equation B-13*

Where: E_{el} is the electrical energy use in kWh day^{-1}.

$t_{stirrer}$ is the stirrer operational time, in hr day^{-1}.

$P_{stirrer}$ is the power rating of the stirrer, in kW.

t_{pump} is the feed pump operational time, in hr day^{-1}.

P_{pump} is the power rating of the feed pump, in kW.

The electrical energy requirement of the pre-digester was calculated from the operational time and the power rating of the stirrer and feed pump (equation B-13).

Table B-3: Data assumptions for pre-digester calculations.

Data	Unit	Value	Source
Residence time	days	7	Estimated, maximum.
Tank headspace	% of working volume	10	Estimated.
Food waste particle density	kg m^{-3}	1000	Estimated.
Food waste specific heat capacity (SHC)	KJ kg^{-1} K^{-1}	3.38	(Törnwall *et al.*, 2017)
Vegetable oil density	kg m^{-3}	918	(Esteban *et al.*, 2012).
Feed input temperature	°C	10	Estimated yearly average
Outside temperature	°C	15	Estimated yearly average
Pre-digester target temperature	°C	63	(Thwaites *et al.*, 2015)
Container wall thickness	m	0.05	Estimated.
Insulation thickness	m	0.1	Estimated.
Stainless steel thermal conductivity (k)	W m^{-1} K^{-1}	14.4	(Engineering Toolbox, 2005)
Insulation thermal conductivity (k)	W m^{-1} K^{-1}	0.0975	(Engineering Toolbox, 2003)
Stirrer operational time	hr day^{-1}	8	Estimated.
Stirrer power rating	kW	2.5	Estimated.
Pump power rating	kW	1.5	Estimated.

B.4 Digester

The waste being fed to the digester was assumed to be 100% of the food waste and vegetable oil produced by the organisation. The dimensions were calculated using the feed volume, equation B-6 and equation B-7 and the heat loss and heat requirements were calculated using equations B-9, B-10, B-11 and B-12. The specific heat capacity of digestate was calculated using equation b- and the values predicted by the mass balance.

It was assumed that siting the plant in a greenhouse would reduce the heating requirement of the pre-digester and digester by 49% as this was the experience of a previous similar study in the same climate (Walker *et al.*, 2017). The heat conserved by the greenhouse was investigated as part of the sensitivity analysis.

B.5 Boiler and CHP

The boiler and CHP power ratings were calculated from the biogas energy content, the assumed conversion efficiency and availability (equation B-14, equation B-15, table B-4).

$$P_{th} = \frac{Q_{bg} \times CV_{bg} \times \eta_{blr} \times Av_{blr}}{24}$$

Equation B-14

Where: P_{th} is the thermal power output of the boiler, in kW.

Q_{bg} is the biogas production rate, in $m^3\ yr^{-1}$.

CV_{bg} is the calorific value of the biogas, in kWh m^{-3}.

η_{blr} is the boiler conversion efficiency, in %.

Av_{blr} is the availability of the boiler, in % of total time – this is to account for maintenance time and stoppages.

24 is the conversion from kWh day^{-1} to kW.

$$P_{el} = \frac{Q_{bg} \times CV_{bg} \times \eta_{eng} \times Av_{eng}}{365\ \times 24}$$

Equation B-15

Where: P_{el} is the rated power of the CHP, in $kW_{el.}$

Q_{bg} is the biogas production rate, in $m^3\ yr^{-1}$.

CV_{bg} is the calorific value of the biogas, in kWh m^{-3}.

η_{eng} is the engine conversion efficiency, in %.

Av_{eng} is the availability of the engine, in % of total time – this is to account for maintenance time and stoppages.

365 x 24 is the conversion from kWh to kW.

Table B-4: Data assumptions used in the TEA model boiler and CHP calculations.

Data	Unit	Value	Source
Methane CV	kWh m^{-3}	9.94	(Engineering ToolBox, 2003)
Methane CV	MJ m^{-3}	35.8	(Engineering ToolBox, 2003)

Boiler heat efficiency	%	90	(Pilli *et al.*, 2015)
Boiler availability	%	99	Estimated.
CHP electrical efficiency	%	30	(U.S. Environmental Protection Agency, 2019)
CHP heat efficiency	%	45	(U.S. Environmental Protection Agency, 2019)
CHP availability	%	92	Estimated.

B.6 Biogas upgrading

During upgrading, biogas (around 60% methane) is converted to biomethane (>95% methane), which when used as a fuel is measured in kg. The output of biomethane in kg and its energy value were calculated (equation B-16, equation B-17).

$$\dot{m}_{BM} = \frac{Q_{BG} \times \mathrm{BG}_{CH4} \times \rho_{BM}}{\mathrm{BM}_{CH4}}$$ *Equation B-16*

Where: $\dot{m}_{BM}$ is the mass flow of biomethane, in kg yr^{-1}.

Q_{BG} is the biogas production rate, in m^3 yr^{-1}.

BG_{CH4} is the proportion of methane in the biogas, in %.

ρ_{BM} is the density of the biomethane, in kg m^{-3}, calculated using the methane and carbon dioxide proportions.

BM_{CH4} is the proportion of methane in the biomethane, in %, by volume.

$$E_{BM} = \frac{\dot{m}_{BM} \times \mathrm{CV}_{BM} \times (1 - Lost_{BM})}{\rho_{BM}}$$ *Equation B-17*

Where: E_{BM} is the energy from biomethane, in kWh yr^{-1}.

$\dot{m}_{BM}$ is the mass flow of biomethane, in kg yr^{-1}.

CV_{BM} is the calorific value of the biomethane, in kWh m^{-3}.

$Lost_{BM}$ is proportion of biogas lost in upgrading process, in %.

ρ_{BM} is the density of the biomethane kg m^{-3}.

$$X_{BM} = \dot{m}_{BM} \times \eta_{BM}$$ *Equation B-18*

Where: X_{BM} is the number of fuel miles produced from biomethane, in miles yr^{-1}.

$\dot{m}_{BM}$ is the mass flow of biomethane, in kg yr^{-1}.

η_{BM} is the fuel conversion of the biomethane miles kg^{-1}.

These calculations used the following data assumptions (table b-).

Table B-5: Data assumptions used in the TEA model biogas upgrading calculations.

Data	Unit	Value	Source
Biomethane CV	MJ m^{-3}	35.8	(Engineering ToolBox, 2003)
Biogas methane content	%	55	Estimated from Mass Balance.
Biomethane methane content	%	97	Estimated. (Prodeval, 2020)
Biogas upgrading process losses	%	2	Estimated
Biomethane density	kg m^{-3}	0.7453	Calculated using equation
Biomethane fuel conversion	miles kg^{-1}	10	(Chesshire and Llewellyn, 2012).

B.7 Separator

The energy demand and running time of the separator were calculated, based on the output of digestate from the mass balance and the amount of digestate separated (equation b-, equation b-).

Equation B-19

$$\dot{E}_{el.sep} = \frac{P_{el.sep} \times \dot{m}_{dig}}{\rho_{dig} \times Q_{sep} \times 365}$$

Where: $\dot{E}_{el.sep}$ is the energy demand of the separator, in kWh day^{-1}.
$P_{el.sep}$ is the electrical power demand of the separator, in kW.
Q_{sep} is the flow throughput of the separator, in m^3 hr^{-1}.
$\dot{m}_{dig}$ is the mass of digestate produced, in tonnes yr^{-1}.
ρ_{dig} is the digestate density in tonnes m^{-3}.
365 is the conversion from tonnes yr^{-1} to tonnes day^{-1}.

Equation B-20

$$T_{sep} = \frac{\dot{m}_{dig} \times 60}{\rho_{dig} \times Q_{sep} \times 365}$$

Where: T_{sep} is the separator running time, in minutes day^{-1}.
$\dot{m}_{dig}$ is the mass of digestate produced, in tonnes yr^{-1}.
60 is the conversion of running time from hrs day^{-1} to minutes day^{-1}.
Q_{sep} is the flow throughput of the separator, in m^3 hr^{-1}.
ρ_{dig} is the digestate density in tonnes m^{-3}.
365 is the conversion from tonnes yr^{-1} to tonnes day^{-1}.

B.8 Willow bed

The area of willow bed that was required was calculated, as well as the volume of wood production per year (equation B-21, equation B-22).

$$A_{wb} = \frac{N_{dig} \times Q_{dig} \times 10000}{Abs_{N.willow}}$$ *Equation B-21*

Where: A_{wb} is the area of willow bed required, in m^2.
N_{dig} is the nitrogen content of the digestate, in kgN m^{-3}.
Q_{dig} is the volume of digestate, in m^3 yr^{-1}.
$Abs_{N.willow}$ is the nitrogen absorbance of willow, in kgN Ha^{-1} yr^{-1}.

$$V_{willow} = \frac{A_{wb} \times \rho_{wb.growth}}{t_{wb.grow} \quad \times \rho_{willow}}$$ *Equation B-22*

Where: V_{willow} is the volume of willow produced, in m^3 yr^{-1}.
A_{wb} is the area of willow bed, in m^2.
$\rho_{wb.growth}$ is the growing density of the willow bed, in TDM ha^{-1}.
$t_{wb.growth}$ is the growing time of the willow, in years.
ρ_{willow} is the density of willow, in kg m^{-3}.

Assumptions were made of the willow nitrogen absorbance, the nitrogen content of the digestate, the growing time of the willow bed and the density of the mature willow (table B-6).

Table B-6: Data assumptions used in the TEA model willow bed calculations.

Data	Unit	Value	Source
Willow nitrogen absorbance	kgN Ha^{-1} yr^{-1}	300	(Labrecque, Teodorescu and Daigle, 1997)
Digestate nitrogen content	kgN m^{-3}	10.3	From mass balance.
Willow density	kg m^{-3}		(Eisenbies *et al.*, 2019)
Willow growing time	years	3	(Forest Research, 2019)
Mature willow plant density	T $hectare^{-1}$	30.17	(Labrecque, Teodorescu and Daigle, 1997)

B.9 Composting

The composting processes produced heat, which was calculated in kWh day^{-1} (equation b-). The amount of compost produced by a combination of different waste resources, and the C:N ratio and total solids content was calculated by combining the properties of the different wastes (equation B-32, equation B-24).

$$Q_{comp.sf} = \frac{\dot{m}_{dig.sf} \times TS_{dig.sf} \times E_{out.comp}}{1000 \times 3.6 \times 365}$$ *Equation B-23*

Where: $Q_{comp.sf}$ is the energy flow from the composting process, in kWh day^{-1}.
$\dot{m}_{dig}$ is the mass of digestate produced, in tonnes yr^{-1}.
$TS_{dig.sf}$ is the total solids (dry matter, DM) content of the solids fraction of the digestate, in %.
$E_{out.comp}$ is the average thermal output of composting organic matter, in kJ/kg DM.
1000 is the conversion from kJ to MJ.
3.6 is the conversion from MJ to kWh.
365 is the conversion from kWh yr^{-1} to kWh day^{-1}.

$$TS_{comp} = \frac{(\dot{m}_{dig} \times TS_{dig}) + (\dot{m}_{wc} \times TS_{wc}) + (\dot{m}_{cb} \times TS_{cb})}{\dot{m}_{dig} + \dot{m}_{wc} + \dot{m}_{cb}}$$ *Equation B-24*

Where: TS_{comp} is the total solids content of the compost, in %.
$\dot{m}_{dig}$ is the mass of digestate produced, in tonnes yr^{-1}.
TS_{dig} is the total solids content of the digestate, in %.
$\dot{m}_{wc}$ is the mass of cardboard added, in tonnes yr^{-1}.
TS_{wc} is the total solids content of the cardboard waste, in %.
$\dot{m}_{cb}$ is the mass of wood chips added, in tonnes yr^{-1}.
TS_{cb} is the total solids content of the wood chips, in %.

$$\dot{m}_{wc.c} = \dot{m}_{wc} \times C\%_{wc}$$ *Equation B-25*

$$\dot{m}_{dig.c} = \dot{m}_{dig} \times C\%_{dig}$$ *Equation B-26*

$$\dot{m}_{gw.c} = \dot{m}_{gw} \times C\%_{gw}$$ *Equation B-27*

$$\dot{m}_{cb.c} = \dot{m}_{cb} \times C\%_{cb}$$ *Equation B-28*

$$\dot{m}_{wc.n} = \frac{\dot{m}_{wc.c}}{CN_{wc}}$$ *Equation B-29*

$$\dot{m}_{dig.n} = \frac{\dot{m}_{dig.c}}{CN_{dig}}$$ *Equation B-30*

$$\dot{m}_{cb.n} = \frac{\dot{m}_{cb.c}}{CN_{cb}}$$ *Equation B-31*

$$CN_{tot} = \frac{\dot{m}_{wc.c} + \dot{m}_{dig.c} + \dot{m}_{cb.c}}{\dot{m}_{wc.n} + \dot{m}_{dig.n} + \dot{m}_{cb.n}}$$ *Equation B-32*

Where: $\dot{m}$ is the mass input, in kg yr^{-1}.

C% is the percentage of carbon present, by mass.

CN is the carbon to nitrogen ratio, by mass.

$_{wc}$ denotes wood chip.

$_{dig}$ denotes digestate.

$_{cb}$ denotes cardboard (shredded).

$_{c}$ denotes carbon.

$_{n}$ denotes nitrogen.

$_{tot}$ denotes total (or combined).

The values for C:N ratio and moisture content were used to work out the optimal mixture ratio of cardboard and wood chips with the digestate. In the waste statistics available, card was often included as part of ‘mixed recyclables’ and the weight of the card itself was not recorded. In UK recycling statistics (Department for Environment and Rural Affairs (DEFRA), 2019), card made up 47% of mixed recycling by mass (figure B-1). The amount of card in mixed recycling was therefore estimated based on this figure.

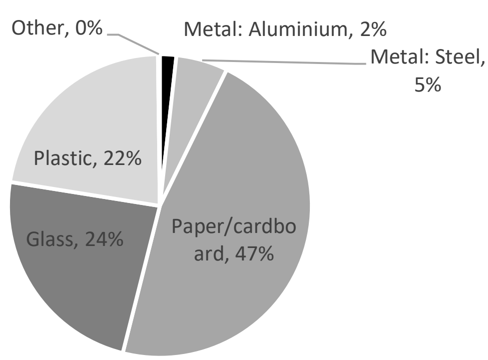


Figure B-1: Mixed recycling proportions by weight, derived from (Department for Environment and Rural Affairs (DEFRA), 2019).

When composting at this scale, the compost heap would take up a large volume, which had to be calculated (equation B-35) to understand how much space would need to be made available and so that a composting vessel could be sized to estimate the potential cost (equation B-34).

$$\dot{V}_{tot} = \left(\frac{\frac{\dot{m}_{wc}}{\rho_{wc}} + \frac{\dot{m}_{dig}}{\rho_{dig}} + \frac{\dot{m}_{gw}}{\rho_{gw}} + \frac{\dot{m}_{cb}}{\rho_{cb}}}{365} \right)$$

Equation B-33

Where: $\dot{V}_{tot}$ is the volume of materials to be composted, in $m^3\ day^{-1}$.

$\dot{m}$ is the mass input, in tonnes yr^{-1}.

ρ is the density, in tonnes m^{-3}.

t is the compost storage time, in days.

$_{wc}$ denotes wood chip.

$_{dig}$ denotes digestate.

$_{gw}$ denotes green/garden waste

$_{cb}$ denotes cardboard (shredded)

Equation B-34

$$V_{composter} = \dot{V}_{tot} \times t_{comp.make}$$

Where: $V_{composter}$ is the composter volume required, in m^3.

$\dot{V}_{tot}$ is the volume of materials to be composted, in $m^3\ day^{-1}$.

$t_{comp.make}$ is the time taken to compost, in days.

Equation B-35

$$V_{stor} = \dot{V}_{tot} \times \left(1 - R_{comp}\right) \times t_{comp.stor}$$

Where: V_{stor} is the composting volume requirement, in m^3.

$\dot{V}_{tot}$ is the compost production volume, in $m^3\ day^{-1}$.

R_{comp} is the reduction in volume during the composting process, in %.

$t_{comp.stor}$ is the compost storage time, in days.

The composting process reduces the volume by half (Breitenbeck and Schellinger, 2004; Cooperband, 2000) so the space required for the finished compost was calculated using this assumption.

Table B-7: Data assumptions used in the TEA model composting calculations.

Data	Unit	Value	Source
Average thermal output of composting organic matter	kJ $kgDM^{-1}$	4302	(Smith, Aber and Rynk, 2017)

Composting process volume reduction	%	50	(Breitenbeck and Schellinger, 2004)
Density of wood chips	T m^{-3}	0.38	(Aqua-Calc, 2019)
Density of digestate	T m^{-3}	0.99	(WRAP, 2011)
Density of green waste	T m^{-3}	0.25	(Levis and Barlaz, 2011)
Density of shredded cardboard	T m^{-3}	0.3	(Aerobelt Australia Pty Ltd, 2019)
Time taken to make compost	days	20	(Irvine, Lamont and Antizar-Ladislao, 2010)
Maximum storage time needed for compost	days	180	Estimated